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Background Paper

Population Status and Threats to Albatrosses and Giant Petrels Listed as Threatened under the *Environment Protection and Biodiversity Conservation Act 1999*

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Cover photograph: Light mantled albatross and chick, North Head, Macquarie Island, 2010; photographer Sarah Way, Tasmanian Department of Primary Industry, Parks, Water and Environment.

CONTENTS

Summary of threats and Issues

Albatrosses and giant petrels face a range of threats in the marine environment and on land. At sea, apart from natural variations in ocean productivity, threatening factors include direct interactions with fishing operations; ingestion of, and entanglement in, marine debris; contamination from pollutants; and over-fishing of prey species. Access to bait, and bycatch and offal discarded by commercial fishers may alter natural foraging behaviour and increase the level of habituation. At some breeding colonies, predation by introduced species has increased mortality and decreased breeding success; degradation of nesting habitat by introduced herbivores, interspecific competition for nest space, and transmission of parasites and disease also occurs. Of these threats, increased mortality among juveniles and adults resulting from interactions with fishing operations is particularly significant.

Ensuring the long-term survival of Australia's albatross and giant petrel populations depends on domestic research and conservation management programs, combined with international action to protect these highly migratory seabirds during the extensive time they spend foraging in the waters of other countries or on the high seas.

Major threats to albatrosses and giant petrels

At-sea threats	Land-based threats
• Incidental catch in longline fisheries • Incidental catch in trawl fisheries • Incidental catch during driftnetting • Incidental catch in trolling operations • Intentional shooting / killing • Competition with fisheries for marine resources • Dependence on discards • Marine pollution • Climate change	• Introduced species • Human disturbance at the nest • Parasites and diseases • Loss of nesting habitat • Competition for nest space

Current understanding of the at-sea distribution and threats facing albatrosses and giant petrels is limited. Further assessment of foraging ranges and dietary requirements of populations of albatrosses and giant petrels is needed. Notwithstanding the present limitations of such information, it is clear that much greater consideration of it is needed when management arrangements for fisheries are being developed or revised by management authorities. Research and management must continue to develop sustainable measures to effectively mitigate against incidental mortality in long-line and other fisheries.

While there has been an increased focus on seabird mortality arising from longline fisheries in recent years, more data are required on the rates, causes, and factors contributing to mortality of albatrosses and giant petrels as a result of trawling operations. Gathering this information is likely to require additional dedicated scientific observer programs and examination of logbook databases. Additional research on the provenance and other characteristics of birds killed in such fisheries would also be valuable.

Over-harvesting of fish and squid species is a global problem that may generate one of the greatest threats to albatrosses and giant petrels by altering the ecosystem balance. For Australian fisheries, the Australian Fisheries Management Authority is required under the

Fisheries Management Act 1991 and the *Environment Protection and Biodiversity Conservation Act 1999* to manage fisheries in an ecologically sustainable manner and to have regard for non-target species. These controls are continually being improved and, while more work is needed, they are potentially effective. However, because of the migratory and or straddling nature of seabirds and target stocks, similar controls on high seas fisheries are also vital. Currently high seas controls vary considerably from good to non-existent.

As albatrosses and giant petrels are long-lived, slow-reproducing birds, long-term population monitoring is essential in understanding population dynamics. Maintaining long term population and demographic monitoring programs on albatross and giant petrel populations is important as they are essential for detecting population changes, enabling management responses to be put in place to arrest declines.

Diseases known to affect seabird populations need to be investigated to understand disease determinants and develop mitigation and potential recovery actions. In particular, the avian pox virus infecting shy albatrosses on Albatross Island needs further investigation.

Management actions need to focus on elimination of introduced species that affect breeding seabirds, as well as imposing stringent quarantine requirements for the prevention of exotic introductions, especially to islands that are currently pest-free.

Marine pollution is becoming increasingly apparent in the Southern Hemisphere and its impact on top-level predators, like albatrosses and giant petrels, is likely to increase in the future. The incidence of ingestion of plastic and its impacts as well as the incidence and level of marine contaminants, such as organochlorins and other toxins, needs to be further investigated for Australian taxa. Marine pollution is a global phenomenon that needs to be addressed through both national and international conservation fora.

The future prospects for many albatross and giant petrel populations are uncertain. Only a thorough understanding of albatross and giant petrel ecology coupled with the much greater application of key management measures, particularly the widespread adoption of effective mitigation measures for longline and trawl fisheries and protection and rehabilitation of their breeding habitats, will ensure their long-term survival.

1 INTRODUCTION

1.1 Purpose

The *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) provides a comprehensive legislative framework to protect Australia's marine environment. A list of threatened species has been established under Part 13 of the Act. Species on this list are considered to be either extinct in the wild, critically endangered, endangered, vulnerable, or conservation dependent. Listed threatened species are protected to help ensure their long-term survival.

The EPBC Act provides for recovery plans for the protection, conservation and management of listed threatened species. Recovery plans must set out the recovery objectives and the actions required to achieve those objectives, including performance indicators and responsibilities for implementation of the actions and timeframes involved.

The majority of the world's albatross species occur in areas under Australian jurisdiction. Hence, Australia has a responsibility for their protection both nationally, under the EPBC Act and State and Territory legislation, and internationally, under agreements such as the Convention on the Conservation of Migratory Species of Wild Animals (CMS).

This Background Paper updates relevant information on the biology and ecology of Australia's albatrosses and giant petrels, identifies issues and threats to these species, and also appropriate management strategies. It will inform the updating of the five-year *National Recovery Plan for Albatrosses and Giant Petrels* (2011). In all, 21 species—19 species of albatross and both of the giant-petrels—have been considered in this paper (Table 1.1).

1.2 Overview of Status

1.2.1 Conservation status of albatrosses and giant petrels

The taxonomy of albatrosses (Family Diomedeidae) has been controversial for many years; it remains a work in progress and continued further development is expected. While a significant amount of new taxonomic information has become available since the 2001 Recovery Plan was finalised, including from genetic studies, this has not resulted in a resolution of the differing views on what is the most appropriate taxonomy. For a variety of reasons – including its international standing, use of the most recent data and review processes – and without wishing to stimulate unproductive taxonomic debate, this document uses the taxonomy adopted by the Agreement on the Conservation of Albatrosses and Petrels (ACAP) and indicates areas of debate as appropriate. The use of the ACAP taxonomy does not substantially or practically alter the conservation actions and management priorities contained in this Plan.

Twenty of the 22 species of albatross in the world occur within the Southern Hemisphere (Table 1.2), 18 of which have been confirmed to occur within the Australian Exclusive Economic Zone (EEZ). Another species, Amsterdam albatross, has not been positively identified within the EEZ, however, the extent of its known distribution suggests that it is possible, or even likely, that some vagrants enter Australian waters. Thus, this species is considered to 'potentially occur' within the EEZ. The remaining three species (waved, short-tailed and black-footed albatrosses) do not occur within the EEZ. There are two giant petrel species (Family Procellariidae) in the world, both of which forage and breed within the Australian EEZ.

In all, 21 species (19 albatross species and both giant petrel species) have been considered in the preparation of this Background Paper (Table 1.1). These have been categorised as:

(i) ‘breeding species’: species that breed on islands in areas under Australian jurisdiction (seven species); and

(ii) ‘foraging species’: species that forage (or potentially forage), but do not breed, within areas under Australian jurisdiction (14 species).

Table 1.1: Albatross and giant petrel species or subspecies considered in this report

Breeding in Australian jurisdictions	Foraging in Australian jurisdictions
Wandering albatross *Diomedea exulans*	Tristan albatross *D. dabbenena*
Black-browed albatross *Thalassarche melanophris*	Antipodean albatross *D. antipodensis*[1]
Shy albatross *T. cauta*	Northern royal albatross *D. sanfordi*
Grey-headed albatross *T. chrysostoma*	Southern royal albatross *D. epomophora*
Light-mantled albatross *Phoebetria palpebrata*	Amsterdam albatross *D. amsterdamensis*
Northern giant petrel *Macronectes halli*	Laysan albatross *D. immutabilis*
Southern giant petrel *M. giganteus*	Campbell albatross *T. impavida*
	White-capped albatross *T. steadi*
	Chatham albatross *T. eremita*
	Salvin’s albatross *T. salvini*
	Atlantic yellow-nosed albatross *T. chlororhynchos*
	Indian yellow-nosed albatross *T. carteri*
	Buller’s albatross *T. bulleri*[1]
	Sooty albatross *P. fusca*

[1] A recent assessment of published molecular, morphometric and other characters determined that two species pairs should be regarded as subspecies: Antipodean albatross *Diomedea antipodensis antipodensis* and Gibson’s albatross *Diomedea antipodensis gibsoni;* and Buller’s albatross *Thalassarche bulleri bulleri* and Pacific albatross *Thalassarche bulleri platei* (Double 2006).

Five of these albatross species are listed as nationally endangered, and twelve are listed as nationally vulnerable (Table 1.2). Only three albatrosses are not listed as threatened under the EPBC Act—the Laysan, light-mantled and Atlantic yellow-nosed albatrosses—although all are listed under the IUCN Red List as endangered or near threatened globally. Both giant petrels are listed nationally as threatened: one as endangered, one vulnerable.

Five albatross species breed on islands within Australian waters (Table 1.2), and one of these, the shy albatross, is an endemic breeding species to Australia. That is, this species only breeds in Australian waters. All five species are protected and several are listed as threatened under State or Territory legislation (Table 1.3).

All albatross species occurring within areas under Australian jurisdiction and both giant petrels are listed in the appendices of the Convention on the Conservation of Migratory Species of Wild Animals (CMS) and on Annex 1 to the Agreement on the Conservation of Albatrosses and Petrels (ACAP).

Table 1.2: National and international conservation status of albatrosses and giant petrels

Species	Forages in the Southern Hemisphere	Forages in areas under Australian jurisdiction	Breeds in areas under Australian jurisdiction	Australian endemic	Listing under EPBC Act (1999)	International conservation status (criteria)[1]
(i) Species which forage and breed in areas under Australian jurisdiction						
Wandering albatross	√	√	√		Vulnerable	Vulnerable
Black-browed albatross	√	√	√		Vulnerable	Endangered
Shy albatross	√	√	√	√	Vulnerable	Near threatened
Grey-headed albatross	√	√	√		Endangered	Vulnerable
Light-mantled albatross	√	√	√			Near threatened
Northern giant petrel	√	√	√		Vulnerable	Least concern
Southern giant petrel	√	√	√		Endangered	Least concern
(ii) Species which forage but do not breed in areas under Australian jurisdiction						
Tristan albatross	√	√			Endangered	Critically endangered
Antipodean albatross[2]	√	√			Vulnerable	Vulnerable
Gibson's albatross[2]	√	√			Vulnerable	Vulnerable
Northern royal albatross	√	√			Endangered	Endangered
Southern royal albatross	√	√			Vulnerable	Vulnerable
Amsterdam albatross	√	√			Endangered	Critically endangered
Laysan albatross	√	√				Near threatened
Campbell albatross	√	√			Vulnerable	Vulnerable
Buller's albatross[2]	√	√			Vulnerable	Near threatened
Pacific albatross[2]	√	√			Vulnerable	Near threatened
White-capped albatross	√	√			Vulnerable	Near threatened)
Salvin's albatross	√	√			Vulnerable	Vulnerable
Chatham albatross	√	√			Endangered	Vulnerable
Atlantic yellow-nosed albatross	√	√				Endangered

Indian yellow-nosed albatross	√	√			Vulnerable	Endangered
Sooty albatross	√	√			Vulnerable	Endangered
(iii) Species which do not occur in areas under Australian jurisdiction[3]						
Waved albatross	√					Critically endangered
Short-tailed albatross						Vulnerable
Black-footed albatross						Endangered

[1] IUCN (2010). *2010.4 IUCN Red List of Threatened Species*. <www.iucnredlist.org>. Downloaded 18 November 2010.

[2] A recent assessment of published molecular, morphometric and other characters determined that two species pairs should be regarded as subspecies: Antipodean albatross *Diomedea antipodensis antipodensis* and Gibson's albatross *Diomedea antipodensis gibsoni*; and Buller's albatross *Thalassarche bulleri bulleri* and Pacific albatross *Thalassarche bulleri platei* (Double 2006). They are treated as such in this document, but the subspecies are shown separately in this table to indicate their current conservation status under Australian legislation.

[3] The waved albatross is confined to the east Pacific Ocean north of 12°S, and the short-tailed albatross and black-footed albatross occur only within the Northern Hemisphere.

Table 1.3: Conservation status under State, Territory and Commonwealth legislation of albatrosses and giant petrels breeding in areas under Australian jurisdiction. Where a species is not listed as threatened, all are fully protected in all Australian States and Territories.

	Wandering albatross	Black-browed albatross	Shy albatross	Grey-headed albatross	Light-mantled albatross	Southern giant petrel	Northern giant petrel
Tas	Endangered	Endangered	Vulnerable	Endangered	Vulnerable	Vulnerable	Rare
Vic	Threatened	Threatened	Threatened	Threatened		Threatened	Threatened
NSW	Endangered	Vulnerable	Vulnerable			Endangered	Vulnerable
Qld						Endangered	Vulnerable
SA	Vulnerable		Vulnerable	Vulnerable			
WA	Vulnerable	Vulnerable	Vulnerable	Vulnerable	Vulnerable		
NT							
Comm.	Vulnerable	Vulnerable	Vulnerable	Endangered		Endangered	Vulnerable

1.2.2 Status of breeding populations under Australian jurisdiction

There are about 150 breeding populations of albatross around the globe. Many have not been surveyed for several years and/or they have not been surveyed systematically. Of the 53 populations for which the status was known and reviewed by Gales (1998), almost half were decreasing. The situation becomes more serious when population size is taken into account, as it is typically the large populations which are decreasing or of unknown trend (reviewed in Gales 1998).

For the populations of northern giant petrels for which information is available, three are increasing and one is stable. For southern giant petrels some populations are increasing, whilst others are stable or decreasing.

Eighteen breeding populations of albatrosses and giant petrels occur within areas under Australian jurisdiction (Table 1.4). The status of albatrosses and giant petrels breeding on Macquarie Island, Albatross Island, The Mewstone and Pedra Branca are monitored annually as part of long-term conservation projects. Eight of these colonies contain 200 or fewer breeding pairs.

All breeding islands under Australian jurisdiction (Macquarie Island, Albatross Island, the Mewstone, Pedra Branca, Heard and McDonald Islands, and islands within the Australian Antarctic Territory) are protected according to their status as Nature Reserves, National Parks or World Heritage Areas (Appendix 2), or a combination of these statuses.

Table 1.4: Most recent population estimates of Australian breeding populations of albatrosses and giant petrels. (More detailed information and the source of these data can be found in Section 3.)

Species	Breeding locality	Current monitoring program (Date commenced)	Survey date	Annual no. breeding pairs	Population trend
Wandering albatross	Macquarie Island	Yes (1994)	2010	4	Stable/Declining?
	Heard Island	No	1967	1	
Black-browed albatross	Macquarie Island	Yes (1994)	2010	57	Stable
	Bishop and Clerk Islets	No	1993	141	?
	Heard Island	No	2000	600	Stable?
	McDonald Islands	No	1981	82 – 89	?
Shy albatross	Albatross Island	Yes (1975)	2009	5 233	Stable
	The Mewstone	Yes (1975)	2005	9 000 -11 000	?
	Pedra Branca	Yes (1975)	2007	circa 220	Decreasing
Grey-headed albatross	Macquarie Island	Yes (1994)	2010	108	Stable
Light-mantled albatross	Macquarie Island	Yes (1994)	2005	1 281[1]	Stable
	Heard Island	No	1954	200 – 500	?
	McDonald Islands	No	?	?	?
Northern giant petrel	Macquarie Island	Yes (1994)	2009	1 689	Stable
Southern giant petrel	Macquarie Island	Yes (1994)	2010	2 534	Stable
	Heard Island	No	1988	3 150	Stable
	McDonald Island	No	1979	1 400	?
	AAT:				
	- Giganteus Island	Opportunistic	2007	3-4	Stable?
	- Hawker Island	Opportunistic	2010	45[2]	Stable?
	- Frazier Islands	Opportunistic	2001	~250	Increasing?

? Population trend is unknown due to a lack of recent or consistent population censuses

[1] Annual monitoring continues at 9 breeding sites. Light mantled albatross breed at intervals ranging from 2 to 4 years and not all sites are accessible, total breeding population probably ranges from around 1,000 to 2,000 annual breeding pairs. Population trends have been determined from the monitored sites which have been largely stable.

[2] Based on preliminary analysis of photographs taken by automated cameras during 2009/10.

2 GENERIC THREATS AND ISSUES

Throughout all stages of their life history, albatrosses and giant petrels are subject to an array of threats, both at sea and on land, that are reducing their survivorship and/or their capacity to reproduce successfully. In addition, significant modification of their foraging habitat (that is, the world's oceans) may also be limiting populations. In combination, these factors are putting the long-term viability of many species at risk.

The life history strategies of albatrosses and giant petrels are a major factor influencing their conservation status. Their strategy for survival is based on low natural adult mortality (in the order of 4-5%), deferred sexual maturity, low reproductive output, often lifelong pairing bonds, relatively high breeding success and a long lifespan. Consequently, populations may be imperilled by even small increases in the rate of mortality (Croxall *et al.* 1990). Furthermore, the breeding season of albatrosses and giant petrels are typically exceptionally long. During this time, the death of one parent also results in the death of the dependent offspring, further jeopardising population viability (Weimerskirch and Jouventin 1987; Croxall *et al.* 1990).

There are several areas of albatross and giant petrel biology and ecology that are not well known. While these are "issues" rather than "threats" as such, they are vital to interpretation and measurement of the likely impact of threats and are important for population viability analysis and other modelling. Despite considerable work in recent years, particularly in studies of foraging distribution (BirdLife 2004b), details of the breeding biology, feeding ecology, foraging distribution and population trend of many species are still lacking (Baker *et al.* 2002).

Most species typically breed on remote oceanic islands. Several islands within Australia's EEZ contain breeding populations of albatrosses and/or giant petrels: specifically, Macquarie Island, Bishop and Clerk Islets, Heard Island, the McDonald Islands, Albatross Island, Pedra Branca, and the Mewstone, as well as Giganteus Island, Hawker Island and the Frazier Islands within the Australian Antarctic Territory (AAT). These islands are critical to the survival of the Australian 'breeding species' of albatrosses and giant petrels and can be regarded as habitat critical to the survival of the species.

The threats and issues that follow in this section are grouped according to subject matter, and do not necessarily appear in order of importance.

2.1 Incidental Catch During Longline Fishing Operations

Each year many thousands of albatrosses and giant petrels are accidentally killed on longline hooks when birds, attracted to fishing vessels by discards and baits, ingest baited hooks and subsequently drown (Baker *et al.* 2002). While most mortality occurs directly when birds are caught during line-setting and, less commonly, hauling, albatrosses and giant petrels may also die after they are released with critical injuries (Huin and Croxall 1996), or following ingestion of fishing hooks when birds eat discarded baits and fish heads containing hooks. In most cases, the death of breeding adults will lead to the subsequent death of dependent chicks.

The level of longline-related mortality is such that longline fishing has been identified as a major threat affecting albatrosses (Gales 1998) and giant petrels (Patterson and Hunter 1998), causing widespread declines in populations throughout the world (Alexander *et al*. 1997; Birdlife International 1995; Croxall 1998; Delord *et al*. 2005; Gales 1998; Nel *et al.* 2002; Poncet *et al.* 2006; Tuck *et al*. 2001). All five species of albatross breeding in Australian waters and both species of giant petrel are seriously threatened by longline fishing (Gales and Brothers 1996; Gales 1998; Environment Australia 2001), as are most of the species that forage within Australia's EEZ (Gales 1998). In addition, it is likely that most or all albatross species also suffer from ingestion of fishing equipment.

Australia was quick to realise the threat that longline bycatch posed to the conservation of seabirds. The incidental catch (or bycatch) of seabirds during oceanic longline fishing operations was listed as a key threatening process on 24 July 1995. As required under Commonwealth legislation (now the *Environment Protection and Biodiversity Conservation Act 1999* — EPBC Act), a Threat Abatement Plan (Longline TAP) was prepared to manage this threat in 1998 (Environment Australia 1998), and subsequently reviewed in 2006 (Department of Environment and Heritage 2006). The Longline TAP sets out to coordinate national action to alleviate the impact of longline fishing activities on all seabirds in Australian waters. It applies to all fisheries under Commonwealth jurisdiction and sets maximum bycatch rates, ranging from 0.01 to 0.05 birds per thousand hooks for longline fisheries known to impact seabirds, as criteria upon which the success of the Longline TAP can be measured.

In some fisheries, such as the Antarctic longline fisheries operating around Macquarie, Heard and McDonald Islands, bycatch limits have been established by the Australian Fisheries Management Authority as a precautionary measure to ensure the impact of bycatch on the small populations of albatrosses and other seabirds that breed in these areas is minimised.

At Heard Island and McDonald Islands (HIMI), a bycatch limit of 10 seabirds per year applied to each longline fishing operator that fished during the longline trial period of 2003-2005. Since then, the Longline TAP rule of 0.1 birds per 1000 hooks has applied. The number of vessels able to fish is also limited. At Macquarie Island, during the four year longline fishing trial from 2007 to 2010, a risk assessment approach was adopted that establishes bycatch limits based on the regional, rather than global, conservation status of seabirds (Hewitt and Hay 2007). The approach was developed for a trial of longline fishing in the Macquarie Island toothfish fishery in 2007. Seabird bycatch limits categorised seabirds into three groups of species with a different limit for each group. The groupings reflected the varying conservation status of the seabird populations breeding on Macquarie Island, the very small size of some breeding populations and their vulnerability to fisheries interactions. The group containing those species with the most critical conservation status and highest risk of interacting with fishing operations (wandering albatross, black-browed albatross, grey-headed albatross, grey petrel and soft-plumaged petrel) had a bycatch limit of one seabird; limits on the other categories were two (southern giant petrel, northern giant petrel, light-mantled albatross and blue petrel) and three individuals (all other seabirds) respectively. In addition, if a total of three seabirds from categories 1 to 3 were killed as a result of interactions with fishing gear, then longline fishing would have ceased for the remainder of the season.

The inclusion of regional information for areas where populations of threatened species are extremely small was considered to have merit by CCAMLR's specialist ad hoc Working Group on Incidental Mortality Associated with Fishing (SC-CAMLR 2007).

The longline fisheries operating at HIMI and Macquarie Island also require fishing vessels to adopt a comprehensive suite of bycatch mitigation measures, as well as carrying an independent scientific observer on all trips. At HIMI, where vessels are also subject to the requirements of CCAMLR, fishing is restricted to a season that falls outside the breeding season of albatrosses and giant petrels (May to September). In addition, vessels must use integrated weight longlines with a minimum sink rate to 20 m depth of 0.2 m/s, or unweighted longlines and external weights with a minimum sink rate of 0.3 m/s; twin streamer lines with at least 60 m aerial coverage; use of an effective deterrent device during line hauling operations (e.g. curtain of streamers around hauling area); and to retain all non-target fish bycatch, fish offal and discards onboard. Since 2005 there has been no restriction on the time of day that vessels may set and haul gear and the fishing season has been progressively extended, subject to a bird bycatch limit of three birds, to include two weeks in April, all of September and two weeks in October.

At Macquarie Island the longline trial in 2007-2010 proceeded with fishing subject to a range of conditions designed to minimise or avoid seabird interactions and bycatch. These included restricting the trial to only one longline vessel; no offal discharge in the fishery; night setting only; use of CCAMLR integrated weight longlines; carriage of two scientific observers; paired streamer lines during setting; conducting sink rate trials prior to commencement of fishing and subsequent trials on a weekly basis; use of a 'brickle curtain' during hauling to avoid bycatch on hauling; minimisation of deck lighting; and an initial season of 1 May to 31 August. As noted above, seabird bycatch limits apply to both the HIMI and Macquarie Island longline fisheries. There have been two incidents in the HIMI Fishery involving giant petrels as a result of interactions with fishing gear (23 August 2008 and 20 May 2009); three cape petrels were caught in separate incidents in June 2010 indicating the need for continued vigilance in the practical implementation of mitigation measures. There have been no reported incidents in the Macquarie Island Fishery.

Implementation of the Longline TAP has resulted in reduced levels of bycatch of albatrosses and giant petrels, and other seabirds, with bycatch rates generally less than 0.05 seabirds per 1000 hooks in all Australian longline fisheries (Australian Fisheries Management Authority, unpublished). However, fishing gear and practices are constantly changing and annual monitoring of bycatch levels in Australian longline fisheries, a requirement of the Longline TAP, is necessary to ensure the downward trend in bycatch of seabirds is maintained (Department of Environment and Heritage 2006).

While there has been an attempt in recent years to address incidental bycatch in some longline fisheries elsewhere in the world (e.g. SC-CAMLR 2005, WCPFC 2006 and 2007, IOTC 2008 and 2010), much remains to be done to address this threat. High levels of bycatch are still recorded in most ocean basins (FAO 1999). The key management regimes for addressing seabird bycatch on these oceanic scales are Regional Fishery Management Organisations (Small 2005) although much of the bycatch still occurs within the EEZs of many albatross range States. There are few comprehensive studies that quantify bycatch in either national or RFMO fisheries. Baker *et al*. (2007a) estimated that 500—600 shy and white-capped albatrosses were killed in the South African Pelagic Longline Fishery in 2005. Petersen *et al*. (2007) estimated the pelagic and demersal longline fisheries operating in South African, Namibian and Angolan waters killed more than 1 334 albatrosses each year, comprising most commonly white-capped albatross (>899 p.a.), as well as more than 203 Atlantic yellow-nosed albatrosses and 58 black-browed albatrosses. Other major 'problem' fisheries for albatrosses and giant petrels are the demersal and pelagic fisheries off the Atlantic coast of South America (Bugoni & Neves 2007; Jimenez *et al.* 2006; Neves *et al.* 2006), and the Japanese pelagic tuna longline fisheries of the Southern Ocean (Kiyota and Takeuchi 2004; SC-CAMLR 2005). However, data on the incidental catch of seabirds are lacking for most longline fisheries, especially those conducted on the high seas, including South Pacific fisheries, and particularly in the Humboldt Current region (although see Moreno 2006); Korean and Taiwanese pelagic tuna longline fisheries of the Southern Hemisphere (SC-CAMLR 2005); pelagic fisheries operating in tropical waters of all oceans (Brothers *et al.* 1999); and Spanish distant water pelagic longline fisheries (A. Black and C. Small, unpublished).

A range of mitigation measures for reducing the incidental catch of seabirds in longline fisheries have been developed (Brothers *et al.* 1999; Dietrich *et al.* 2004; Bull 2007) that can be employed according to circumstance. They include night setting; line weighting; seasonal and/or area closures; bird scaring lines; and avoiding or controlling offal discharge, especially in demersal longline fisheries. Bait thawing was historically thought to be critical to achieving fast sink rates but has since been shown to make little difference once the bait is sufficiently thawed to insert the hook (Robertson and van den Hoff 2010). These measures focus on reducing bycatch during the critical period of setting following release of the bait from the stern of the longline vessel until it has sunk out of reach of diving seabirds by increasing the sink rate of bait; deterring birds from foraging where baits are being set; blocking access to baits, and minimising the congregation of seabirds around vessels. Each has different attributes, costs and potential to successfully reduce seabird catch. However, in most longline fisheries, the greatest reduction in bycatch comes from using a combination of measures. Some measures such as night-setting and line weighting have been consistently successful in a number of longline fisheries (Baker and Wise 2005; Gales *et al*. 1998; Gilman *et al*. 2005; Klaer and Polacheck 1997; McNamara *et al*. 1999; SC-CAMLR 2005; Robertson *et al*. 2010), while the effectiveness of others has varied between vessels and seabird species (ACAP Seabird Bycatch Working Group 2007).

While considerable progress has been made in mitigating bycatch in demersal longline fisheries (e.g. Moreno *et al*. 2007), principally through the development of effective bird scaring lines (Melvin 2003; Melvin *et al.* 2004), integrated weight line in autoline systems (Robertson *et al*. 2006), night setting and seasonal closures (SC-CAMLR 2005), proven and accepted seabird avoidance measures in pelagic fisheries require substantial improvement. In 2007, ACAP's Seabird Bycatch Working Group reviewed available research on seabird bycatch mitigation measures for pelagic longline fishing (ACAP Seabird Bycatch Working Group 2007; also see Melvin and Baker 2006). Development is currently underway on a number of mitigation measures for this gear type, with bird scaring lines, an underwater bait setting capsule and side setting assessed as being the highest priority for research. Other measures considered priorities for research include weighted branchlines, a bait pod, smart hooks, circle hooks and blue dyed squid. Night setting and line weighting are currently the only mitigation measures proven to be widely effective with pelagic longline gear, but widespread adoption of night setting is constrained because it is considered to reduce operational efficiency when targeting some pelagic fish species and there is sporadic resistance to use of line weighting due to crew safety concerns. The ACAP Seabird Bycatch Working Group (SBWG) prepared revised best practice advice for all major fishing methods in April 2010 (see Report of SBWG3 at www.acap.aq). Best practice mitigation for pelagic longline fisheries consists of appropriately weighted branch lines in combination with bird scaring lines (also called streamer or tori lines) and night setting.

To understand the conservation implications of fisheries bycatch mortality on albatrosses and giant petrels, improved knowledge of the level of bycatch in all major fisheries known to kill these birds is urgently needed. For many of the world's fisheries, independent observer coverage is either non-existent or falls below the level required to accurately estimate bycatch levels (Baker *et al.* 2007a; Small 2005). It is also important that fisheries observers retain all seabirds killed in fishing operations and return carcases for analysis to determine species, age, sex, breeding status and, where possible, provenance. This is a mandatory requirement of the Longline TAP, and is essential in assessing risk to species and improving knowledge of fishery impacts (Department of Environment and Heritage 2006; Gales 1998; Abbott *et al*. 2006a).

Bycatch risk assessments for all fisheries need to be developed and regularly reviewed. Spatio-temporal effort in fisheries is dynamic and fluctuates in response to market forces and the status of target stocks. Changes in effort or how fishing gear is rigged can rapidly change the impact upon bycatch species. Waugh *et al.* (2007) recognised several factors as pivotal to the highly effective management of seabird bycatch in CCAMLR longline fisheries that have wider application to fisheries management throughout the world. Primary among these is the detailed annual review of information on fishery performance, the seabird species that interact with the fishery, and improvements in bycatch mitigation practice. This has resulted in regular revision to conservation measures, ensuring that the mitigation measures are close to international best-practice for the fishery at any time. Risk assessments should also consider establishing bycatch limits in fisheries and areas adjacent to breeding colonies of albatrosses and giant petrels with small populations (Hewitt and Hay 2007), as discussed above. This has been a standard practice in new and developing fisheries in Antarctic waters (CCAMLR 2006).

An essential tool in reducing bycatch in fisheries is the establishment of a working group that meets regularly to consider all aspects of interactions with seabirds in a fishery (refer also Section 2.16.iii). Perhaps the best example of how to successfully manage bycatch in a fishery is that of CCAMLR, which has seen seabird bycatch virtually eliminated in its Antarctic longline fisheries over the last 10 years (SC-CAMLR 2007). Pivotal to the success of CCAMLR in reducing seabird bycatch has been the introduction of scientific observers on every longline fishing vessel and the formation of the specialist ad hoc Working Group on Incidental Mortality Associated with Fishing (IMAF). This group was established in 1994 to specifically provide advice to CCAMLR's Scientific Committee on seabird interactions. The group meets annually and reviews all fishing data from the previous year, together with fishing proposals, including those seeking to use new methods or fish in new areas using existing methods, for the forthcoming year.

Matters considered by IMAF include the performance of each fishing vessel in avoiding bycatch; the effectiveness of mitigation measures in use; recent developments with mitigation and their applicability to CCAMLR fisheries; an annual risk assessment for all fisheries to identify the risk of capture of seabirds in fishing operations; and a regular review of existing conservation measures in light of observer data about their effectiveness and recent research results. Updated advice from IMAF is taken to the CCAMLR Commission via the Scientific Committee on an annual basis, ensuring best-practice seabird bycatch mitigation measures and advice can be rapidly adopted. At this stage no other RFMO and few domestic fisheries have a similar process in place. Adoption of this model by all fishery managers would be a significant step toward substantially reducing incidental mortality of albatrosses and giant petrels in both coastal and high seas fisheries.

2.1.1 Issues relating to longline fisheries

- Current longline fishing techniques continue to represent a significant, and probably the primary, threat at-sea to albatross and giant petrel populations across the globe.
- Despite widespread acknowledgement, including amongst fishery managers, and evidence that bycatch is the most serious threat facing many albatrosses and giant petrels, voluntary uptake of effective seabird bycatch mitigation measures remains limited and mandating use of the most effective mitigation measures remains the rare exception in coastal and high seas fisheries.
- The impacts of longline fishing on all albatrosses and giant petrels within the Australian EEZ have been significantly reduced through implementation of the Longline Fishing TAP. However, fishing gear and practices are constantly changing and continued annual monitoring of bycatch levels in Australian longline fisheries and further mitigation innovation are necessary to ensure the downward trend in bycatch of seabirds is maintained and bycatch is avoided or minimised.
- A precautionary approach to longline fishing in Australia’s Antarctic fisheries has been successful, largely because seabird bycatch mitigation measures and limits, area closures and 100% observer coverage have been implemented from the start of the fishery as standard operating conditions.
- Australian breeding populations also forage outside of the EEZ where they are vulnerable to longline fishing fleets operating with weak or non-existent bycatch mitigation measures. It is critical that this issue continues to be addressed through international conservation and fishing fora such as ACAP, CMS, and RFMOs, and bilaterally with other States whose flag vessels fish on the high seas.
- For all fisheries, annual review of information on fishery performance, the seabird species that interact with a fishery, and improvements in bycatch mitigation practice, would provide RFMOs with the information necessary to ensure the adoption of mitigation measures that are close to international best-practice at any time. Establishment of scientific observer programs to monitor seabird bycatch

and seabird bycatch working groups, modelled on CCAMLR's IMAF Working Group, to collect and review relevant information would facilitate this process.

2.2 Incidental Catch During Trawl Fishing Operations

While the bycatch of seabirds associated with longline fishing has received considerable attention, until recently less emphasis has been placed on the problems associated with the large numbers of birds that routinely follow trawl vessels. It is now recognised that large numbers of albatross are killed in trawl fisheries worldwide (Sullivan and Reid 2002; Sullivan 2004; Gonzalez-Zevallos and Yorio 2006; Sullivan *et al.* 2006; Baker *et al.* 2007a). Sullivan *et al.* (2006b) reported high levels of mortality of albatrosses, predominantly black-browed albatrosses, in the Falklands Island (Islas Malvinas) finfish fleet in 2002/2003. Observers estimated that >1 500 seabirds, predominantly black-browed albatross were killed by finfish trawlers over a 157-day period. Baker *et al.* (2007a) estimated that over 8 500 shy and white-capped albatrosses may be killed annually by trawl and longline fishery operations, with most birds being killed in South African, Namibian and New Zealand waters. Trawl fisheries were responsible for 75% of these deaths.

Traditionally, high levels of seabird mortality caused by trawlers had been associated with netsonde cable collisions (e.g. Bartle 1991; Weimerskirch *et al.* 2000), which are now prohibited in many Southern Hemisphere fisheries (e.g. Weimerskirch *et al.* 2000; CCAMLR 2006 (although CCAMLR prohibited these cables from the 1994/95 season)). The banning of these cables was considered to have largely resolved the trawling bycatch problem. However, more recently significant levels of trawler mortality are reported to have been caused by net entanglements (SC-CAMLR 2001, 2002) and warp cable strikes (Sullivan and Reid 2002, 2003). Most net related mortality recorded in recent years has been caused by pelagic trawlers. Pelagic nets remain at or near the sea surface for extended periods, in contrast to demersal nets which are weighted to sink quickly. Mortality is predominantly caused by birds diving into the net and becoming entangled, particularly in the intermediate size meshes (Weimerskirch *et al.* 2000; SC-CCAMLR 2001, 2002). P. Hicken and I. Everson, in Hooper *et al.* (2003), provide excellent descriptions on trawling operations and the ways that seabirds become entangled.

Collisions with trawl warps in particular, but also with other components of trawling equipment, including trawl doors, backstrops, bridles, sweeps and paravanes, can cause injury or death if the collision is sufficiently severe (Wienecke and Robertson 2002; Gonzalez-Zevallos and Yorio 2006). Sullivan *et al.* (2006b) reported birds being killed after being dragged underwater by the warp cable while feeding on factory discharge at the stern of the vessel. An unknown proportion of these birds slid down the cable and become impaled on a splice in the cable. Sometimes birds are killed when they become stuck to lubricated cables and dragged through trawl winches (Wienecke and Robertson 2002). The problem of interactions with trawl gear is exacerbated when large numbers of birds are present around vessels and the competition for offal becomes intense.

Within Australia, historic trawl bycatch does not appear to be high although there is little scientific monitoring of trawling operations that has specifically focused on seabird bycatch. Since 1994/95, a single trawl vessel has targeted Patagonian toothfish around Macquarie Island (Wienecke and Robertson 2002). A single vessel also trawls for Patagonian toothfish (demersal trawl) and mackerel icefish (pelagic trawl) around the Heard and McDonald Island group (Williams and Capdeville 1996; Wienecke and Robertson 2002). The distribution of some albatrosses and giant petrels from the small breeding populations on these islands overlaps extensively with trawling operations (Lawton *et al.* 2007; Terauds *et al.* 2006b; Trebilco *et al.* 2006), and at times birds attend these fishing vessels in large numbers (AFMA unpublished Observer Reports). Vessels employ simple, effective mitigation measures and as a result few serious interactions between trawl equipment and seabirds have been reported, despite high observer coverage. At Macquarie Island, no deaths of albatrosses and giant petrels have been recorded, as was the case for the demersal trawl fishery at Heard and McDonald Islands for many years. However, in 2004/05 seven black-browed albatrosses and five white-chinned petrels were killed off Heard Island in mid-water trawl operations (Lawton *et al.* 2007) over a few days, demonstrating the potential risks imposed by the fishery. Trawlers have since been required to cease the use of mid-water trawl gear between 1 February and 31 March each year when albatrosses are provisioning chicks, and other mitigation measures are required, including net cleaning prior to shooting to remove items that might attract birds, minimisation of deck lighting, and adoption of shooting and hauling procedures that minimise the time the net is lying on the surface of the water with the meshes slack. At other times of the year midwater trawling can only occur at night. No seabirds have been killed in this fishery since the adoption of the closed season and other mitigation measures for mid-water trawling.

There is considerable trawl fishing elsewhere within Australian waters, with vessels targeting a range of deep water crustacean and finfish species. Much of this fishing effort occurs within areas prospected by albatrosses and petrels, including areas near important breeding colonies around Tasmania. Hundreds of seabirds routinely attend fishing vessels during these and other trawling operations (Gales and Brothers 1996; Sagar *et al.* 2000; AFMA unpublished Observer Reports). Incidental mortality may be occurring in these fisheries but incidents are unlikely to be detected unless observers are specifically tasked with quantifying seabird interactions (Baker *et al.* 2002; Sullivan *et al.* 2006b). The need for such observer programs is demonstrated by recent experience in a pelagic trawl fishery conducted off south east Australia that has historically been thought to have little seabird bycatch. Initial investigations in 2009 and 2010, which included deploying specialist observers, have shown a potentially significant level of albatross bycatch, much more than previously suspected; as a result, all vessels are required to develop an appropriate vessel management plan to manage risk factors, such as offal discharge practices, and employ appropriate mitigation measures (I.Hay pers. comm.).

Collection of data on seabird incidental capture in trawl fisheries is typically limited because of scarce observer coverage and because priority is given to other duties required of observers, potentially leading to seabird deaths going unnoticed. Such a situation existed in longline fisheries in the 1980s when seabird bycatch undoubtedly occurred but was rarely reported. Another difficulty in quantifying bycatch is that some birds initially caught in the net may be lost underwater (Ministry of Fisheries and Department of Conservation 2000), and birds that hit warps during various stages of trawling operations are often undetected because they fall into the water in an area where they may not be captured in the net, thus avoiding detection (Sullivan *et al.*

2006). Reliable data on the levels of seabird bycatch in Australian trawl fisheries will require observer programs to be established to specifically focus on this issue.

Studies to determine the effectiveness of seabird mitigation measures in trawl fisheries are scarce, and accordingly few mitigation devices have been developed and tested. A review by Løkkeborg (in prep.) identified only three devices, which have been described and tested by Sullivan *et al.* (2006a). Sullivan *et al.* (2004) trialled the Falkland Island (Islas Malvinas) warp scarer, Brady baffler, tori (streamer) lines and compared them with a control (fishing with no mitigation technique). All reduced mortality, particularly the warp scarer and tori lines, which also reduced heavy contacts. Bull (2007) also reviewed trawler mortality mitigation techniques and recommended a combination of offal and discard management, the banning of net monitoring cables, paired streamer lines, and a reduction in the time the net is on or near the surface as likely to be the most effective in reducing seabird interactions with the warp cables and net.

The few studies conducted in finfish trawl fisheries to date indicate interactions between seabirds and trawl gear are rare at times of no offal discharge. These studies therefore suggest that a no-discharge policy, or no discharge while gear is in the water, would virtually eliminate seabird mortality. Limiting factory discharge to 'dirty water', resulting from processing, that does not attract large numbers of seabirds (Sullivan *et al.* 2006; Wienecke and Robertson 2001) would also be effective. However, the development and testing of appropriate bird-scaring devices to protect warps may also be useful in mitigating the problem. The use of a suite of measures, including net binding to secure the meshes at the time of setting, removal of fish 'stickers' from nets prior to shooting gear, considering adding weight to the cod end to assist gear in sinking rapidly and retaining offal during shooting and hauling of trawl gear, with full offal retention where feasible, has recently been adopted by CCAMLR as best-practice mitigation for new pelagic finfish fisheries (SC-CAMLR 2007).

The use of netsonde monitor cables or equivalent gear has been prohibited within New Zealand and the CCAMLR Convention Area for many years (Bartle 1991; Murray *et al.* 1993). They are also banned within Australian subantarctic fisheries, but are still permitted elsewhere within the EEZ. However, very few (if any) domestic trawling vessels still use a netsonde cable, preferring the use of hull-mounted transducers or towed aquaplanes on which transducers are set (AFMA logbook databases).

2.2.1 Issues Relating To Trawl Fisheries

- The incidence of mortality caused by the many large trawling fleets around the world that discharge factory waste and attract large bodied seabirds (e.g. albatrosses and giant petrels) requires immediate investigation.
- There is a need to develop best practices for observer data collection to facilitate research and analysis to reduce bycatch of protected species by trawl gear (Deitrich *et al.* 2007) and for investigation into more effective measures to reduce interactions between trawl fisheries and albatrosses and giant petrels.

- Complete observer coverage currently exists on trawl vessels operating within Australia’s Antarctic fisheries (Heard Island and McDonald Islands Fishery, Macquarie Island Fishery). The observer programs responsible for the collection of these data should be maintained in these sensitive areas.
- Management authorities of all other trawl fisheries operating south of 25˚S, particularly where offal is discharged during trawling, need to collect, analyse and publish observer data on seabird interactions, including on the levels and causes of seabird bycatch, focusing especially on recording of warp interactions and trawl entanglements.
- Few, if any, vessels operating within the EEZ use netsonde monitor cables. However, fishers are currently still legally able to use this equipment if they wish to, despite the known adverse impact on seabirds. Ideally, the use of such equipment where additional cables are required would be prohibited in Australia.
- The use of netsonde monitor cables or equivalent equipment in global fisheries can cause substantial mortality to albatrosses and giant petrels and should be discouraged or prohibited through international conservation and fishing forums.
- There is probably potential for considerable reductions in seabird mortality rates in all trawl fisheries by employing appropriate and effective mitigation measures. Retention of offal during shooting and hauling trawl gear should be considered minimum best-practice, with full offal retention, where feasible, as the preferred approach.
- Notwithstanding the present low levels of seabird bycatch, the use of bycatch limits or maximum bycatch rates, an established practice in Australia’s Antarctic and other longline fisheries, should be adopted for trawl (or other methods used in) fisheries operating around Macquarie, Heard Island and McDonald Islands to limit the potential impact of bycatch on the small populations of albatrosses and other seabirds that breed in these areas.
- For all fisheries, annual review of information on that fishery's seabird bycatch performance, the seabird species that interact with it and improvements in bycatch mitigation practice, would provide fishery managers, including RFMOs, with the information necessary to assess the need for, and adopt and refine, effective mitigation measures that are close to best-practice at any time. Establishment of seabird bycatch observer programs and working groups, modelled on CCAMLR’s IMAF Working Group, to collect and review relevant information would greatly facilitate this process.

2.3 Incidental Catch During Driftnetting Operations

The focus on seabird bycatch in driftnets has largely been with regard to high seas drift gillnet fisheries (Northridge 1991). Large-scale driftnet fisheries operated until the end of 1992 when the UN General Assembly enforced a global moratorium on pelagic driftnetting due to the excessive levels of bycatch. Despite this, it is possible that a significant level of illegal driftnetting persists on the high seas and in some EEZs and coastal regions (Alexander *et al.* 1997). The detection of illegal demersal gillnets in CCAMLR waters where albatrosses and giant petrels forage extensively is of concern (SC-CAMLR 2007). Smaller driftnets are still in wide use (Majluf *et al.* 2002). At Punta San Juan, Peru, smaller driftnets were in use between 1992 and 1994 (Majluf *et al.* 2002). An observer recorded that most bycatch between 1992 and 1998 occurred 1992–1994 (c. 80%) when the fishery used surface drift gillnets, and capture rates declined dramatically when they switched to fixed demersal gillnets during the period 1995–1998. Albatrosses, probably Salvin's or Chatham albatross, were captured in small numbers.

In Australia, the incidence of albatross and giant petrel bycatch in coastal gillnets is unknown but large numbers of other seabirds such as shearwaters and penguins have been caught and drowned. The risk to seabirds is increased if nets are left unattended and/or set overnight. Commercial gillnetting in some situations may be less likely to impact on seabirds as, in most cases, nets will be attended or pulled regularly to ensure that fish quality is maintained and wastage avoided. However, Lyle (2000) reported that in Tasmania at least a quarter of all gillnet sets had soak times of 24 hours or greater and, in addition, 75% of all recreational gillnets were set overnight. Recreational use of gillnets during the night is now prohibited in Tasmania.

Recreational gillnetting in coastal waters is prohibited in all States in Australia, except Western Australia and Tasmania. Western Australia has attendance regulations for recreational netting whereas Tasmania continues to allow unattended (daytime) recreational gillnetting in coastal waters. Brothers *et al.* (1996) recommended that gill netting in Tasmania be prohibited in close proximity to islands with breeding colonies of seabirds.

The impact of coastal gillnetting on albatrosses and giant petrels in Australia is unlikely to be significant as commercial gillnets used to target sharks are set at depths beyond the diving capabilities of most albatrosses and recreational nets are usually set adjacent to the shoreline where these birds do not usually occur.

2.3.1 Issues Relating To Driftnetting

- The significance of albatross and giant petrel bycatch by global smaller driftnet fisheries is unknown, but of potential concern. While a low priority, the incidence of driftnetting and associated bycatch in areas frequented by albatrosses and giant petrels should be assessed.
- Within Australia, the impact of driftnets on albatrosses and giant petrels is likely to be very low.
- Few seabird bycatch reduction methods have been developed for gillnet fisheries, although increasing the visibility of the net reduces seabird bycatch (Bull 2007). Further studies are required to determine the efficacy of this technique and its influence on target species catch rates, however this is a low priority.

2.4 Incidental Catch During Trolling Operations

The commercial and recreational practice of trolling a fishing line at or near the surface (for pelagic species such as albacore tuna *Thunnus alalunga*) has the potential to cause albatross and giant petrel mortality if birds are caught on hooks.

This practice is unlikely to cause significant levels of albatross or giant petrel injury or mortality. However, it could be eliminated if troll lines were set at least 2m below the surface of the water.

2.4.1 Issues relating to trolling operations

- The incidence of this source of mortality is unknown but unlikely to be significant.
- This potential problem could be rectified by using educational and other strategies to encourage commercial and recreational trollers to set their fishing lines at least 2m below the surface of the water.

2.5 Intentional Shooting/Killing

Despite it being an offence under Commonwealth and most Australian States' legislation, albatrosses are sometimes intentionally shot for sport by recreational fishers or to reduce scavenging from commercial fishing vessels both inside and outside of Australian waters (Adams 1992; Brothers *et al.* 1998; DPIPWE unpubl. data; Awkerman *et al.* 2006). Knowledge of the rate of this form of mortality is limited.

A significant number of adult waved albatrosses is being taken incidentally, as well as a lesser number intentionally (for human consumption), in the Peruvian artisanal longline and gillnet fisheries, and this is reflected in the lower annual adult survival observed in 1999–2005 in comparison to that of the 1960s (Awkerman *et al.* 2006). The extent of this intentional take is unknown, but an action plan has been developed by ACAP, in consultation with the governments of Peru and Ecuador (ACAP 2007d) to address this and other threats.

Both wandering albatrosses and shy albatrosses are known to have been illegally shot by personnel involved with the Tasmanian dropline fishery in an attempt to reduce bait loss (DPIPWE unpubl. data). Wandering albatrosses have also been reported to be intentionally shot off the New South Wales coast (Blakers *et al.* 1984; Tomkins 1985) with a report of the shooting of an albatross (species unknown) in this region in 2010 (I. Hay pers. comm.). It seems likely that other species of albatross and giant petrels have been shot for the same reasons from time to time; however the prevalence of such practices and the extent to which they persist is unknown.

Wandering, black-browed, shy and grey-headed albatrosses and many ‘foraging species’ are also illegally killed in South African waters by fishers for sport, food and for use as bait (Adams 1992). Wandering albatrosses and black-browed albatrosses are also deliberately shot off Uruguay (Stagi *et al.* 1996).

2.5.1 Issues relating to intentional shooting/killing

- The incidence of this source of mortality is unknown but probably not significant.

- Regulation of the carriage of firearms on fishing vessels would probably be the most effective way to eliminate this source of mortality in Australia.
- Mechanisms for educating Australia's professional and amateur fishers regarding the threatened status of albatrosses and giant petrels, the problems with intentionally shooting them and the penalties for doing so, should be developed and implemented.
- Intentional shooting or killing of albatrosses and giant petrels outside of the Australian EEZ needs to be addressed by coastal States and international conservation and fishing forums, including RFMOs and ACAP Parties.

2.6 Impacts of Introduced Pest Species

Introduced mammals are the greatest land-based threat to seabirds on subantarctic islands (Jouventin and Weimerskirch 1991). Alien species are reducing seabird populations via nest predation, nest destruction and habitat modification. Albatrosses and giant petrels are especially vulnerable to alien mammals for several reasons, specifically, their lack of effective anti-predator behaviour; their habit of building their nests on the ground and leaving chicks unattended during long-range foraging bouts; and their low annual productivity.

Three mammal species have posed the most significant conservation problems at Australian seabird breeding sites in recent years: – cats (*Felis catus*), rats (*Rattus* spp.) and rabbits (*Oryctolagus cuniculus*). Cats and rats directly impact seabirds through predation of eggs, chicks and adults, and rabbits damage vegetation leading to erosion, increased exposure to natural predators and loss of breeding habitat (Baker *et al.* 2002). Both cats and rabbits have been listed as Key Threatening Processes under the EPBC Act and Threat Abatement Plans have been prepared to manage their impact (Environment Australia 1999a; 1999b).

Some of the species of albatross that forage in Australia also suffer predation by feral dogs (*Canis familaris*), coatimundis (*Nasua nasua*), ferrets (*Mustela furo*), pigs (*Sus scrofa*) and stoats (*Mustela erminea*) (Moors and Atkinson 1984). In addition, rabbits, cattle (*Bos taurus*), sheep (*Ovis aries*), goats (*Capra hircus*), reindeer (*Rangifer tarandus*) and pigs introduced to breeding islands may limit the colony size of several 'foraging species' by inadvertently trampling on nests, overgrazing vegetation and modifying habitat required for nesting (Croxall *et al.* 1984b; Robertson and Bell 1984).

Many islands that are important breeding sites for seabirds are currently free of predators, and hence feral predators would not be considered an immediate threat at these sites. However, the risk of alien introductions is always present, particularly where islands are visited regularly by humans. Small populations of seabirds, in particular, could be immediately threatened if the predator-free status of important breeding sites was lost. Appropriate quarantine and other regulations should be put in place at these breeding sites to minimise this threat.

Within Australia, Macquarie Island is the only albatross or giant petrel breeding site where feral pests — rabbits, rats and mice — currently pose a threat.

Feral cats were present on Macquarie Island, preying upon the eggs or small, unattended chicks of all four albatross and both giant petrel species (Rounsevell and Brothers 1984), but were declared to be successfully eradicated in 2002. Quarantine and monitoring of Macquarie Island continues and should ensure the island remains cat free (Parks and Wildlife Service 2006).

Wekas (*Gallirallus australis*), a flightless rail, were introduced to Macquarie Island in the mid-1800s as a source of food for sealers (Cumpston 1968). They became numerous and widespread in the coastal regions and north half of the island (Marchent and Higgins 1993). An eradication program was commenced in 1985, which resulted in the last recorded weka being destroyed in 1988 (Parks and Wildlife Service 2006).

Rats, rabbits and house mice still remain on Macquarie Island. The introduction of rabbits to Macquarie Island in the 1870s has significantly modified the distribution of vegetation alliances (Rounsevell and Brothers 1984), and has significantly degraded and destabilised nesting habitat. Rabbit population control began in 1978, reducing numbers from in excess of 150 000 to an estimated 3 300 animals (Parks and Wildlife Service 2006) and this resulted in rapid recovery of most plant communities (Copson & Whinam 1998, 2001). However, from a low in the 1980s the rabbit population has increased to over 100 000 rabbits, which may be attributable to the combined effect of the eradication of cats, warmer drier weather and the possible reduction in effectiveness of biological control methods (Parks & Wildlife Service 2006). Eradication measures are currently the only alternative for effective control of rabbits on Macquarie Island.

Ship's rats (*Rattus rattus*) inhabit the tussock grasslands used by most albatross and giant petrel species on Macquarie Island, and may opportunistically prey upon eggs and unattended chicks (Copson unpubl. data). An aerial poisoning program was conducted on Campbell Island in 2000 to remove Norwegian rats (*R. norvegicus*) and after thorough on-ground searches in 2003 the program was declared successful (www.doc.govt.nz/templates/page.aspx?id=33380). Cold temperate Campbell Island (11 000ha) is very similar to subantarctic Macquarie Island (12 800ha).

Unlike rabbits there are no viral control agents for rodents. Attempts to locally control rodent numbers were conducted from 1999 to 2003 using brodifacoum rodenticide, however continual control of rats using rodenticides requires a high level of resources and is not seen as effective in the long-term due to reinvasion of the treated areas. This program ceased in 2003 due to concerns that bait shyness or anti-coagulant tolerance from partial poisoning may compromise the success of an eradication operation.

House mice occur in all habitats and vegetation communities on Macquarie Island but have a preference for tussock grassland (Parks and Wildlife Service 2006). Like rats, mice are also documented as eating the eggs of smaller bird species. Evidence on subantarctic Gough Island has identified mice as being responsible for increased mortality of several species of seabird fledglings, including the Tristan albatross, which is of a similar size to the wandering albatross (Cuthbert and Hilton 2004).

The naturalisation of rabbits, rats and mice may have secondary effects on Macquarie Island's ecosystem resulting from the ecological relationships between predators and their prey. These naturalised pest species act as additional food sources for introduced predators, as well as for natural predators such as subantarctic skuas *Catharacta lonnbergi*. Skuas are opportunistic predators of seabird chicks and eggs. It has been suggested that the introduction of rabbits to Macquarie Island has allowed the subantarctic skua population to increase artificially, thereby increasing nest predation pressure on other seabirds. An increase in the subantarctic skua population could prevent albatrosses and giant petrels from moving into traditional breeding areas other than along the protected coastal rocks (Rounsevell and Brothers 1984). A subsequent reduction in rabbit numbers through eradication programs might force the elevated numbers of skuas to further intensify predation pressure upon ground-nesting seabirds (Scott 1996).

The cumulative effect of these introductions is a major modification of the local vegetation and a great reduction in the populations and distribution of several species of ground-nesting seabirds. Today, some seabird species are only found breeding on islets and sea stacks adjacent to the main island that are uninhabited by feral species. It may be significant that only 40 pairs of black-browed albatrosses breed annually on Macquarie Island whereas about 140 pairs breed on nearby Bishop and Clerk Islets, which, though tiny in comparison to Macquarie Island itself, are free of the introduced pests that adversely affect seabirds on Macquarie.

An eradication program for rabbits, rats and mice on Macquarie Island has been developed (Parks and Wildlife Service 2007b). It will cost approximately $25 million over seven years, and has been jointly funded by the Tasmanian and Australian Governments. The field phase of the eradication of rabbits and rodents commenced in 2010 and involved helicopters dropping pellet baits (brodifacoum) targeting rabbits, rats and mice right across the island. Field teams will be used to follow up on the ground to eliminate individual rabbits that have survived the baiting. These teams will use a range of techniques including shooting, fumigating burrows, trained dogs and trapping over a four-year period to ensure that all rabbits are removed.

The cost of the program reflects a number of factors including the isolation of the island, costs of shipping and helicopters, contingency funds to allow for unforeseen delays, and the need for a substantial number of personnel and up to 14 highly trained dogs for the follow-up part of the project.

While it was expected that there would be some loss of individuals of non-target species in undertaking the eradication program through the effects of primary and secondary poisoning (Parks and Wildlife Service 2007b), despite efforts to avoid or minimise such losses, the losses in 2010 were much higher than expected, even though bad weather prevented flying of helicopters and meant only 8% of the island was baited in June. Two months or so of persistent bad weather caused the 2010 program to be suspended. Prior to the baiting commencing, the most susceptible native species were predicted to include both species of giant petrels, the subantarctic skua, and the kelp gull (*Larus dominicanus*), and possibly black duck.

By 9 February 2011, 947 dead birds had been found, including 298 northern giant petrels (NGPs) (approximately 8% of the breeding population), 16 southern giant petrels (0.3%); and 226 subantarctic skuas (11%). The actual number of bird mortalities is likely to have been higher as not all areas of Macquarie Island were able to be intensively searched and some individuals may have died at sea. Additionally, 4 SGPs (1 bird banded at Macquarie Island) were found dead in the New Zealand subantarctic and tested positive for brodifacoum. Further, while only 10 NGP carcases were examined, 90% were males, probably because of their more coastal-oriented foraging (females tend to be more pelagic), exacerbating the impact on the breeding population. The primary cause of bird deaths was brodifacoum poisoning resulting from the presence of carcasses of poisoned bird and target species, the accessibility of these carcasses and the scavenging behaviour of the bird species. While the deaths of giant petrels and skuas were mainly due to secondary poisoning, the deaths of kelp gulls and black gulls may have been caused by both primary and secondary poisoning.

A review of the 2010 operations was conducted (DPIPWE 2010). To reduce the impact of the eradication program on non-target species, several changes are expected when baiting resumes in 2011, including increased efforts to systematically search for and remove poisoned target and bird carcasses. The number of bird deaths that will occur after a future baiting operation if these additional measures are implemented cannot be quantified, however the systematic removal of poisoned carcasses is expected to significantly reduce the incidence of deaths arising from secondary poisoning. Whatever mitigation measures are put in place, it is likely that bird deaths will follow any future baiting. However, no bird population is expected to be lost through baiting whereas if the pest eradication program is not undertaken, catastrophic damage to the ecosystems of Macquarie Island will continue and some seabird breeding populations on the island will probably become extinct. Populations of bird species which were poisoned in the 2010 baiting operation are likely to recover even though their populations will again be reduced by any future baiting.

An integral part of the Macquarie Island feral pest eradication program is studying the responses of native and feral species to changes in status of those species being targeted or since removed, i.e. cats and rabbits. This includes long-term programs to monitor vegetation changes, the abundance and distribution of nesting seabirds and rabbit and rodent numbers (Parks and Wildlife Service 2007b).

2.6.1 Issues relating to feral pest management

- Feral pests pose a very significant land-based threat to albatrosses and giant petrels breeding on Macquarie Island.
- A *Macquarie Island Vertebrate Pests Management Plan: 2000 – 2005* has been completed. A multi-year program for the eradication of rabbits, rats and mice commenced in 2010 with aerial dropping of pellet baits targeting rabbits, rats and mice across part of the island; secondary poisoning of non-target species, including northern giant petrels, was higher than expected and additional mitigation measures are planned in future baiting operations to minimise such problems.
- Comprehensive follow up on the ground to eliminate individual rabbits that have survived the bait drop is vital to prevent future re-establishment of rabbit populations.
- Monitoring the response of the vegetation and seabirds is essential and will provide indicators of the success of the eradication program.

- Full documentation of the eradication operation and related monitoring programs should occur so that lessons learnt from such a large-scale eradication program can be applied elsewhere.

Stringent requirements for the prevention of introductions of exotic species to Macquarie Island, the Territory of Heard and McDonald Islands, and other Australian breeding islands should be continued where they currently are in place, and implemented for areas where quarantine plans are lacking.

2.7 Human Disturbance at the Nest

Although albatrosses may appear to be undisturbed by the presence of humans, biotelemetric studies demonstrate that nesting seabirds become stressed (as indicated by a marked increase in heart rate and stress hormones, such as corticosterone) as soon as humans are visible (Holberton and Wingfield 1994). The presence of humans too close to the nest can cause breeding failure as the stressed adults abandon or inadvertently crush eggs or small chicks.

Southern giant petrels can be nervous around humans and easily disturbed during nesting, often resulting in breeding failure. The site may be abandoned if a colony is persistently visited (Williams 1984; Bretaganolle 1989).

Many of Australia's breeding colonies of albatrosses and giant petrels are monitored regularly to detect changes in population status and trend. All research on albatrosses and giant petrels requires provision of a scientific permit and approval from an Animal Ethics Committee.

Of all albatross and giant petrel breeding localities under Australian jurisdiction, Macquarie Island hosts the greatest number of human visitors. Nesting colonies on Albatross Island, Pedra Branca, the Mewstone, Heard and McDonald Islands, and within the Australian Antarctic Territory receive considerably fewer visitors each year.

(i) Australian Antarctic Territory

It was speculated that activities associated with the presence of the three Australian research stations located within the AAT may have caused local population declines of southern giant petrels (Woehler *et al.* 1990; Woehler 1993), although this view has been disputed (Wienecke *et al.* 2009) as the available population data are infrequent, use different count units and are ambiguous, and other factors may be contributing to population changes. All known southern giant petrel breeding sites in the AAT are within Antarctic Specially Protected Areas and access during sensitive breeding times is restricted to permit holders only. The establishment of research bases at other Antarctic sites is also blamed for drastic or complete depletion of nearby breeding populations (e.g. Jouventin *et al.* 1984; Rootes 1988). However, the southern giant petrel population at Heard Island has also declined markedly since the 1950s (Kirkwood *et al.* 1995), yet there has been virtually no human presence on the island; additionally, mostly in the light of population increases in the South Atlantic region, including at some sites where regular human visitation occurs, southern giant petrels have been downlisted by the IUCN from near threatened to of least concern (IUCN

Red List 2010, version 2010.4). It seems unlikely then that direct human intervention at the breeding colonies is the sole cause of southern giant petrel population declines. Nonetheless, such localised disturbances may have exacerbated local and global population decline and a precautionary management approach is warranted. Further population monitoring and related research, using a consistent approach to maximise comparability of data, is needed to better assess population status and trends and determine whether there are differences or similarities between higher Antarctic latitude colonies.

(ii) Heard Island and the McDonald Islands

All access to the Territory of Heard Island and McDonald Islands is restricted to permit holders only. The number of tourists visiting Heard Island is very low and likely to remain at current levels for the foreseeable future (Australian Antarctic Division 2005). All tourist and scientific activities are required to minimise their environmental impacts, including any effects on colonies of nesting seabirds. The operation of vehicles and aircraft is also restricted. The management plan for the Territory specifies minimum separation distances for visitors on foot of 100m for Wandering albatrosses and southern giant petrels, *inter alia*, and 50m for other albatrosses (Australian Antarctic Division 2005). Approach closer than these distances is allowed in only limited circumstances and requires a special permit.

The *Heard Island and McDonald Islands Marine Reserve Management Plan 2005* (Australian Antarctic Division 2005) divides the Reserve into seven zones in order to facilitate protection of those areas that are susceptible to the impacts of human activities, such as vegetated areas, EPBC Act listed threatened species and sensitive geological features, and to confine human activity to sites that can sustain it. The level and type of protection varies between zones, with activities in the Reserve being restricted to the most appropriate areas and managed accordingly. The zones comprise a Main Use Zone, a Visitor Access Zone, a Heritage Zone, a Wilderness Zone, a Restricted Zone, an Inner Marine Zone and an Outer Marine Zone.

The Main Use Zone areas provide suitable locations in which access and support operations can be conducted while confining the potential environmental impacts associated with these activities to areas which have been the focus of most past and current activities. The Visitor Access Zone areas provide for appropriate management of low impact, short–term, land–based visitor activities. The Heritage Zone restricts activities in areas encompassing culturally significant remains of early and pre– Australian National Antarctic Research Expedition (ANARE) buildings.

The Wilderness Zone provides for the management of human activities to maintain the relatively undisturbed and wilderness qualities of the majority of the terrestrial component of the Reserve. Activities that would result in long–term impacts to the natural qualities of the Reserve are not permitted in this zone, with access primarily being allowed for scientific research, environmental monitoring and management activities.

The Restricted Zone comprises those areas with environmental values that are highly sensitive to the potential impacts of human activities for which it is particularly desirable to conserve existing minimal levels of human disturbance, or where other concerns such as visitor safety are paramount.

The McDonald Island group and other small offshore rocks and shoals have been infrequently, if ever, visited and warrant the highest level of protection to maintain their undisturbed state. Access to, and activities in, these areas are restricted to essential management purposes and tightly controlled. McDonald Island, which roughly doubled in size due to volcanic activity between the 1980s and 2004, has been visited briefly on only two occasions, in 1971 and 1980.

The Inner Marine Zone provides for the management of activities in the nearshore marine areas of the Reserve to ensure protection of the coastal environment of the islands, the nearshore foraging areas of wildlife, and the values of that marine area. The Outer Marine Zone provides for the management of activities in the marine areas of the Reserve that extend beyond the Inner Marine Zone.

(iii) Macquarie Island

The Macquarie Island Nature Reserve is a restricted area under Section 25 of the Tasmanian *National Parks and Wildlife Act 1970*. As such, permits are required to enter the reserve. The *Guidelines for Tourist Operations and Visits to Macquarie Island Nature Reserve and World Heritage Area* set stringent guidelines for tourist operations on the Island, and provide limited access for educational tourism purposes to promote appreciation and awareness of the values of the reserve.

Tourist numbers are limited at Macquarie Island (Parks and Wildlife Service 2006) and tourists are not permitted in areas where albatrosses and giant petrels breed. Access to the Island is allowed on short sections of the beach along tracks at the Isthmus and on boardwalks leading to viewing platforms. The operation of vehicles and aircraft is also restricted.

The ANARE research station and associated infrastructure has been largely confined to the Isthmus at the northern extremity of Macquarie Island.

Special Management Areas (SMAs) have been designated to further protect natural or historical values. SMAs include very sensitive areas where there are high densities of breeding wildlife and sporadic breeding colonies that are difficult to enter without disturbing threatened wildlife or causing damage to fragile vegetation. Special authorisation is required to visit SMAs for specific research, monitoring or management purposes. Access to some areas (e.g. albatross breeding areas) will only be granted under very exceptional circumstances and other areas are not able to be accessed for any reason due to the proximity of extremely sensitive breeding wildlife.

The *Macquarie Island Nature Reserve and World Heritage Area Management Plan 2006* describes SMAs. The SMAs are reviewed annually and changes to the extent and season of access are imposed as required. Approach distances to albatrosses and giant petrels are also reviewed annually.

Albatrosses and giant petrels on Macquarie Island are monitored as part of a long-term conservation and research project. Chicks are banded just prior to fledging to minimise the risk of nest disturbance. Scientific permit provisions are reviewed annually.

Current research into human disturbance issues

- DPIPWE monitor the effects of visitor and researcher activity on the breeding success of the four species of albatrosses nesting on Macquarie Island (wandering, grey-headed, black-browed, light-mantled) as well as northern and southern giant petrels.
- DPIPWE use these data to provide recommendations regarding the effective management of visitors to albatross and giant petrel breeding colonies. This project began in 1994/95 and is currently funded to 2010. Funding for this work is provided by DPIPWE, the Antarctic Science Advisory Committee (ASAC) and the AAD.

(iv) Albatross Island, Pedra Branca and the Mewstone

Pedra Branca and the Mewstone are designated part of the Tasmanian World Heritage Area, and Albatross Island is a Nature Reserve. Access is currently unrestricted to these three breeding sites, although a draft Management Plan for Albatross Island recommends that permits be required for access to that site.

2.7.1 Issues related to human disturbance

- Human disturbance near the nest can disrupt or reduce albatrosses and giant petrels nesting attempts or breeding success.
- The *Macquarie Island Nature Reserve and World Heritage Area Management Plan 2006* provides management guidelines for control of human disturbance of native wildlife on Macquarie Island.
- The *Heard Island and McDonald Island Marine Reserve Management Plan 2005* provides management guidelines for control of human disturbance of native wildlife on Heard Island and McDonald Island Marine Reserve.
- Appropriate area closures, activity restrictions and approach distances should be maintained (or implemented as required) to minimise human disturbance to albatrosses and giant petrels at all breeding sites.
- Access to and activities at Macquarie Island, Heard and McDonald Islands, and the AAT are by permit only. Finalisation of similar management provisions for access to Albatross Island, Pedra Branca and the Mewstone should be addressed as a matter of urgency.
- Tourists and visitors to breeding sites should continue to be educated on the vulnerability of albatrosses and giant petrels to human disturbance.

2.8 Avian Parasites and Diseases

The breeding success of albatrosses and giant petrels may be reduced through natural agents such as parasites or disease. Infectious diseases have potential to cause serious declines, but the incidence and prevalence of disease is poorly known.

The outbreak of two diseases in the 1980s (Avian cholera and another pathogenic bacterium, *Erysipelas*) has been identified as a cause of the decline of the yellow-nosed albatross on Amsterdam Island, a key breeding site for this species comprising 55% of the global population (ACAP 2007c). These diseases affect mainly young chicks, with a cyclic pattern between years, but also kill adult birds. The diseases may be currently threatening the rare Amsterdam albatross with extinction, and are probably also affecting sooty albatrosses (Weimerskirsch 2004).

An avian pox virus transmitted by fleas and ticks *Ixodes* spp. is a major cause of shy albatross chick mortality during some years at the Albatross Island breeding colony. Heavily infested nestlings carry ticks clustered around the gape and along the soft, exposed skin on the underside of the bill. Such chicks appear weak and underweight (Johnstone *et al.* 1975), and ultimately die. While the effects of this disease vary inter-annually, infestations can be so severe in some years (breeding success can be reduced as low as 10% in some colonies) that they represent a significant factor restricting the recovery of the Albatross Island population (Johnstone *et al.* 1975; N. Brothers pers. comm., in Gales 1993; Woods 2004).

It is likely that the avian pox virus is able to persist in the environment and on the various arthropod vectors that spread the disease. How a chick responds to disease is highly dependent on its health status (nutritional status, degree of endoparasitic load, parental care) and the level of stress it is exposed to – environmental conditions, ectoparasitism, disease and starvation (Woods 2004). The poor condition of many birds suggests that nutritional factors may be significant determinants of disease expression. Based on the preliminary results of the study carried out by Woods (2004), it would appear that there is sufficient evidence to warrant a structured disease investigation into the cause of death of shy albatross chicks at Albatross Island.

Ticks on adults and chicks at colonies of black-browed albatrosses at the Falkland Islands (Islas Malvinas) are known to spread an avian pox virus, causing localised sporadic mortality. These ticks are also present on Macquarie Island (Selkirk *et al.* 1990; G. Copson pers. comm., in Gales 1993).

2.8.1 Issues relating to parasites and diseases

- Outbreaks of an avian pox virus can be one of the most significant causes of shy albatross chick mortality at Albatross Island in some years. This disease represents a significant factor restricting the recovery of this population.
- Knowledge of the disease and its determinants is currently limited.
- A structured investigation into the diseases of shy albatrosses at Albatross Island is needed, to build on the work of Woods (2004), with the objective of understanding the disease determinants and developing potential recovery actions.
- Unusual mortality events of albatrosses and giant petrels should be investigated under appropriate wildlife health and disease guidelines.

2.9 Loss of Nesting Habitat and Competition for Nest Space

Albatrosses, giant petrels and many other seabirds typically nest on isolated, relatively small islands. As a consequence, competition (both within- and between-species) for limited nest space can be extreme, particularly on smaller islands.

Albatross and giant petrel colonies may become displaced by the change in vegetation cover on islands and coastlines—such as the spread of introduced weeds, or the damage to vegetation by rabbits on Macquarie Island (see Section 2.6). For species that prefer open ground, the regrowth of sedges, shrubs or forest after a fire may cause populations to shift to new breeding sites. The removal of browsing mammals such as goats, sheep, cattle, and rabbits from islands has sometimes initiated new problems for seabirds (e.g. Taylor 2000a).

Storms and cyclones have the potential to have serious effects on the nesting substrate, vegetation and wildlife on remote seabird breeding islands, in addition to impacting seabirds at sea. Such natural factors can place additional pressures on seabird populations adversely affected by anthropogenic influences. On the Sisters and Forty-fours Islands in the Chatham Islands Group (New Zealand), a severe easterly storm in 1985 stripped the islands bare of vegetation and soil cover. The albatrosses that nest on these islands have been unable to construct proper nest sites and subsequently there has been greatly increased egg mortality. To compound the problem most of the breeding population of the normally biennially nesting northern royal albatross now nests annually owing to low breeding success, thus further limiting nest site availability (Robertson 1998; Taylor 2000).

Interspecific competition for nest space has been identified as a potential threat to shy albatrosses nesting on Pedra Branca (DPIPWE unpublished). Australasian gannets (*Morus serrator*) have been increasing in numbers on Pedra Branca where nest sites are limited (Bunce *et al.* 2002). In 1995 there were an estimated 3 317 gannet pairs, an increase of 2% per year since 1939 and 7% per year since 1978. The gannets appear to be more aggressive at nest interactions and tend to displace shy albatrosses from potential nesting sites, particularly in areas where gannets outnumber albatrosses (ACAP 2006; DPIPWE 2007; N. Brothers pers. comm.). Shy albatross productivity in 2006/7 was 31 chicks, was around 30 chicks for last five years before dropping to 21 chicks in April 2010 (DPIPWE 2010) and has shown a steady, steep decline from about 150 in 1998/9 (DPIPWE 2007). The relative abundance of the two species on Pedra Branca may also be changing due to other natural processes or to anthropogenic climate change. Alternatively, it is possible that certain human activities (e.g. discards from commercial fishing) have a greater impact upon the albatrosses, giving the gannets a competitive advantage.

2.9.1 Issues relating to nesting habitat loss and interspecific competition

- To protect albatross nesting habitat, the eradication of rabbits from Macquarie Island is a priority (see Section 2.6).
- Australasian gannets on Pedra Branca may be outcompeting shy albatrosses for nest space, thereby reducing their potential colony size. It is unclear if this is a natural process or one caused by human activities. The relative distributions and abundances of shy albatrosses and Australasian gannets on Pedra Branca should be monitored.

- Most seabirds are adversely affected by stochastic events, such as storms and cyclones, but there is no practical or feasible way to manage this process.

2.10 Competition with Fisheries for Marine Resources

The progressive degradation of the marine habitat, particularly via the potential global over-extraction of marine resources, may have long-term effects on the status of albatrosses and giant petrels and be as serious as the more direct and acute pressures of interactions with fisheries (Croxall 1998).

All of the world's major fishing grounds are being exploited at or beyond sustainable limits, and many have suffered serious declines. According to the FAO, 70% of the world's fish stocks are now fully exploited, overfished, depleted or rebuilding from prior overfishing. Production has fallen dramatically in 13 of the world's 15 major fishing areas (Birdlife International 1995).

Such over-extraction has profound implications for the marine ecosystem, particularly for higher order predators such as albatrosses and giant petrels. Seabird populations have and probably will continue to decline through direct competition with fisheries for prey (Croxall 1998). The ecological sustainability of fisheries is thus particularly crucial to albatrosses and giant petrels. For example, Hedd and Gales (2001) identified that the primary prey species for shy albatross (redbait (*Emmelichthys nitidus*), Gould's squid (*Nototodarus gouldi*) and jack mackerel (*Trachurus declivis*)) are, or have been, commercially harvested.

The dietary requirements of albatross and giant petrel populations need to be taken into account when management arrangements (e.g. total allowable catches or TACs) of fisheries that overlap with the foraging grounds of albatrosses and giant petrels are being developed or revised. It is, however, particularly difficult to accurately determine the level (and the effects) of competition for food resources between seabird populations and fisheries (Hedd and Gales 2001). Any assessment of the effects of competition for food resources requires a thorough knowledge of:

- the dietary requirements of each species, including seasonal, annual and geographical variability;
- the foraging range of each species;
- the range and availability of prey items; and
- the distribution of fishing effort.

Each of these factors is potentially difficult to determine with accuracy. Within Australia, responsibility for ensuring the ecological sustainability of Commonwealth fisheries rests with the Australian Fisheries Management Authority (AFMA). The activities of AFMA are governed and guided by the legislative objectives contained in Section 3 of the *Fisheries Management Act 1991*. One objective in that Act requires 'ensuring that the exploitation of fisheries resources and the carrying on of any related activities are conducted in a manner consistent with the principles of ecologically sustainable development and the exercise of the precautionary principle, in particular the need to have regard to the impact of fishing activities on non-target species and the long term sustainability of the marine environment'.

The EPBC Act requires that strategic environmental assessments be carried out for Commonwealth-managed fisheries. A strategic environmental assessment assesses the relevant impacts of actions taken under a management plan for a fishery. The Minister for Sustainability, Environment, Water, Population and Communities considers the reports and assesses the ability of the management arrangements to manage the fishery in an ecologically sustainable manner. If satisfied with the management arrangements the Minister may accredit the fishery as exempt from matters of national environmental significance, interactions with protected species and export of native species for a period of up to five years. The outcomes of the assessment must be included in the management plan or arrangements for each fishery. All exporting fisheries have received accreditation under the EPBC Act (AFMA 2006). The Southern and Eastern Scalefish and Shark Fishery export accreditation was renewed in February 2010, subject to a range of conditions, including relating to seabirds.

Outside Australian waters, however, there is no similar standard or approach for most high seas fisheries, including those in areas adjacent to the Australian EEZ where Australian breeding seabirds are known to forage. Thus, it is important that regional fisheries organisations with responsibility for managing high seas fisheries continue to be urged to give greater consideration to the needs of seabirds when developing their management arrangements.

2.10.1 Issues relating to competion with fisheries

- Over-harvesting of fish and squid species is a global problem that may be a signficant threat to albatrosses and giant petrels.
- The foraging ranges, prey species and dietary requirements of populations of albatrosses and giant petrels and other dependent species in direct competition with fisheries need to be assessed and taken into account when management arrangements (e.g. TACs) for fisheries are being developed or revised by AFMA under the EPBC Act and other relevant fishery managers.
- There is a need to increase understanding of prey species or albatrosses and giant petrels and how diet varies across regions, years and seasons.
- AFMA is required under the *Fisheries Management Act 1991* and the EPBC Act to manage fisheries in an ecologically sustainable manner, and to have regard for non-target species.
- RFMOs should continue to be urged to give greater consideration to the needs of seabirds when developing high seas fisheries' management arrangements.

2.11 Dependence on Discards from Fishing and Tourist Vessels

(i) Dependence upon discards from fishing vessels

Some seabird species have become dependent upon the offal discarded from fishing vessels during operations and/or processing at sea. They scavenge dead prey and fishery discards and bait (Croxall and Prince 1994), with larger species being more predisposed to boat-following (Baker *et al.* 2002).

There are essentially two issues arising from this dependence upon discards. First, the disposal of offal further encourages albatrosses and giant petrels to follow fishing vessels, significantly increasing their likelihood of becoming injured or killed during fishing operations by direct interactions with fishing gear (see Sections 2.1 and 2.2).

Second, some populations have become habituated to the regular food source and have altered their foraging ranges and dynamics accordingly (Ryan and Moloney 1988; Adams 1992; Acros and Oro 1996; Blaber *et al.* 1998; Weimerskirch 1998; Sagar *et al.* 1999). Votier *et al.* (2004) argue that discards are a key food resource for many seabird species. Evidence indicates that the additional food made available by commercial fishing operations may influence breeding success and hence population sizes in some seabird species (e.g. Blaber *et al.* 1998). However, consideration of these 'benefits' needs to be weighed against the negative side of the balance sheet (Baker *et al.* 2002). For example, in the North Sea, reduced rates of discarding, particularly when coupled with reduced availability of small shoaling pelagic fish, can result in an increase in predation by great skuas on other birds (Votier *et al.* 2004).

The availability of this additional food may not always benefit a species. The consequence of birds becoming habitually attracted to the offal discarded from fishing vessels may be that they return less frequently to the nest during critical phases of the nesting period, causing the nesting attempt to fail (Terauds and Hamill 1999). This indirect threat has been specifically identified as potentially affecting black-browed albatrosses breeding on Macquarie Island (Terauds and Hamill 1999; Terauds *et al.* 2006a). Weimerskirch (1998) reported that, in 1994, when black-browed albatrosses breeding on Iles Crozet concentrated foraging in an area of high natural prey and largely ignored a vessel fishing in an adjacent foraging area, fledging success was the highest on record.

There are few available data to quantify this issue. Greater use of the extensive satellite-tracking data that are now accumulating (BirdLife International. 2004b) is one option to determine the level of association and degree of overlap between fishing vessels and foraging albatrosses and giant petrels, and their dependence on discards.

(ii) Dependence upon discards from tourist boats

Many tourist boat operators that conduct wildlife viewing trips off the coastline of Australia and other parts of the world throw 'chum' (such as frozen squid) to attract seabirds—particularly albatrosses—to the vessel. This technique is used to provide tourists with the opportunity to see flocks of seabirds feeding and competing at a close range.

This practice offers another artificial food source for the birds, and further encourages and habituates them to follow boats, again increasing their likelihood of interacting with fishing vessels. In many cases, 'chumming' is carried out by experienced birdwatchers and is widely considered as acceptable behaviour by the birding fraternity (e.g. Onley and Schofield 2007). Tour operators, bird watchers and the seafaring public need to be educated about the risks this practice poses for albatrosses and giant petrels, and encouraged to advocate, promote and practice safe and environmentally responsible wildlife observations at sea. It is perhaps more appropriate for tourists and bird watchers to observe seabirds displaying their natural behaviours, rather than scavenging for an artificial food source. The practice needs to be discouraged through responsible tour operators, ornithological societies and other non-government organisations (e.g. the International Association of Antarctica Tour Operators — IAATO, Birds Australia, BirdLife International).

2.11.1 Issues relating to discards

- The discharge of offal is prohibited by regulation in Australian Antarctic trawl and longline fisheries. Regulations restricting offal discharge also apply to all longline vessels operating in Commonwealth manged fisheries via implementation of the Longline Fishing Threat Abatement Plan (2006): all vessels are required to retain all offal during line setting. However offal discharge remains largely unregulated, and thus a potential problem, in most of Australia’s trawl fisheries.
- Some albatrosses and giant petrels may become preoccupied with scavenging offal discards from fishing vessels making them less inclined to return to their nest.
- More research is needed to determine the extent and impact of association between fisheries, and their discard practices, and albatrosses and giant petrels.
- The practice of intentionally providing food to seabirds by commercial tour operators and birdwatchers needs to be actively discouraged through existing Codes of Practice, such as those employed by the International Association of Antarctica Tour Operators, and through education and advocacy with ornithological non-government organisations such as BirdLife International and their Australian partner Birds Australia.

2.12 Marine Pollution

(i) Chemical Contaminants

Chemical contaminants are almost universal. They can be categorised into two broad types, persistent organic pollutants (POPs) and heavy metals. Persistent organic pollutants, such as polychlorinated biphenyls (PCBs) were first introduced into the environment in the 1930s for a variety of industrial purposes (notably the plastic industry). Chlorinated hydrocarbon insecticides (including DDT) were introduced soon after (Moriarty 1975). Heavy metals, such as arsenic, mercury and cadmium, were introduced into the environment, particularly prior to the 1970s through uncontrolled industrial wastes that were deposited or stored at a variety of land- and water-based disposal sites (reviewed in Moore and Ramamoorthy 1994). Both POPs and heavy metals continue to be deposited into the environment, through the use of insecticides, herbicides, fungicides, coal and petroleum byproducts, and for dozens of other industrial, rural and domestic purposes.

Contamination with these chemicals is now global in nature. Elevated levels of POPs and heavy metals can be found in the plasma of adults, chicks and eggs of seabirds from every continent (including Antarctica) and virtually all islands across the globe (Croxall *et al.* 1984a, 1984c; Auman *et al.* 1997; Ludwig *et al.* 1998).

Organochlorines and heavy metals degrade very slowly in the environment. These chemical contaminants are retained by organisms and passed along the trophic levels of the food chain, becoming increasingly concentrated in the tissues of each higher consumer (a process known as biomagnification). Consequently, top order predators, such as albatrosses and giant petrels, may consume potentially hazardous levels of synthetic chemicals. Furthermore, because albatrosses and giant petrels are long-lived and typically highly dispersive species, they have even greater opportunity to accumulate high levels of chemical contaminants (Muirhead and Furness 1988; Luke *et al.* 1989; Lock *et al.* 1992; Auman *et al.* 1997; Ludwig *et al.* 1998; Stewart *et al.* 1999).

Within twenty years of their introduction, persistent organic pollutants were clearly implicated in the decline of a number of predatory seabirds (Moriarty 1975). Elevated levels of POPs can have deleterious population level effects through diminished reproductive success caused by eggshell thinning, embryo inviability and offspring deformities (Croxall *et al.* 1984a; Ludwig *et al.* 1998).

Residue levels of PCBs, other organochlorine products and mercury in the body tissues of northern and southern giant petrels increased, and in some cases doubled, between 1978 and 1983. The increases in chemical contaminants have been explicitly implicated in their population declines (Luke *et al.* 1989). Excessive loads of organochlorine compounds have also been located in the plasma of adults and chicks of black-footed albatrosses (*Diomedea nigripes*) on Midway Atoll, contributing to population declines (Auman *et al.* 1997; Ludwig *et al.* 1998).

Several studies have revealed that giant petrels and a number of albatross species possess unusually high concentrations of certain heavy metals, in particular cadmium and mercury (Muirhead and Furness 1988; Luke *et al.* 1989; Lock *et al.* 1992; Thompson *et al.* 1993; Stewart *et al.* 1999; Hindell *et al.* 1999: Becker *et al.* 2002). Mercury levels in the liver of long-lived species such as wandering albatrosses, sooty albatrosses and royal albatrosses are among the highest recorded for free-living birds (Stewart *et al.* 1999) and may be increasing (Becker *et al.* 2002). Indeed, the mercury concentration in the liver of one wandering albatross analysed is the highest recorded for any vertebrate (Muirhead and Furness 1988). Adult wandering albatrosses carry significantly higher mercury levels than juveniles. Similarly, cadmium levels in shy albatrosses are higher in adults than juveniles (Hindell *et al.* 1999).

The significance of heavy metals in the tissues of marine organisms is not well understood as trace amounts also occur naturally in marine ecosystems (Thompson *et al.* 1993). It is often difficult to determine if the concentrations measured in seabird tissues exceed natural background concentrations (Ludwig *et al.* 1998).

(ii) Fuel and oil spills

Bulk fuel and/or oil spills also have the potential to affect large numbers of seabirds. Birds coming into contact with oil can become physically smothered. The matting of the plumage by the oil allows water to penetrate the air spaces between the feathers and the skin, greatly reducing the bird’s insulation and waterproofing, often resulting in mortality. The increased heat-loss results in an increased metabolism of food reserves in the body which, if not countered by a corresponding increase in the food intake, may lead to emaciation. The risk of starvation is further heightened, as a severely oiled bird is unable to hunt and capture prey efficiently. Furthermore, the matted plumage reduces the bird's buoyancy and may cause them to sink and drown (Baker 1983; GESAMP 1993).

On a broader ecological scale, oil may be retained in sediments for many years, leading to the temporary or permanent loss of species critical to the ecological balance of a habitat. In addition, crude oil is essentially a mixture of many hydrocarbon compounds, some of which are toxic and/or persistent. These can accumulate in the marine food chain (described above) and may potentially lead to lethal or sub-lethal changes in metabolic functions (Baker 1983; GESAMP 1993). Since albatrosses and giant petrels spend much of their time on the sea surface, they are particularly vulnerable to the hazards of oil or fuel spills.

(iii) Marine debris

Marine debris is one of the world's five major marine pollutants (ANZECC 1995) and is increasing worldwide. The disposal of plastic materials at sea is totally prohibited by the *International Convention for the Prevention of Pollution from Ships* (MARPOL) to which Australia is a signatory, but the disposal of other types of garbage is permitted from vessels more than 12 nautical miles from land.

'Injury and fatality to vertebrate marine life caused by ingestion of, or entanglement in, harmful marine debris' was listed as a key threatening process under the EPBC Act in 2003 and a draft Threat Abatement Plan (TAP) is currently being developed. Eight of the albatross species and the northern giant petrel are listed in the draft TAP as being adversely affected by ingestion of, or entanglement in, harmful marine debris.

Harmful marine debris refers to all plastics and other types of debris from domestic or international sources that may cause harm to vertebrate marine life. This includes land-sourced waste and garbage, abandoned fishing gear from recreational and commercial fisheries, and ship-sourced solid, non-biodegradable, floating materials disposed of at sea. Most of the marine debris affecting albatrosses and giant petrels appears to derive from material jettisoned by vessels at sea (Huin and Croxall 1996; Taylor 2000a).

Studies have documented impacts of harmful debris on marine wildlife in all of the world's oceans and available information suggests that at least 20 threatened species are being harmed and killed by marine debris. Marine debris can impact upon albatrosses and giant petrels through ingestion or entanglement (Derraik 2002).

Many albatross and giant petrel species ingest considerable quantities of plastic and other marine debris. Ingestion of debris has a wide range of lethal or sub-lethal effects. The debris can cause physical damage, or perforation, mechanical blockage or impairment of the digestive system, resulting in starvation. Some plastics are also a source of toxic pollutants, which are released into the blood stream as the bird's digestive system attempts to break down the substance (Ryan 1988; Ryan *et al.* 1988). The subsequent reduction in fitness can lower the bird's ability to reproduce successfully, catch prey and/or avoid predation (Fry *et al.* 1987; Sileo *et al.* 1990).

Albatross and giant petrel chicks appear to be at greater risk than adults because of their high rates of ingestion and low frequency of regurgitative casting of indigestible material. When the plastics are regurgitated to chicks, the physical impaction and internal ulceration are likely to lower post-fledging survival. In addition, the chick receives less food, lowering its nutrient intake and increasing its chances of starvation (Fry *et al.* 1987; Sileo *et al.* 1990) and dehydration.

Plastic ingestion affects many Australian breeding albatrosses and petrels (Baker *et al.* 2002). In a study of the stomach contents of 540 shy albatross chicks that had recently died of natural causes, 1% of stomachs contained plastic debris, ranging from segments of plastic bags to solid, coloured pieces of plastic (Hedd and Gales 2001). Wandering, black-browed, and grey-headed albatrosses and southern giant petrels have all been observed regurgitating plastic debris to their chicks on breeding sites outside of Australia (Huin and Croxall 1996).

Plastic ingestion has also been observed affecting Antipodean, Tristan, Laysan, northern royal, southern royal and yellow-nosed albatrosses (Fry *et al.* 1987; Ryan 1987; J. Cooper pers. comm., in Gales 1993; J.P. Croxall pers. comm., in Gales 1993; Robertson 1998; A. Wiltshire pers. comm.). Ninety per cent of Laysan albatross chicks had plastic items lodged within their upper gastrointestinal tract (Fry *et al.* 1987). It is likely that most or all other species ingest plastic debris without it being observed or documented.

Some seabirds are also killed after becoming entangled in marine debris (Nel and Nel 1999). Such entanglement can constrict growth and circulation, leading to asphyxiation. Entanglement may also increase the bird's drag coefficient through the water, causing the animal to die due to its reduced ability to catch prey or avoid predators. The rate of this source of mortality remains completely unknown.

(iv) Food discharges

Considerable quantities of food waste can be generated at a rapid rate in ships, particularly those with large numbers of people onboard. By virtue of the amounts involved and its nature, food waste is potentially the most difficult to manage component of a ship's garbage stream. In many sea areas it is dealt with by direct discharge to sea. However, disposal to sea is not always possible due to restrictions imposed by MARPOL 73/78 and other marine pollution control instruments. Only minimal attention is paid to food waste management by some ship and port operators and advisory bodies, and there is little information in the available literature. The determination that management of ships' food waste is inconsequential can be incorrect (Polglaze 2003). Potentially, in areas of regular discharge, it can have environmental impacts via turbidity, nutrients, disease and so forth. The impact on albatrosses and giant petrels is unknown.

2.12.1 Issues relating to marine pollution

- The presence of hatching failure due to eggshell thinning, oiled birds and regurgitated marine debris at albatross and giant petrel breeding colonies should be monitored and documented.

- Accurately quantifying the mortality associated with marine debris entanglement would be extremely difficult, if not impossible, as most deaths are likely to occur at sea.
- The Commonwealth of Australia passed the *Protection of the Sea (Prevention of Pollution from Ships) Act 1983* to give effect to the International Convention for the Prevention of Pollution from Ships (MARPOL) Annex V relating to the management of ship borne rubbish. The main objective of MARPOL is to minimise marine debris which, if achieved, would significantly reduce the chances of ingestion or entanglement by albatrosses and giant petrels.
- Marine pollution is a global phenomenon that needs to be addressed and rectified through international conservation and other forums.
- A Threat Abatement Plan for the EPBC listed Key Threatening Process of *injury and fatality to vertebrate marine life caused by ingestion of, or entanglement in, harmful marine debris* should be finalised and its recommendations implemented.

2.13 Climate change

Climate change is one of the major factors likely to affect the earth's ecosystems in the coming decades. The global average surface temperature has increased over the 20th Century by around 0.6°C and this has been associated with changes in weather patterns, precipitation, sea-temperatures and sea-level (Ainley 2000; IPPC 2001). Average sea temperatures in the Southern Ocean have increased by over 0.5°C since the 1950s (Allan *et al.* 1996; Cunningham & Moors 1994) and the pH (the measure of acidity) is decreasing. There are also regional sea temperature changes that are part of the ENSO (El Nino-Southern Oscillation) cycle that cause sea temperatures to warm or cool in some years (Allan *et al.* 1996).

Of the bird species listed on the CMS, 84% face some threat from climate change, almost half because of changes in water regime; this is equivalent to the (summed) threats due to all other anthropogenic causes (Robinson *et al.* 2005). Further understanding of how populations will respond, through knowledge of climate impacts on breeding performance and survival, will be necessary for successful predictions of impacts.

A major effect of climate on migratory species will be changes in prey distribution (Robinson *et al.* 2005). Such changes are a major threat in marine ecosystems. Large shifts in distribution (as much as 10° latitude) and abundance (with declines to a hundredth or a thousandth of former values) of plankton communities in response to changes in sea surface temperature have already been demonstrated (particularly for krill, a key component of marine foodwebs). These changes are likely to have profound effects on seabirds as they will influence the distribution and abundance of plankton, crustaceans and predatory fish and squid that feed on these species at oceanic upwellings and convergence fronts (Wormworth and Mallon 2006). Some impacts may be gradual such as a reduction in breeding success as birds need to forage further from colonies and hence chicks starve in nests or eggs are abandoned by incubating partners.

Other impacts are catastrophic and involve mass food failure in one or more seasons. Adult birds may occasionally starve but typically birds do not attempt to nest, breeding attempts are abandoned or breeding success is very low in catastrophe years. The occurrence of such catastrophes has been documented in the tropical Pacific (Warham 1996) but similar events are less well known here. For example, northern royal albatross at Taiaroa Head have had problems in recent years with high temperatures and low soil moisture levels causing eggs to dry out (resulting in hatching failure), incubating adults suffering heat stress, and an increased impact of fly strikes on hatching chicks (Robertson 1998). Albatrosses whose pelagic wanderings circumnavigate the globe are likely to be affected by changing ocean currents, particularly if altering their routes brings them into greater conflict with fishing activities (Tuck *et al.* 2001; Croxall *et al.* 2005).

Weimerskirch *et al.* (2004) showed that air temperatures steadily increased over the past 50 years in the southern Indian Ocean, particularly in the subantarctic sector, and at the same time, with a time lag of 2-9 years with temperatures, the population size of most seabirds monitored on several breeding sites decreased severely. These changes, together with the indications of a simultaneous decrease in secondary production in subantarctic waters and the reduction of sea-ice extent further south, indicate that a major system shift has occurred in the Indian Ocean part of the Southern Ocean, illustrating the high sensitivity of marine ecosystems, and especially upper trophic level predators such as albatrosses, to climate change.

The outbreak of avian cholera on Amsterdam Island may have been favoured by the marked increase in temperature that has taken place in the Indian Ocean during the 1970s (Weimerskirch et al. 2003; Weimerskirch 2004). One effect of climate change may be to act as a catalyser of epizootics, especially infectious diseases such as avian cholera, which may pose a major threat for albatrosses and giant petrels in the future, especially in the Southern Ocean environment where ecosystems have evolved in isolation (Weimerskirch 2004).

2.13.1 Issues relating to climate change

- Loss of climatic habitat, including an increase in sea surface temperature, caused by anthropogenic emissions of greenhouse gases is a potential threat to all Australian seabirds.
- Management of this process requires both domestic and international action, and is beyond the scope of species recovery plans.
- Given the incomplete knowledge of the oceanic distribution for the species covered by this document, and their prey, it is difficult to assess the impact of climate change and thus develop appropriate management responses.
- Environmental conditions should be monitored in parallel with albatross and giant petrel breeding parameters to determine any correlation.

2.14 Population Monitoring and Foraging Ecology Programs

Management of small populations of any organism requires adequate biological and ecological knowledge to ensure appropriate conservation action. The efficacy of past and current Recovery Plans can be measured via a system of regular population monitoring programs. Such programs calculate current population sizes, and when conducted over several years allow an assessment of a population's status. Furthermore, monitoring programs allow quantification of adult survival and juvenile recruitment rates to populations, providing fundamental information concerning the future viability of populations. Hence, these programs supply the vital demographic information necessary for the recovery process.

Unfortunately for many populations of albatrosses and giant petrels, there is little relevant biological and ecological data available, although Australian breeding populations fare better than many other populations. Around 140 breeding populations of albatross exist outside of the Australian EEZ. Despite increased efforts in population monitoring, knowledge of the status of two-thirds of these populations is still lacking. About 50 albatross populations contain less than 100 annual breeding pairs, making them extremely vulnerable to stochastic events (Gales 1998). There is limited information regarding the current status of giant petrel populations breeding outside of Australian territory. Many of the populations have not been surveyed for at least 15 years, making any assessment of their current status unreliable.

There are 12 breeding populations of albatross within areas under Australian jurisdiction (Table 1.4). At least five of these colonies contain critically low populations, numbering 200 or fewer breeding pairs. Seven populations are currently being monitored annually, which has enabled their status to be identified and provides essential information regarding the viability of all in the future. The current status is known for only two of the seven giant petrel breeding populations within areas under Australian jurisdiction. Four populations have not been surveyed for at least a decade. Consequently, their current status or trend cannot be accurately assessed (Table 1.4).

Annual population monitoring programs exist for albatrosses and giant petrels breeding on Macquarie Island, Albatross Island, Pedra Branca and the Mewstone. Most of these programs include banding and recapture or resighting of individual birds to permit estimation of demographic parameters (juvenile and adult survival, age of first breeding, recruitment), monitoring of breeding success (hatching and fledging), and population surveys. They have also collected information on basic breeding biology, particularly frequency of breeding, as well as supporting distributional studies to identify key foraging areas for both breeding and nonbreeding birds. Since the mid 1990s, these monitoring programs have been funded by DPIPWE and the AAD, with additional logistical and other support from the AAD and the Antarctic Science Advisory Committee for work on Macquarie Island.

Since the mid 1980s, the AAD has opportunistically visited and estimated the size and breeding success of southern giant petrel colonies breeding at Giganteus Island, Hawker Island and the Frazier Islands within the Australian Antarctic Territory. A monitoring program using aerial photography and automated ground-based cameras began at Hawker Island in 2009 and continued in 2010 and 2011, with additional automated camera deployments planned for some of the Frazier Islands in the 2011/12 summer. There are currently no systematic monitoring programs for the populations breeding on Heard Island or the McDonald Islands. Systematic, non-intrusive surveys of colonies at all of these locations conducted over many years are needed to determine the size and long term status of all Australian breeding populations.

Investigations into remote monitoring techniques such as aerial photographic surveys have returned encouraging results. Photographic censusing techniques are now routinely used for shy albatross colonies in Australia (DPIPWE unpublished), and have been recently adopted for surveys in South America and New Zealand (Arata *et al.* 2003; Robertson *et al.* 2003b; Lawton and Robertson 2006; Baker *et al.* 2007b; Robertson *et al.* 2007). Remote techniques should be further refined and employed wherever appropriate. The availability of suitable aircraft within operational range of some of these colonies will dictate the suitability of this technique.

Collection of biological and demographic information often involves levels of research intensity that can potentially disturb breeding birds. There may be colony-specific differences in these parameters, i.e. levels of disturbance that are tolerated by one population may not be tolerated by another. Existing monitoring programs have sought to ensure minimal disturbance and care needs to be taken in initiating any future studies to maintain consistent research and monitoring protocols.

ACAP continues to prioritise and encourage the collection of demographic data for populations of albatrosses and giant petrels as essential in assisting Parties to the Agreement to prioritise their actions and measure progress in meeting the Agreement's objectives. Maintaining the monitoring programs that exist for Australia's populations is necessary to improve knowledge of population processes and will ensure Australia continues to meet its obligations under ACAP. Similar studies should be initiated on other populations where these are practical and feasible.

ACAP recently completed a series of global species assessments for all species currently listed on Annex 1 of the Agreement (STWG 2010). These assessments include comprehensive information on population status and trends, taxonomy, breeding locations, threats, foraging distribution and overlap with fisheries operations and organisations, and are an important element in maintaining contemporary data on population trends.

The Australian population monitoring programs have provided a platform for foraging ecology studies. Satellite-tracking (PTT) and geolocator (GLS) equipment has been used on the four species of albatrosses and two giant petrel species that nest on Macquarie Island; shy albatrosses from Albatross Island, Pedra Branca and the Mewstone; and black-browed and light-mantled albatrosses from Heard Island. These studies have aimed to reveal the at-sea distribution of Australia's breeding albatrosses and giant petrels throughout their annual and life cycles, and the degree of overlap with RFMOs and Marine Protected Areas (MPAs) (R. Alderman unpublished; Hedd *et al.* 2001; Lawton *et al.* 2007; Terauds *et al.* 2006a; Trebilco *et al.* 2006; Weimerskirch and Robertson 1994). Funding for this research has been provided by DPIPWE and the AAD, and all tracking data have been submitted to the global Procellariiform tracking database, which contains data from multiple data owners and is supported and maintained by BirdLife International (BirdLife International 2004). Recognising the substantial potential of these remote-tracking data for conservation applications, BirdLife International has used the database to assist RFMOs in risk assessments for fisheries management (e.g. Small and Taylor 2006) with the agreement of the data owners and further financial assistance from ACAP.

2.14.1 Issues relating to monitoring and foraging ecology

- Population monitoring programs are essential in determining the viability of populations and for the planning and implementation of conservation actions.
- The current status of several albatross and giant petrel populations breeding within areas under Australian jurisdiction is unknown. This situation needs to be remedied.
- Existing population monitoring programs for albatrosses and giant petrels breeding on Macquarie Island, Albatross Island, Pedra Branca, and the Mewstone must be maintained and conducted regularly.
- Remote population monitoring techniques need to be further developed and used wherever appropriate. These techniques may be useful for making assessments of representative populations breeding on Heard Island, the McDonald Islands and in the AAT.
- Tracking studies have provided valuable information on the at-sea distribution of albatrosses and giant petrels, and particularly in relatation to overlap with fisheries and MPAs. A review of all tracking data collected was undertaken by the AAD and DPIPWE to determine information gaps for albatrosses and petrels and provided recommendations for future research.
- Submission of Australian population and demographic data to ACAP should continue to be supported.
- Following analysis, data owners should be encouraged to submit all tracking data sets to BirdLife's global Procellariiform tracking database so that it can contribute to global assessments of bycatch, and other fisheries management tasks as required.

2.15 Education and Communication Strategies

Education, public awareness and community involvement are critical components of the conservation and recovery process. The impact of several threats to albatrosses and giant petrels can be greatly decreased via the development of educational strategies targeting (i) commercial and recreational fishers, (ii) visitors to breeding colonies, and (iii) the general public.

(i) Commercial and recreational fishers

Commercial and recreational fishers need to be encouraged to employ effective bycatch mitigation measures. This can best be achieved by educating them on the ecological importance and economic gains of using such measures, and the central importance of albatrosses, giant petrels and other wildlife to marine ecosystems. The Longline Fishing Threat Abatement Plan (2006) lists the following actions:

1. AFMA and the Department of the Environment and Heritage (now Department of Sustainability, Environment, Water, Population and Communities) will report as

appropriate to key stakeholders on the analysis of bycatch data and seabirds collected in relation to achieving the objectives of the Threat Abatement Plan.

2. AFMA will implement extension and training programs for longline fishers where appropriate.

Other threats that could be reduced through a targeted educational strategy include trawl and other types of fisheries interactions (Section 2.2, 2.3 and 2.4), and the intentional shooting of albatrosses and giant petrels (and other wildlife) by recreational and commercial fishers (Section 2.5).

(ii) Visitors to breeding colonies

Section 2.7 discusses the need for visitors to albatross and giant petrel breeding colonies to be made aware of their potential impact on nesting attempts. Educational material regarding the impacts of wildlife disturbance and the need to comply with quarantine procedures to prevent the introduction of exotic species on breeding islands should continue to be provided to all tourists and ANARE expeditioners prior to arrival at Macquarie Island, the AAT and Heard Island (also see Appendix 2).

(iii) The general public

Marine pollution has been identified as a potential threat to albatrosses and giant petrels (Section 2.12). The general public needs to be informed of the environmental impacts of using industrial, agricultural and domestic chemicals and the central importance of conserving albatrosses, giant petrels and other wildlife.

Public awareness of, appreciation and support for conservation of albatrosses and giant petrels and their breeding habitats need to be raised. Management plans for the Macquarie Island Nature Reserve encourage off-reserve educational activities as far as possible through films, media coverage, exhibitions, books, the internet and other means (Parks and Wildlife Service 2006). Similarly, the *Heard Island and McDonald Island Marine Reserve Management Plan 2005* also recommends off-site measures to present the Reserve to the community. The public should also be encouraged to similarly regard all other albatross and giant petrel breeding sites.

2.15.1 Issues relating to education

To reduce threats to albatrosses and giant petrels:

- Public awareness of the threats to albatrosses and giant petrels needs to be increased.
- Strategies are needed to increase fishers' awareness of seabird bycatch issues and to promote continuing communication between fishers, researchers and fisheries managers.

2.16 International Conservation Agreements and Obligations

The highly dispersive and migratory nature of albatrosses and giant petrels is well known. Populations that breed on Australian islands may spend a large proportion of their lives foraging outside of the EEZ. Thus, it is significant that many of the threats affecting albatrosses and giant petrels within the EEZ are also occurring outside it.

In addition, some human-induced threats only occur outside of the EEZ. Large-scale driftnet fisheries operated legally until the end of 1992 when the UN General Assembly imposed a global moratorium on driftnetting due to the excessive levels of bycatch. It is possible that a significant level of illegal driftnetting persists on the high seas and in some EEZs and coastal regions (Alexander *et al.* 1997). This could continue to have a negative impact on the survival of albatrosses, giant petrels and other marine birds and mammals. Similarly, illegal, unreported and unregulated fishing (i.e. in broad terms, fishing in contravention of management arrangements adopted by coastal and flag States and RFMOs), especially longline fishing, is also known to be having a potentially significant adverse impact on albatrosses, giant petrels and other seabirds.

For these reasons, it is most likely to be impossible to restore all populations breeding on Australian islands solely by eliminating threats occurring within Australia's EEZ. Hence, it is imperative that international agreement is reached in a variety of fisheries, conservation and other fora and management and conservation measures to ameliorate threats caused by human activity are implemented throughout the entire range of albatrosses and giant petrels.

The Australian Government is actively pursuing international action through various fora, including:

- The Convention on the Conservation of Migratory Species of Wild Animals (CMS);
- The Agreement on the Conservation of Albatrosses and Petrels (ACAP);
- RFMO's including the Commission for the Conservation of Southern Bluefin Tuna (CCSBT), Indian Ocean Tuna Commission (IOTC), Commission for the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean (WCPFC), and the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR);
- The Food and Agriculture Organisation of the United Nations' Committee on Fisheries (COFI); and
- The International Convention for the Prevention of Pollution from Ships (MARPOL).

(i) Convention on the Conservation of Migratory Species of Wild Animals

In 1997 Australia successfully proposed that all Southern Hemisphere albatross species be listed under the Convention on the Conservation of Migratory Species of Wild Animals (CMS). This was an important step toward promoting a cooperative framework for the conservation and management of Southern Hemisphere albatrosses. The November 1999 Conference of Parties to the CMS recommended that all range States actively participate in the development and successful conclusion of a regional agreement for the conservation of albatrosses. In 1999 the development of the

Agreement on the Conservation of Albatrosses and Petrels, or ACAP, was commenced (see (ii) below).

(ii) Agreement on the Conservation of Albatrosses and Petrels (ACAP)

The Agreement on the Conservation of Albatrosses and Petrels, or ACAP, has been developed under the auspices of the *Convention on the Conservation of Migratory Species of Wild Animals* (CMS). It seeks to coordinate international activity to mitigate known threats to albatross and petrel populations throughout the Southern Hemisphere. In particular, Annex 1 of the Agreement provides a framework for implementation of effective conservation measures for albatrosses and petrels.

ACAP entered into force in February 2004 and now has 13 Parties—Argentina, Australia, Brazil, Chile, Ecuador, France, New Zealand, Norway, Peru, South Africa, Spain, the United Kingdom and Uruguay. An Advisory Committee has been established to guide the implementation of the Agreement, which is served by four Working Groups: the Taxonomy Working Group, Status and Trends Working Group, Breeding Sites Working Group, and the Seabird Bycatch Working Group (www.acap.aq).

Through ACAP there is now a stronger and increasing international commitment to protect albatrosses and petrels, but much needs to be done to ensure the agreement becomes an effective mechanism to assist in eliminating threats to albatrosses and petrels, both at sea and on land, and ensuring that population declines are reversed. ACAP is in its infancy and awareness of the Agreement needs to be raised (Cooper *et al.* 2006). There are several important range States still to accede, notably major distant water fishing nations whose fishing fleets encounter albatrosses and petrels and those States with jurisdiction over breeding areas for the three recently added northern hemisphere albatross species. The Action Plan, established by Article VI of the Agreement, remains to be fully implemented, and further capacity building is needed in range States that require training, information and institutional support.

(iii) Regional Fisheries Management Organisations (RFMOs)

Regional Fisheries Management Organisations are of central importance to sustainable management of the world's oceans. Their obligation to conserve fish stocks and threatened species such as albatrosses has been established by legal instruments, including agreements such as the Code of Conduct for Responsible Fisheries and the UN Fish Stocks Agreement.

Small (2005) evaluated the 14 RFMOs whose areas overlap with albatross distribution, assessing their performance in fulfilling their duties to minimise bycatch, especially albatross bycatch, within their fisheries. The five RFMOs with the greatest overlap with albatross distribution were the Commission for Conservation of Southern Bluefin Tuna (CCSBT), followed by the Western and Central Pacific Fisheries Commission (WCPFC), the Indian Ocean Tuna Commission (IOTC), the International Commission for the Conservation of Atlantic Tunas (ICCAT), and the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR). Only one of these, CCAMLR, has undertaken a wide range of measures that has reduced bycatch by over 99% (SC-CAMLR 2007) with no albatrosses and near-zero seabirds observed to be killed during the most recent seasons (2008-09). The tuna RFMOs lag behind considerably in comparison. The WCPFC only came into force in 2004, and while it has adopted some initial mitigation measures, these are not likely to be sufficiently effective or comprehensive enough to fully reduce bycatch. However, encouragingly, the WCPFC is presently part way through an ecological risk assessment process which, once complete, may result in revised bycatch conservation measures being adopted to significantly improve the now relatively weak and outdated Resolution 2007-04 adopted in December 2007. CCSBT has encouraged its vessels to use a bird-scaring line south of 30°S for over 10 years, and was the first tuna RFMO to agree on a mitigation measure for seabird interactions in longline fisheries. However, the CCSBT's early momentum in respect of seabird bycatch has not been sustained and few tangible benefits have been delivered with repeated refusals to adopt seabird bycatch mitigation measures proposed by several Parties. Since 1997, the seabird mitigation measure, which is not regarded as an effective or comprehensive approach to seabird bycatch mitigation, has not been amended to reflect international developments in mitigation measures. The CCSBT is also failing to monitor or assess the compliance with, or effectiveness of, this measure, and was recently assessed as lagging behind other RFMOs in management of non-target species (Lack 2007). In 2008, rather than adopt a binding resolution on seabird bycatch mitigation measures, the Commission agreed to a non-binding undertaking to comply with the seabird bycatch mitigation requirements of the WCPFC and the IOTC when fishing for SBT in those Convention Areas

Of the remaining two important RFMOS, IOTC adopted modest, but useful, seabird bycatch mitigation measures in June 2008 and March 2010 and ICCAT has only recently considered, albeit unsuccessfully, adopting mitigation measures to reduce bycatch of non-target species. Despite considerable evidence that their seabird bycatch is globally significant, none of the four tuna RFMOs yet have in place comprehensive onboard observer programs – which are essential to understanding the extent of seabird bycatch and guiding, refining and monitoring the effectiveness of the mitigation measures being implemented – nor do they even have a standardised, mandatory reporting of bycatch.

The need for improved performance by regional fisheries management organisations (RFMOs) in their data collection, management and enforcement of management measures for albatrosses and giant petrels and other non-target species is now widely accepted. In just the last few years, *inter alia*, the United Nations General Assembly, the 2006 Review Conference on the Implementation of the United Nations Fish Stocks Agreement, the 27th meeting of the Food and Agriculture Organisation of the United Nations Committee on Fisheries (COFI) and the joint meetings in Kobe, Japan in 2007, in San Sebastian, Spain in 2009 and in Brisbane, Australia in 2010 of tuna RFMOs have each identified this need (Anon. 2007; Lack 2007).

There is considerable information and advice available to the tuna RFMOs in relation to best practice in seabird mitigation measures. While no single measure or even combination of measures is likely to completely resolve the problem, there is sufficient evidence to indicate that combinations of measures are effective in significantly reducing seabird interactions. In 2006 and 2007, international workshops on mitigation of seabird catch in pelagic longline fisheries endorsed the approach proposed by the WCPFC of identifying combinations of measures that could be used and selected on the basis of the operational characteristics of the vessel

(Melvin and Baker 2006; ACAP Seabird Bycatch Working Group 2007). Later meetings of bycatch mitigation experts have further elaborated the most effective components of this concept and several research findings support this approach as current best practice (Melvin 2010, Robertson 2010, ACAP Seabird Byactch Working Group 2010).

Small (2005) and CCAMLR also identified a more general concern for RFMOs to generate within their members the collaboration and political will to agree on and adopt effective mitigation measures. These measures need to be applied both within their domestic jurisdictions and on the high seas. Until all members of the tuna RFMOs acknowledge their responsibility to adopt a precautionary and ecosystem-based approach to management there is little likelihood of better assessment and management of impacts on albatrosses and giant petrels. This acknowledgement needs to include equal priority to data collection, research and management considerations for target and non-target species. Some RFMOs, such as CCSBT, have been slow or failed to recognise the merits of a risk-based approach to management of non-target species (Lack 2007; Waugh *et al.* 2007). This approach is embedded in CCAMLR management practices (Waugh *et al.* 2007) and slowly becoming increasingly common in other RFMOs. ICCAT has agreed that an ecological risk assessment framework may be a good way to prioritise research activities and is undertaking such an assessment using the available data on species taken by ICCAT fisheries (ICCAT Sub-committee on Ecosystems, in Lack 2007). Similarly, the WCPFC Scientific Committee has endorsed ecological risk assessment (ERA) as an appropriate way to assist in prioritising species for management action or further research (WCPFC Scientific Committee, in Lack 2007) however its ERA, begun in 2008, had not been concluded as at the end of 2010 and the WCPFC prescribed bycatch mitigation measures lag well behind the findings from the last two to three years research.

Pivotal to the success of CCAMLR in reducing seabird bycatch to zero or near-zero levels has been the formation of the specialist ad hoc Working Group on Incidental Mortality Associated with Fishing (refer also Section 2.1.1). This group was established in 1994 to specifically provide advice to CCAMLR's Scientific Committee on seabird interactions. The group meets annually and reviews all fishing observer and other data from the previous year, together with fishing proposals for the forthcoming year. Matters considered include the performance of each fishing vessel in avoiding bycatch; the effectiveness of mitigation measures in use; recent developments with mitigation and their applicability to CCAMLR fisheries; an annual risk assessment for all fisheries to identify the risk of capture of seabirds in fishing operations; and a review of existing conservation measures. Updated advice from the IMAF Working Group is taken to the CCAMLR Commission via the Scientific Committee on an annual basis, ensuring best-practice seabird bycatch mitigation measures and advice can be rapidly adopted. At this stage no other RFMO has a similar process in place. Adoption of this model by all tuna RFMOs would be a significant step toward substantially reducing incidental mortality of albatrosses and giant petrels in high seas fisheries.

(iv) FAO's Committee on Fisheries (COFI)

In 1998 the Food and Agriculture Organisation of the United Nations (FAO) Committee of Fisheries (COFI) implemented an *International Plan of Action for Reducing Catch of Seabirds in Longline Fisheries* (IPOA). The objective of the IPOA is to reduce the primary global threat to albatrosses and giant petrels (the incidental catch of seabirds in longline fisheries) wherever it occurs. FAO encourages all States and fishing entities to implement the voluntary IPOA.

The IPOA stipulates that States with longline fisheries should conduct an assessment of these fisheries to determine whether a bycatch problem exists. When this is the case, a National Plan of Action (NPOA) for reducing the incidental catch of seabirds in longline fisheries should be adopted. The Longline Fishing Threat Abatement Plan (see Section 2.1) fulfils Australia's obligation to the FAO's IPOA within Commonwealth-managed fisheries, which encompass the very large majority of longline effort within Australian waters. Australia has developed a draft NPOA, which will apply to all Commonwealth longline fisheries, however this draft was developed before the FAO adopted its 2009 Technical Guidelines for the IPOA (called Best Practices to Reduce Incidental Catch of Seabirds in Capture Fisheries) which recommend NPOAs also be extended to cover other relevant fishing gears, including trawls and gillnets. At the time of writing, it had not be decided how or when these other gear types would be incorporated in the draft NPOA.

The objective of the NPOA-Seabirds is to facilitate a nationally coordinated approach to reduce seabird interactions in all Australian longline fisheries. The NPOA-Seabirds is still in draft form. However, based on an assessment of the problem of incidental bycatch in Australia (Commonwealth of Australia 2003), five key directions have been identified to reduce seabird mortality in Australia's longline fisheries:

- implementation of effective mitigation measures to reduce the incidental catch of albatrosses, giant petrels and other seabirds in longline fisheries;
- development and maintenance of a comprehensive understanding of the type and extent of interactions between seabirds and longline fisheries within the EEZ;
- development of national interaction reporting criteria (logbook, observer, and research) to enable the assessment of interactions across fisheries and jurisdictions;
- facilitation of the research and development of mitigation measures to reduce the incidental catch of seabirds; and
- raising awareness about bycatch in longline fisheries and effective mitigation measures.

An inter-governmental working group will be created to oversee the implementation of the NPOA once it is finalised, although it is proposed that each State and Territory jurisdiction develop their own suite of practical mitigation measures, both mandatory and voluntary.

(v) The International Convention for the Prevention of Pollution from Ships (MARPOL)

Australia is a signatory to the *International Convention for the Prevention of*

Pollution from Ships (MARPOL). The main objective of MARPOL is to minimise marine debris, which is one of the world's five major marine pollutants (ANZECC 1995) and increasing worldwide. Ingestion of marine debris or entanglement is identified in this document as posing a threat to the conservation of albatrosses and giant petrels (see section 2.12).

Australia has actively sought to fulfil it's obligations under MARPOL and passed the *Protection of the Sea (Prevention of Pollution from Ships) Act 1983* to give effect to Annex V of the Convention. This Act seeks to control the management of ship borne rubbish by Australian vessels, which would significantly reduce the chances of ingestion or entanglement by albatrosses and giant petrels.

2.16.1 Issues relating to international agreements

- Albatrosses and giant petrels are particularly vulnerable on the high seas where the lack of national jurisdiction makes it essential that countries assume their shared responsibility in the conservation of this common natural heritage.
- Interactions with authorised longline and trawl vessels, and illegal, unreported and unregulated driftnet and other vessels, on the high seas is causing significant mortality of albatrosses and giant petrels and needs to be addressed by international conservation and fishing forums.
- RFMOs are of central importance to minimising mortality of albatrosses and giant petrels on the high seas. While CCAMLR has successfully minimised bycatch in Antarctic waters, four tuna RFMOs (CCSBT, WCPFC, IOTC and ICCAT) must make substantial progress, such as by implementing mandatory implementation of ACAP's best-practice mitigation measures, before the threat of high seas fisheries-related mortality is diminished.
- Members of RFMOs need to generate the collaborative and political will to agree on and adopt effective mitigation measures and observer programs and rigorously and transparently apply them throughout their jurisdictions.
- Ecological risk assessment for both target and non-target species is one appropriate way for RFMOs to prioritise species for conservation action or further research. The adoption of a risk assessment framework has been pivotal in reducing bycatch of albatrosses and giant petrels in CCAMLR and should be urgently adopted in CCSBT, WCPFC, IOTC, ICCAT and other high seas fisheries where vessels interact with seabirds.
- As recommended for all fisheries in Sections 2.1 and 2.2, annual review of information on fishery bycatch and performance of mitigation measures in use, the seabird species that interact with a fishery, and what improvements in bycatch mitigation practice are required, would provide RFMOs with the information necessary to ensure the adoption of mitigation measures that are close to international best-practice at any time. Establishment of seabird bycatch working groups, modelled on CCAMLR's IMAF Working Group, to collect and review observer data and relevant scientific and other information would facilitate this process.

- Collation and provision of information from foraging and population studies to International Agreements such as ACAP and RFMOs should be encouraged to contribute to global conservation initiatives and improved fisheries management.

3 BIOLOGY AND ECOLOGY OF SPECIES

3.1 Species Breeding in Areas under Australian Jurisdiction

This section describes the breeding and non-breeding distributions, breeding biology, foraging ecology, and population status of each of the five albatross and two giant petrel species that breed on Australian islands. Further information on the species is well summarised and updated by BirdLife (2007) and also in the ACAP species assessments available at www.acap.aq.

3.1.1 Wandering Albatross *Diomedea exulans* Linnaeus 1758

Previous name

Wandering albatross *Diomedea exulans exulans*

Jurisdiction	Breeding locality
Australia	**Macquarie Island**
France	Crozet Islands, Kerguelen Islands
South Africa	Marion Island, Prince Edward Islands
Other	South Georgia (Islas Georgia del Sur)

Recent research by Alderman *et al.* (2005) suggests that wandering albatross populations on Macquarie, Crozet and Prince Edward Islands are genetically similar to each other.

Distribution

The wandering albatross disperses widely in all the southern oceans, from the edge of the pack ice (68°S), north to at least the Tropic of Capricorn and sometimes beyond. It approaches 10°S along the western coasts of South America and Africa, and vagrants have even been seen off California and in the northern Atlantic. In winter, wandering albatrosses are more often found north of the Antarctic Convergence (Blakers *et al.* 1984; Marchant and Higgins 1990; Nicholls *et al.* 1995, 1997, 2000).

Wandering albatrosses are highly dispersive. Several have been recovered more than 10 000 km from where they were banded, travelling 100–200 km a day (Jouventin and Weimerskirch 1990; Prince *et al.* 1992; Nicholls *et al.* 1992, 1996; Nicholls and Murray 1997; Weimerskirch *et al.* 1993; Walker *et al.* 1995). Individuals from Macquarie Island, South Georgia (Islas Georgia del Sur), Marion Island and the Crozets and Kerguelen Islands have all been recaptured off the NSW coast (Blakers *et al.* 1984; Battam and Smith 1993). Wandering albatrosses have been recorded off the coasts of southern Australia, from Fremantle in the west to Brisbane in the east, and occasionally north to the Whitsunday Passage (Blakers *et al.* 1984; Reid *et al.* 2002). This species also occurs in Australia's pelagic, offshore and inshore waters (even into harbours) at all times of year, though it is most common off south-east Australia (especially the Tasman Sea) from October–April (Battam and Smith 1993; Reid *et al.* 2002).

Comparisons of results of satellite tracking has revealed that distances and patterns of dispersal are variable between breeding stages and populations (BirdLife International 2004b and references therein). Breeding adults from Macquarie Island are known to forage in distant oceanic waters over 2 000 km away from Macquarie Island. In contrast, non-breeding birds from Macquarie Island are known to forage in waters north of Macquarie Island, including New Zealand shelf waters (DPIPWE unpublished information). Satellite telemetry of juvenile wandering albatrosses from the Crozet Islands showed an average distance covered of 184 000 km during their first year at sea, and restricted their dispersal to the subtropical Indian Ocean and the Tasman Sea (Weimerskirch *et al.* 2006).

Breeding biology

Wandering albatross pairs invest heavily in each breeding attempt, which lasts 55 weeks. Breeding is at least biennial if not longer (many successful breeders do not breed again for another three or four years and unsuccessful breeders often will not breed for two or three years (Croxall *et al.* 1990; Nel *et al.* 2002a). Breeding pairs return to the nest site between early November to early January, depending on location (Paulian 1953 and Mougin 1970, in Marchant and Higgins 1990); at Macquarie Island, most have returned by late November.

On South Georgia (Islas Georgia del Sur) 20% of wandering albatross re-use the same nest (Tickell 1968) and on the Crozet Islands 23.3% of breeders and 37.9% of failed breeders do (Marchant and Higgins 1990). Pair fidelity is high; only 0.7% of pairs divorced (Nel *et al.* 2002). However, at least on South Georgia (Islas Georgia del Sur) in 1998–1989, there were moderate levels of extra-pair paternity (6–21% of chicks; Burg and Croxall 2006).

Wandering albatrosses breed in loose colonies generally on exposed tussock covered ridges near the sea. The single egg is laid in December at Macquarie Island and between December and February at other locations (Croxall *et al.* 1990). Incubation lasts about 79 days. The chick hatches in February–April, and remains in the nest for another 277–304 days during which time it may build a new nest for itself (Croxall *et al.* 1990).

The chick fledges between mid-November and early February. By this time most of the next season's pairs have already arrived to breed (Paulian 1953, and van Zinderen Bakker 1971, in Marchant and Higgins 1990). At Macquarie Island, fledging occurs in early November to early January. From 1964 to 2004 mean breeding success (% of eggs which resulted in a fledgling) of wandering albatrosses on Macquarie Island was 64% (DPIPWE unpublished). At Marion Island between 1984–2001 breeding success was 75%, greater for older than younger pairs (Nel *et al.* 2002a). Mean breeding success at Bird Island, South Georgia (Islas Georgia del Sur) is 71% (Croxall *et al.* 1998), while the mean breeding success at Possession Island (Crozet Islands group) approaches 69% (Weimerskirch *et al.* 1997a).

Sex ratio of chicks varies with age and phenotypic quality of parents, leading to complex age structure (Weimerskirch *et al.* 2005). Higher quality pairs produce males. Overall, more male chicks fledge but males suffer higher mortality than females so that the sex ratio is balanced by the age of recruitment into the breeding population.

The immature birds remain at sea for the first 3–11 years of their life, until they return to their natal colony to breed. The young birds then begin pair formation, which usually takes another 2–3 years. Breeding eventually begins at 7–16 years of age, with females tending to breed at a slightly younger age than males (Weimerskirch and Jouventin 1987; Pickering 1988; Croxall *et al.* 1990). Although adults are highly philopatric to breeding sites, recently it has been recognised that there is some juvenile dispersal that can influence population dynamics via recruitment on other, sometimes distant, islands (Inchausti and Weimerskirch 2002).

Mean annual survival at Macquarie Island 1955–2000 was 95% for adults and 46% for juveniles (Teruads *et al.* 2003). These adult survival rates are higher than those recorded for wandering albatrosses at South Georgia (Islas Georgia del Sur) (91.9%) and Crozet (93.1%), while the juvenile survival rates are intermediate between other records (48.9% at South Georgia (Islas Georgia del Sur); 38.2% at Crozet; Croxall *et al.* 1998; Weimerskirch and Jouventin 1998).

Foraging ecology

Wandering albatrosses may form flocks of up to 50 individuals at rich food sources, particularly behind fishing vessels (Dixon 1933, in Marchant and Higgins 1990). They are voracious scavengers, out-competing all other seabirds for fishing discards and baited hooks (Weimerskirch *et al.* 1986; Brothers 1991). Cephalopods and fish make up most of the diet. At South Georgia (Islas Georgia del Sur), there was little inter-annual variation in cephalopods in the diet and 30–80% of cephalopods are scavenged (Xavier *et al.* 2003).

Older reports indicated that most ‘natural’ feeds are nocturnal (Harper 1987). More recently, individuals fitted with stomach temperature sensors took 89% of prey items during daylight hours (Weimerskirch and Wilson 1992; Weimerskirch *et al.* 1997b; Waugh and Weimerskirch 2003); however, it is not known whether some of these telemetered birds were feeding on fishery discards or natural prey items.

Depth gauges attached to wandering albatrosses indicate that individuals seize most prey on the surface and rarely submerge (Prince *et al.* 1994a).

There is also sexual and age-related segregation of foraging areas. At Crozet Island, during the breeding season, female wandering albatross foraged in subtropical waters to the north of the colony, whereas males preferred colder, higher latitude waters (Robertson *et al.* 1993). In non-breeding years, individuals also appeared to have a preferred home range, 1 500–8 500 km from Crozet, still with the same sexual segregation, females in warmer water than males (Weimerskirch and Wilson 2000). Weimerskirch *et al.* (2006) tracked 13 juvenile wandering albatrosses; they frequented subtropical waters of the Indian Ocean and Tasman Seas where wind velocity and productivity were both low—regions typically unused by adult birds.

Over two years, foraging effort appeared to be related to energy acquisition per unit effort, so that food intake levels remained stable; flight costs were the lowest recorded for any seabird (Schaffer *et al.* 2001).

Experimental manipulation suggests that wandering albatrosses do not use magnetic cues to navigate between foraging and nesting areas (Bonadonna 2005).

Global population status

About 8 000 wandering albatrosses breed each year (Table 3.1: reviewed in Gales 1998, ACAP Species Assessment). This implies that there are currently around 26-28 000 mature individuals (BirdLife International (2007), and perhaps 50-55 000 birds in total. The reliability of survey data for this species is generally good. All monitored populations have shown substantial decreases at some stage during the last 20 years (reviewed in Gales 1998). Overall the population trend is listed as decreasing (BirdLife International (2007).

Table 3.1: Breeding populations of the wandering albatross

Breeding locality	Annual no. breeding pairs	Year of census	Census reliability	Population trend
Macquarie Island[1]	4	2010	High	Stable/ Decreasing?
South Georgia (Islas Georgia del Sur)				
- Bird Island[2]	802	2007	High	Decreasing
- Other islands[2]	618	2004	High	Decreasing
Crozet Islands				
- Ile de la Possession	349	2007	High	Increasing[3]
- Ile aux Cochons	1,060	1998	High	?
- Ile de l'Est	329	1982	High	?
- Iles des Apotres	120	1982	High	?
Kerguelen Islands	1,187	2007	Moderate	Increasing[3]
Marion Island	1,730	2006	High	Decreasing[4]
Prince Edward Island	1,850	2002	Moderate	Decreasing?

[1] Data from DPIPWE (unpublished).
[2] Data from Poncet *et al.* 2006, other data from sources cited in Gales (1998) and ACAP Wandering Albatross Species Assessment (www.acaq.aq).
[3] Population is currently stable or increasing at low levels after previous population declines (Nel *et al.* 2002a), ACAP Species Assessment (www.acap.aq).
[4] Population is decreasing at 1.5% per year (ACAP Species Assessment, www.acap.aq).

Population status within areas under Australian jurisdiction

Selkirk *et al.* (1990) have suggested that the population of wandering albatrosses on Macquarie Island may have always been small compared to large populations on other similar sized islands (for example, an estimated 2 000 pairs once bred on Marion Island with an area of 290 km^2). Macquarie Island may represent marginal habitat for wandering albatrosses because the richer continental shelf around Macquarie Island is relatively small compared to that around other subantarctic islands (Selkirk *et al.* 1990).

To the contrary, there is circumstantial evidence to suggest that wandering albatrosses may have once been numerous on Macquarie Island. Seal and penguin oil harvesters occupied the island from 1810 to 1920, using the wandering albatrosses as a source of food (Cumpston 1968; Townrow 1988). In addition, the remains from more than 100 wandering albatrosses were discovered in a cave at Aurora Point during the 1949/50 ANARE field season (de la Mare and Kerry 1994). Finally, based on the known distribution of nesting sites and vegetation alliances used by wandering albatrosses over the past 50 years, and low nest densities on Macquarie Island, there appear to be large areas of suitable and/or previously used nesting habitat that are currently left vacant.

By the time the Australasian Antarctic Expedition surveyed Macquarie Island in 1913 only one wandering albatross pair was left breeding. Once harvesting ceased, the population gradually increased to 24-28 annual breeding pairs by 1967/68 (Carrick and Ingham 1970; Terauds *et al.* 2006b). However, during the 1970s the breeding population declined rapidly. By the early 1980s only five annual breeding pairs remained (de la Mare and Kerry 1994) and only two in 1985 (Terauds *et al.* 2006b).

The latest population estimate is four annual breeding pairs in 2010, and the maximum in recent years has been a total of 15 breeding pairs in the mid 1990s (Terauds *et al.* 2006b, DPIPWE unpublished data). Thus, the Macquarie Island population of wandering albatrosses is the smallest in the world. Since it contains less than 50 mature individuals it can be considered *Critically Endangered* according to IUCN (1996) criteria. That is, this population is "a taxon that is facing a very high risk of extinction in the wild in the immediate future" (Baillie and Groombridge 1996).

Johnstone (1982) first recorded one pair of wandering albatross brooding a small chick on Heard Island in 1980. The male had been banded as a non-breeding adult on Macquarie Island in 1967. The female was not seen. Johnstone (1982) also noted the presence of two old nest mounds nearby, suggesting breeding had been attempted in previous years as well.

Population status outside areas under Australian jurisdiction

The populations of wandering albatrosses breeding on the Crozet Islands, Kerguelen Island and Prince Edward Island had all been severely reduced by the turn of the 20^{th} Century via exploitation from sealers and whalers (Croxall *et al.* 1984a). The Marion Island population had been decreasing at an average annual rate of 0.7% until 1992, when the population began to increase (J. Cooper pers. comm., in Gales 1998). Likewise, the Prince Edward Island population has also suffered declines (Watkins 1987). The wandering albatross population at Possession Island (Crozet Islands) has declined by more than 50% over the last 20 years but has increased steadily since 1986. The population at Kerguelen Island decreased by around 60% during the 1970s and early 1980s but has also stabilised since 1986 (Weimerskirch and Jouventin

1998). Between 1961 and 1996, wandering albatross populations at South Georgia (Islas Georgia del Sur) have decreased by 30% since 1984 at an annual average rate of 1.8% (Poncet *et al.* 2006). The accelerating rate of declines at South Georgia (Islas Georgia del Sur) significantly threatens the long term viability of wandering albatrosses at this site.

3.1.2 Black-browed Albatross *Thalassarche melanophris* Temminick 1828

Previous name

Black-browed albatross *Diomedea melanophris*

Jurisdiction	Breeding locality
Australia	**Heard Island, McDonald Island, Macquarie Island, Bishop and Clerk Islets**
Chile	Islas Diego Ramírez, Isla Ildefonso, Isla Diego de Almagro, Islote Evangelistas, Islote Albatros
France	Crozet Islands, Kerguelen Islands
New Zealand	Antipodes Islands, Campbell Island, Snares Island
Other	Falkland Islands (Islas Malvinas), South Georgia (Islas Georgia del Sur)

Recent research by Alderman *et al.* (2005) suggests that Macquarie Island black-browed albatross populations belong to a genetic grouping that includes the Chilean, South Georgia (Islas Georgia del Sur) and Kerguelen populations.

Distribution

The black-browed albatross is probably the most widespread of all albatrosses. It has a circumpolar distribution in the southern oceans, occurring from the Antarctic pack-ice to the equator, but principally between 40-70°S (Marchant and Higgins 1990; Tickell 1995). There are records of vagrants in the North Atlantic, north to Greenland, Iceland and Norway (Shirihai 2002).

From August to April most adults occur in the Antarctic and subantarctic shelf-waters adjacent to their breeding grounds. However, they are migratory and in April they leave their colonies for the warmer coastal or shelf waters of Australia, New Zealand, South Africa and South America (Weimerskirch *et al.* 1985, 1986).

The over-wintering areas for the various colonies are thought to be distinct. South Georgia (Islas Georgia del Sur) colonies winter off Australia, New Zealand or the west coast of South Africa. Falkland Island (Islas Malvinas) birds winter off the east coast of South America. Finally, Kerguelen Island birds winter off southern Australia (Croxall *et al.* 1998; Prince *et al.* 1998; Weimerskirch 1998). However, the few bands returned from the Macquarie Island and Heard Island colonies are also from southern Australia (Milledge 1977), indicating that the segregation at sea is not complete. Indeed, satellite tracking of birds during the incubation period from Chile and the Falkland Islands (Islas Malvinas) showed broadly overlapping foraging areas (BirdLife International 2004b).

In Australia, black-browed albatrosses forage along the southern coasts (sometimes entering bays and harbours) from Brisbane around to Perth. However, they are less common north of Sydney (Blakers *et al.* 1984; Marchant and Higgins 1990; Reid *et al.* 2002). Sub-adults are observed in Australian waters all year round. Consequently, 99% of black-browed albatrosses seen in south-eastern Australian waters between October and January are immature birds (Reid *et al.* 2002).

Adult black-browed albatrosses usually obtain food for the chick by commuting rapidly and directly to the continental shelf breaks or frontal zones adjacent to their colonies. Individuals often revisit the same areas on successive foraging sorties, signifying the predictability of their prey (Weimerskirch *et al.* 1986; Cherel and Weimerskirch 1995; Prince *et al.* 1998). The overall pattern of foraging by Falkland Islands (Islas Malvinas) birds accords with recent studies of the species at South Georgia (Islas Georgia del Sur) and Kerguelen Islands and confirms this species' preference for foraging over shelf areas rather than over deeper waters as at Campbell Island (Huin 2002). Thus, at the Crozets, black-browed albatrosses forage mainly within 40 km of the islands, while at other colonies they will search for prey more than 400 km away (Croxall and Prince 1987; Weimerskirch *et al.* 1988). However, at Macquarie Island, where the continental shelf is particularly small, satellite-tracked adults have been recorded foraging for the chick in Antarctic seas south of 60°S over 1 200 km away (Terauds *et al.* 2006a). Compared to incubation, foraging areas typically contract during the chick-rearing period (BirdLife International 2004b). At Macquarie Island during late incubation and chick rearing over 90% of black-browed albatross foraging time was contained within the Exclusive Economic Zone around Macquarie Island (Terauds *et al.* 2006a).

Intraspecific differences are also evident, at least in the Falklands (Islas Malvinas), where breeding birds from the northern islands foraged in a different part of the Patagonian Shelf to birds from the southern islands (Huin 2002). Phillips *et al.* (2004) detected sexual segregation during incubation but not during brooding when birds foraged closer to the colonies.

Breeding biology

Black-browed albatrosses breed annually. Breeding begins between late August to mid-October depending on location, with the colonies south of the Antarctic convergence initiating breeding slightly later than their northern counterparts (Kirkwood and Mitchell 1992). Adults begin returning to Macquarie Island in late August (Copson 1988; Terauds *et al.* 2005) and to Heard Island before September 18 (Downes *et al.* 1959). Males normally begin arriving one to two weeks before the females (Tickell and Pinder 1975).

Breeding is normally colonial with nests 1–2 m apart. The same nest is normally (94%) used for several years (Tickell and Pinder 1975). Egg-laying is from late September and through October at Macquarie Island (Terauds *et al.* 2005), and from October 20 at Heard Island (Downes *et al.* 1959). Incubation lasts for 65-72 days, and the hatchling is brooded for the first three weeks. The adults feed the chick almost every day until it fledges in April-May at about four months of age.

At Macquarie Island mean breeding success (measured as the total number of chicks fledged from eggs laid) from 1994–2003 averaged 48% (Terauds *et al.* 2005). At Heard Island, breeding success (the number of chicks raised to at least five weeks as a percentage of total breeding pairs) was 17% in 1954/55, and 68% in 1987/88 (Downes *et al.* 1959; Kirkwood and Mitchell 1992).

Young fledge between mid-April and mid-May at Macquarie Island (Copson 1988; Terauds *et al.* 2005) and around mid-April at Heard Island (Downes *et al.* 1959). Fledging at other sites also occurs in April or May (Tickell 1966; Tickell and Pinder 1975).

Black-browed albatrosses display extremely high levels of philopatry (Copson 1988; Prince *et al.* 1994b). A detailed capture-recapture study at South Georgia (Islas Georgia del Sur) found no evidence of breeding birds moving among colonies (Prince *et al.* 1994b). Most immatures begin returning to their natal colony at 3–8 years of age. At Macquarie Island, birds do not commence breeding until seven or eight years of age (Copson 1988). At other sites, the age at first breeding varies from 6–13 years (Jouventin and Weimerskirch 1988; Prince *et al.* 1994b). There is high pair fidelity and, at least on South Georgia (Islas Georgia del Sur) in 1998–1989, low levels of extra-pair paternity (0–9% of chicks; Burg and Croxall 2006).

Foraging ecology

Black-browed albatrosses have been observed taking most prey (98%; n = 232) by surface-seizing, with limited surface-plunging (2%: Harper 1987). Harper (1987) also noted that some individuals were capable of remaining submerged for almost 20 seconds in pursuit of prey. Prince *et al.* (1994a) attached capillary gauges to 21 black-browed albatrosses to record their maximum dive-depths on foraging trips. Individuals dived to a mean maximum depth of 2.5 m and an overall maximum depth of 4.5 m. All individuals dived to more than 1m, indicating that diving is a more common mode of capturing prey than previously realised. In terms of how black-browed albatrosses navigate, work by Bonadonna *et al.* (2003) concluded that geomagnetic navigation was not significant.

In the Kerguelen Islands, dietary differences suggest interspecific segregation of foraging areas, at least (Cherel *et al.* 2002). Sympatric, chick rearing black-browed, grey-headed, yellow-nosed albatrosses took similar sized prey, but black-browed albatross fed on cephalopods, fish and penguins in roughly equal proportions; grey-headed albatross fed more on squid; and yellow-nosed albatrosses fed more on fish and did not take penguins (Cherel *et al.* 2002).

The distribution of the prey species indicates that black-browed albatrosses breeding in Chile obtained the bulk of their food over the South American continental shelf, but also foraged at the Antarctic Polar Front. The prevalence in the diet of fish species discarded from fishing operations, and the presence of fish hooks and fish bait species, indicate a strong association with fisheries in southern Chile (Arata and Xavier 2003).

Global population status

Black-browed albatrosses are the most widely distributed of all albatross species and their population status varies with respect to colony. The current population is approximately 530 000 breeding pairs (BirdLife Factsheets 2007).

Table 3.2: Breeding populations of the black-browed albatross

Breeding locality	Annual no. breeding pairs	Year of census	Census reliability	Population trend
Macquarie Island [1]	47	2010	High	Stable
Bishop and Clerk Islets	141	1993	Moderate	?
Heard Island[2]	600	2000/01	Moderate	Stable?
McDonald Island	82–89	1981	Moderate	?
Falkland Islands (Islas Malvinas) [3]	318 000			Decreasing
South Georgia (Islas Georgia del Sur) [4]				
- Bird Island	8 264	2003/04	High	Decreasing
- Other islands	66 032	2003/04	High	Decreasing
Chile				
- Diego Ramírez (total)[5,]	55 000	2002	High	?
- Isla Ildefonso[5, 6]	47 000	2002	High	?
- Isla Diego de Almagro[7]	15 594	2001	High	Stable?
- Islote Evangelistas[8]	4 670	2002	?	?
- Islote Albatros[6]	62	2006	High	?
Crozet Islands	980	1981	High	?
Kerguelen Islands	3 115	1995	High	Decreasing
New Zealand				
- Antipodes Islands	~ 100	1992	Low	?
- Campbell Island	> 30	1995	Low	?
- Snares Island	1	1986	Moderate	?

Data from sources cited in Gales (1998) except: [1]DPIPWE (unpublished); [2] Woehler *et al.* 2002; [3]BirdLife Factsheets 2007, [4]Poncet *et al.* 2006; [5]Robertson *et al.* 2003b: [6] Robertson *et al.* 2007; [7] Lawton *et al.* 2003; [8] Arata *et al.* 2003.

Population status within areas under Australian jurisdiction

The world's smallest and most vulnerable populations of black-browed albatross occur on the Australian and New Zealand subantarctic islands.

The present status of the Macquarie Island population is difficult to assess because its history is poorly known. The records for black-browed albatrosses breeding on

Macquarie Island are incomplete prior to 1985. Black-browed albatrosses were first recorded breeding there in small colonies in 1949/50 (Copson 1988; Selkirk *et al.* 1990). It is difficult to determine if the present colonies are the tiny remnants of larger populations decimated by sealers and oil-gatherers by the early 1900s, or have re-colonised after being eradicated from the island. Alternatively, the population may have always been small, constrained by the limited continental shelf surrounding the island (Copson 1988).

Of the 120 chicks banded on Macquarie Island during the 1970s and 1980s, only four have ever returned to breed, equating to a minimum recruitment rate of only 3.3% (Copson 1988). Currently, about 40 pairs breed on Macquarie Island each year (Table 3.3: Terauds *et al.* 2005). The small colony at 'North Tussocks' on Macquarie Island began declining in numbers in the 1950s (Copson 1988), and has now disappeared entirely. No chicks have fledged there since the 1970s. The small size of the population at Macquarie Island combined with the apparent extremely low rates of recruitment is cause for grave and urgent concern for the viability of the population in the future.

In 1965 a population of black-browed albatrosses was discovered breeding on Bishop and Clerk Islets, 37 km to the south of Macquarie Island. The population was assessed in 1993 when 141 active nests were found (Gales 1998). There is no information on the current status of this population.

The situation is less bleak at Heard Island where in 2000/01 there were about 600 breeding pairs, as there were in 1987/88 (Kirkwood and Mitchell 1992; Woehler *et al.* 2002). Woehler *et al.* (2002) concluded that since the first census in 1947/48 the population at Heard Island had increased at all four known breeding localities, from a total of 200 pairs to 600 pairs in 2000/2001, mostly likely due to climate amelioration and discards from trawlers. This interpretation however, that relies upon comparison of few data separated by many decades, should be treated with caution, noting there is 100% scientific observer coverage and a strict nil discharge policy on fishers operating within the Australian EEZ around Heard and McDonald Islands.

The status of the McDonald Islands populations is also unclear due mostly to the sporadic nature of the data collection. The most recent estimate for McDonald Island is from 1981 when 82–89 pairs were breeding (Keage and Johnstone 1983). However, since then, there has been significant volcanic activity on the island that would probably have caused birds to relocate, even if only temporarily.

Population status outside areas under Australian jurisdiction

Black-browed albatross populations are known to be in decline at Bird Island and the Falkland Islands (Islas Malvinas), with previous declines also documented for the Kerguelen Island population Croxall *et al.* 1998; Poncet *et al.* 2006; Weimerskirch and Jouventin 1998).

The Falkland Islands (Islas Malvinas) have the largest population, and steep population decreases have been recorded at that site over the last two decades (BirdLife Factsheet 2007). Chile holds the second largest black-browed albatross population in the world, comprising about 20% of numbers, but there is no reliable information on population trends at these breeding sites (Robertson *et al.* 2007). At South Georgia (Islas Georgia del Sur), by 1996, the population at Bird Island had decreased by 31% since 1976, at an average rate of 1.8% per annum. Most (or all) of this decline appears to have occurred between 1989 and 1996 at an average rate of 6.9% per annum (Croxall *et al.* 1998). More recent surveys show that the population has decreased at a rate of 4.0% pa between 1989 and 2003, a loss equivalent to 44% of the population (Poncet *et al.* 2006). The Kerguelen population decreased in size by 30% between 1978–88, and seems to have stabilised somewhat since then, with an overall rate of change of -0.2% pa between 1979 and 1995 (Jouventin and Weimerskirch 1991; Weimerskirch and Jouventin 1998). Given the decreases being recorded at several important sites for this species it is important that regular monitoring be continued or implemented where it is lacking.

3.1.3 Shy Albatross *Thalassarche cauta* Gould 1841

Previous names

Shy albatross *Diomedea cauta cauta*

Jurisdiction	Breeding locality
Australia	**Albatross Island, The Mewstone, Pedra Branca**

The only albatross species endemic to Australia.

Originally a member of the polytypic species *Diomedea cauta* (Gould 1841), *T. cauta* was elevated to specific status when *Diomedea cauta* was placed in the genus *Thalassarche* and split into four species: *T. cauta* (shy albatross), *T. steadi* (white-capped albatross), *T. eremita* (Chatham albatross) and *T. salvini* (Salvin's albatross) (Robertson and Nunn 1998, Abbott and Double 2003 a, b). The recognition of *T. cauta* and *T. steadi* remains controversial (Brooke 2004) although following scrutiny of morphological, genetic and behavioural data by the ACAP Taxonomy Working Group, BirdLife International has endorsed recognition of *T. cauta* and *T. steadi* as separate species.

Distribution

The recent separation of the shy albatrosses from other closely related taxa confounds our understanding of its at-sea distribution. Band recoveries, satellite-tracking data, and genetic identification of birds caught in fishing operations show that shy albatrosses are most frequently found around Tasmania and southern Australia (Brothers *et al.* 1997; Hedd *et al.* 2001; Abbott *et al.* 2006) but its range also extends to southern Africa (Barton 1979; Blakers *et al.* 1984; Tickell 1995; Reid *et al.* 2002; BirdLife International 2004b; Abbott *et al.* 2006; ACAP 2006).

Sighting data show that shy albatrosses are less pelagic than many other albatross species, are usually found over the continental shelf, and regularly venture close to shore along the coasts of Tasmania and southern Australia, even entering bays and harbours (Brothers *et al.* 1998; Hedd *et al.* 2001; Reid *et al.* 2002).

Satellite telemetry has recently been used to determine the foraging areas of breeding shy albatrosses from Albatross Island and Pedra Branca (Brothers *et al.* 1998; Hedd *et al.* 2001; Hedd and Gales 2005). During breeding, adults forage close to their colonies, usually within 300 kms, in waters less than 200m deep (Hedd *et al.* 2001). During incubation birds from Albatross Island foraged off north-west Tasmania in an area encompassing 27 700 km^2 of ocean. Incubating birds from Pedra Branca tended to forage over a smaller area (9 500 km^2) towards the east or south-east edge of the continental shelf. Shy albatrosses fed exclusively in neritic waters close to their colonies (Brothers *et al.* 1998; Hedd and Gales 2005). The maximum foraging range of any breeding bird was 200–265 km from its colony (Brothers *et al.* 1998; Hedd *et al.* 2001). 72% of flying was in the daytime and 28% at night, the later particularly during full moon (Hedd *et al.* 2001).

The broad routes of post-fledging dispersal appear to be colony specific. Young birds from Albatross Island have been found only as far west as south-west Western Australia and east to Queensland. In contrast, juveniles and immatures from the Mewstone have been recovered off both South Africa and New Zealand (Brothers *et al.* 1997). None of the immature birds banded at Pedra Branca have ever been recovered away from the colony (Brothers *et al.* 1997).

Breeding biology

Shy albatrosses nest in colonies and have an annual breeding cycle lasting about eight months, from September until April. Mean nest densities are 1–2 nests per m^2 (Brothers 1979a).

Most eggs are laid in September or early October. Breeding is asynchronous among colonies, with the mean egg-laying date at Pedra Branca (and probably the Mewstone) being about 1–2 weeks later than on Albatross Island (N. Brothers pers comm). The egg is incubated for about ten weeks. The chick hatches in December and is brooded for a further three weeks (Johnstone *et al.* 1975; DPIPWE unpublished information). Chicks fledge in April at about 4.5 months old. Immature birds begin returning to the natal colony after at least three years at sea. After a minimum of 5–6 years, most shy albatrosses have paired and begin breeding annually (N. Brothers unpubl. data). Adult birds frequent the colonies for ten months of the year, between July and April (Hedd and Gales 2005).

Breeding success at Albatross Island averages 37% (+/- 7%) (ACAP 2006). In some years an avian pox virus contributes to high levels of chick mortality (Woods 2004). Between 1981 and 2003 chick production increased from approximately 1000 to 3000 chicks per year. Since 2003 the chick production has consistently decreased with fewer than 1800 chicks fledged in 2006. Similarly, at Pedra Branca the number of chicks has declined from a high of about 150 in 1998/9 to 31 in 2006/7 (ACAP 2006: DPIW 2007).

Observations and genetic paternity analysis showed that most copulations were within pairs but that some females solicited extra-pair mating and only 7–10% of chicks were extra-pair (Abbott *et al.* 2006).

Foraging ecology

Most observations of shy albatrosses feeding at sea have been of birds seizing dead or moribund prey at the surface, taking fish from surface schools while flying, or occasionally making shallow dives or surface plunges (Barton 1979; Harper *et al.* 1985; Croxall and Prince 1994). However, Hedd and co-workers (1998) used time-depth recorders and maximum depth gauges attached to adult shy albatrosses to demonstrate that this species routinely penetrates the water surface to take prey. The majority of plunge-dives were to within 3 m of the surface, lasting less than six seconds. Shy albatrosses also actively swam underwater for up to 19 seconds to a depth of 7.4 m. Nine of the 15 birds monitored in the study dived below 5 m indicating that it is a standard foraging strategy used by this species. Diving only occurred between 07:00 and 22:00 hours. The deepest dives occurred between 10:00–12:00 hours (Hedd *et al.* 1998).

The diet of shy albatrosses at Albatross Island has only been examined through examination of food delivered to chicks on Albatross Island (Hedd and Gales 2001). Between 1995 and 1998, the food samples delivered by parents were mostly fish (89% by wet mass) and cephalopods (10% by wet mass), with small amounts of tunicates and crustaceans (Hedd and Gales 2001). Prey selection appeared to be relatively constant across seasons and years. Most (80%) of the fish delivered by adults to chicks were pelagic schooling Jack Mackerel *Trachurus declivus* and Redbait *Emmelichthys nitidus*, while 84% of the cephalopods were Gould's Squid *Nototodarus gouldi*. Thus, there is considerable evidence to indicate that shy albatrosses capture most of their prey live during the day, from on or just below the surface (Hedd and Gales 2001).

Shy albatrosses usually forage singly or in flocks of up to 20 birds (Barton 1979). They will also aggregate behind fishing vessels into flocks of over 100 birds (T. Reid pers. comm.) where they are usually able to out-compete all smaller Procellariiformes (i.e. all but the wandering, Tristan, Antipodean, and royal albatrosses: Brothers 1991).

Global population status

The shy albatross is the only albatross species endemic to Australia. The total breeding population is currently around 15 000 breeding pairs (Table 3.4). Gales (1998) estimated that approximately 55 000–60 000 individuals currently exist.

Table 3.4: Breeding populations of shy albatrosses

Breeding locality	Annual no. breeding pairs	Year of census	Census reliability	Population trend
Albatross Island	5233	2009	High	Stable
The Mewstone	9000-11000	2005	Moderate	?
Pedra Branca	220	2007	Moderate	Declining

Data from DPIPWE (2009).
? Population trend is unknown due to a lack of recent or consistent population censuses

Population status within areas under Australian jurisdiction

The first European sighting of the shy albatross colony on Albatross Island was by George Bass in 1798, when about 20 000 breeding pairs were thought to have nested on the island annually. By 1909, however, plume and egg hunters had decimated the colony to only 250–300 nests (Green 1974; Johnstone *et al.* 1975). Population surveys taken since then indicate that at Albatross Island the population has staged a recovery, but appears to have stabilised at only about 25% of the estimated original population size. The current breeding effort level of ca 5 000 pairs has been stable since 1995.

The colonial histories of the Mewstone and Pedra Branca have not been well documented and it is not clear whether these are separate populations or part of the same population. Due to the difficulties of surveying nesting seabirds on these islands, early estimates are perhaps unreliable, and hence the status of these populations remains unknown (Tables 3.6, 3.7). The Pedra Branca population is critically low and may have always been very small; however, productivity is in decline at approximately 10% pa (ACAP 2006). Competition for nesting space from Australasian gannets (*Morus serrator*) is likely contributing to the steadily declining productivity of shy albatrosses from Pedra Branca each year.

There are no published estimates of either adult or juvenile survival for this species.

Population status outside areas under Australian jurisdiction

Shy albatrosses do not breed outside of the AFZ; however, they do disperse to areas outside of the AFZ.

3.1.4 Grey-headed Albatross *Thalassarche chrysostoma* Forster 1785

Previous name

Grey-headed albatross *Diomedea chrysostoma*

Jurisdiction	Breeding locality
Australia	**Macquarie Island**
Chile	Islas Diego Ramírez, Isla Ildefonso
France	Crozet Islands, Kerguelen Islands
New Zealand	Campbell Island
South Africa	Marion Island, Prince Edward Island
Other	South Georgia (Islas Georgia del Sur)

Distribution

The grey-headed albatross is a bird of the open oceans that occupies a circumpolar pelagic range. During the breeding period adults travel hundreds or thousands of kilometres from the colony (generally to waters within or south of the Antarctic Polar Frontal Zone) in order to obtain food for their offspring (Weimerskirch *et al.* 1988; Prince *et al.* 1998).

Breeding adults travel enormous distances in search of prey for the chick. The maximum foraging ranges of grey-headed albatrosses breeding on Bird Island (South Georgia (Islas Georgia del Sur)) have been recorded to be 500–800 km (Prince and Francis 1984; Rodhouse *et al.* 1990). At Prince Edward Island grey-headed albatrosses have been observed foraging 350 km from the nest (Hunter and Klages 1989), while at the Crozet-Kerguelen area they have been seen foraging some 1 850 km from their nest (Weimerskirch *et al.* 1986, 1988). Body measurement differences between colonies have been attributed, at least in part, to foraging distances and the length of the nestling period (Waugh *et al.* 1999c).

Grey-headed albatrosses breeding at Macquarie Island typically foraged in waters south of the island, frequently travelling though CCAMLR waters. The Marine Protected Areas surrounding the island therefore affords little protection to this species and it remains at risk from legal and illegal fishing operations on the high seas (Terauds *et al.* 2006a). Grey-headed albatrosses breeding at South Georgia (Islas Georgia del Sur) disperse widely into oceanic waters south of the colonies and whilst key foraging areas are evident, there is considerable inter-annual variability in foraging site selection (BirdLife International 2004b).

Non-breeding adults and immature birds disperse widely over the Southern Ocean, mostly between 65°S and 35°S (del Hoyo *et al.* 1992). In summer, they are found in subantarctic and Antarctic seas between 46°S and 64°S, avoiding pack ice. Most leave the Antarctic Zone in winter for the warmer seas between 39°S and 51°S. Some also follow the Humboldt Current north to 15°S in western South America (Marchant and Higgins 1990). Non breeding grey-headed albatrosses from South Georgia (Islas Georgia del Sur) have a circumpolar winter distribution (BirdLife International 2004b) and three distinct foraging strategies have been identified, including one or more global circumnavigations, the fastest in just 46 days (Croxall *et al.* 2005).

The grey-headed albatross is a regular visitor to Australia and New Zealand, especially in winter. It is seen at sea with some frequency south and west of

Tasmania, occasionally in Victorian waters, rarely in South Australia and Western Australia, and only as a very rare vagrant in New South Wales. It has only been recorded once in Queensland (Blakers *et al.* 1984; Reid *et al.* 2002).

Breeding biology

Typically, grey-headed albatrosses are biennial breeders. However, this depends somewhat on the breeding success of the previous year, as failed breeders tend to re-nest in the following year (Hector *et al.* 1986; Prince *et al.* 1994b).

Adults return to the breeding grounds from early September to early October. Older, more experienced birds often return before younger breeders (Tickell and Pinder 1975; Weimerskirch *et al.* 1986). At Macquarie Island grey-headed albatrosses return after September 12 (Copson 1988), signalling the beginning of a breeding season that lasts for 10–11 months. Pairs build their nests about 1–2 m apart in dispersed colonies. The egg is laid during October. Both parents share the circa 72-day incubation period, in shifts averaging 5–15 days. Hatchlings emerge from December 12 to January 19 to be brooded almost constantly for 18–28 days (Tickell and Pinder 1975; Prince *et al.* 1994b).

The offspring achieve independence after 140–152 days, fledging between late April and mid-June, depending on breeding locality (Tickell and Pinder 1975; Prince *et al.* 1994b). At Macquarie Island all juveniles and adults have departed the breeding grounds by late May (Copson 1988). Between 1995 and 2005 breeding success of grey-headed albatrosses averaged 59% (DPIPWE data), which is slightly higher than other well documented colonies (e.g. South Georgia (Islas Georgia del Sur) mean = 39%; range = 5-60%: Prince *et al.* 1994b; Croxall *et al.* 1998).

Grey-headed albatrosses banded as chicks at Macquarie Island began to breed after 7–10 years (Copson 1988). The modal age at first breeding is 12 years at South Georgia (Islas Georgia del Sur) (*n* = 52: Prince *et al.* 1994b). Once established the bird will breed for many years. No breeding bird has ever been observed to move between colonies (Copson 1988; Prince *et al.* 1994b). Pair fidelity is high although there is evidence of some extra-pair paternity (3–10% of chicks, South Georgia (Islas Georgia del Sur), 1998–1989; Burg and Croxall 2006).

Foraging ecology

Most prey is taken by surface-seizing (Wood 1992). Prince *et al.* (1994a) also discovered that grey-headed albatrosses can dive to at least 6 m below the surface, and swim underwater for up to 11 seconds, in search of prey.

Phillips *et al.* (2004) described sexual segregation during incubation, with largely mutually exclusive core foraging ranges for each sex of grey-headed albatrosses. In the Kerguelen Islands, dietary differences also suggest interspecific segregation of foraging areas (Cherel *et al.* 2002). Chick rearing black-browed, grey-headed and Indian yellow-nosed albatrosses took similar sized prey, although grey-headed albatrosses fed more on squid (Cherel *et al.* 2002). Similarly, at Campbell Island grey-headed albatrosses were oceanic foragers for cephalopods (Waugh *et al.* 2004).

Global population status

The global breeding population of grey-headed albatrosses is estimated to be 92 300 pairs per year (BirdLife Factsheets 2007). This corresponds roughly to 250 000 mature individuals, or 600 000 individuals in total (reviewed in Gales 1998). Globally, this species is classified as *Vulnerable* because of an estimated overall decline of c.48% over three generations (90 years), probably largely owing to mortality on longline fisheries. If the major declines observed at some sites are shown to be also occurring elsewhere, the species would warrant uplisting to Endangered (BirdLife International 2007).

Table 3.8: Breeding populations of the grey-headed albatross

Breeding locality	Annual no. breeding pairs	Year of census	Census reliability	Population trend
Macquarie Island [1]	108	2010	High	Stable
South Georgia (Islas Georgia del Sur)[2]				
- Bird Island	5,120	2003/04	High	Decreasing
- Other islands	42,554	2003/04	High	Decreasing
Chile				
- Diego Ramírez (total)[3]	17,000	2002	High	?
- Islas Gonzalo[3]	6,155	2002	High	?
- Isla Ildefonso[3]	~ 8	2002	High	Stable?
Kerguelen Islands	7,900	1984–87	?	?
Crozet Islands	5,946	1980–82	?	?
Marion Island	6,217	1995	High	Stab le
Prince Edward Island	1,500	1979	Low	?
Campbell Island	~ 6,400	1995	Moderate	Decreasing[4]

? Population trend is unknown due to a lack of recent or consistent population censuses
Data from Gales (1998) except: [1] data from DPIPWE; [2] data from Poncet *et al.* 2006; [3] Data from Robertson *et al.* 2003b, 2007; [4] Moore 1999.

Population status within areas under Australian jurisdiction

Currently, between 65-115 pairs of grey-headed albatrosses breed on Macquarie Island each year and the population is stable at these low levels (Terauds *et al.* 2005; DPIPWE data). The records for grey-headed albatrosses on Macquarie Island prior to 1985 are incomplete and based upon casual estimates only. However, the population appears to have increased since the early 1900s when roughly 40 nests were found on the island (Falla 1937).

Population status outside areas under Australian jurisdiction

The recruitment rate of immature birds at Bird Island (South Georgia (Islas Georgia del Sur)) has declined drastically from 35% to 5% over the last two decades (Croxall *et al.*1998). Adult survival has also decreased, from 95% to 93%. As a result, the population has declined. Comparison of survey data from 1985 and 2003 shows a reduction of 18.7% (1.1% pa). Notwithstanding this, the South Georgia (Islas Georgia del Sur) population is still the most important site globally for the species (Poncet *et al.* 2006).

The Chilean grey-headed albatross population (Diego Ramirez Archipelago) is much larger than previously estimated. After South Georgia (Islas Georgia del Sur), Chile has the world's second largest population of grey-headed albatrosses, comprising about 23% of the annual breeding population for the species (Robertson *et al.* 2007). A lack of temporal population data makes trend analyses not possible for this population.

Over the last fifty years albatross colonies at Campbell Island (in which grey-headed albatrosses predominate) have decreased by 79–85% (Moore 1995), averaging a loss of 3–4.8% of numbers annually at different colonies since the 1940s (Waugh *et al.* 1999a).

The only population increase recorded for this species has been at Marion Island since 1992, although this population now appears stable (Nel *et al.* 2002b). This population, which represents 7% of the global population, had previously been decreasing at 0.7% per annum since the 1970s (J. Cooper pers. comm., in Gales 1998).

3.1.5 Light-mantled Albatross *Phoebetria palpebrata* Forster 1785

Previous name

Light-mantled Sooty albatross *Phoebetria palpebrata*

Jurisdiction	Breeding locality
Australia	**Heard Island, Macquarie Island, McDonald Islands?**
France	Crozet Island, Kerguelen Island
New Zealand	Antipodes Island, Auckland Island, Campbell Island
South Africa	Marion Island, Prince Edward Island
Other	South Georgia (Islas Georgia del Sur)

Distribution

Light-mantled albatrosses have a wide, circumpolar range throughout the Southern Ocean. They are highly dispersive over pelagic waters, and have the most southerly distribution of any albatross, ranging the temperate waters south of 35°S to the pack ice around 78°S. Many traverse northwards with the Humboldt Current along the coast of Chile and Peru to 20°S (Marchant and Higgins 1990).

Light-mantled albatrosses are regular visitors to the pelagic waters of south and south-east Australia, especially in winter. They are commonly seen over open waters south and west of Tasmania. Many of the birds seen in mainland waters are breeding adults foraging on behalf of their offspring (Marchant and Higgins 1990; Reid *et al.* 2002).

Breeding adults forage great distances whilst raising chicks. Light-mantled albatrosses breeding at Macquarie Island were found to forage up to 2 200 km from their nest (Weimerskirch and Robertson 1994). Similarly large foraging ranges have been found at other sites where light-mantled albatrosses commute rapidly to specific areas in southern waters, not making extensive use of well defined frontal systems (Weimerskirch and Robertson 1994; Akkers 2002; Phillips *et al.* 2005). Very little is known about the foraging strategies of light-mantled albatrosses outside the breeding season.

Breeding biology

Light-mantled albatrosses nest solitarily or in loose colonies on steep cliffs (Taylor 2000). Breeding is biennial or triennial, with 75% of successful pairs returning to breed every third year. Even unsuccessful pairs usually (60%) breed only after two years (Jouventin and Weimerskirch 1988).

Adults return to their breeding grounds in early September to mid-October (Weimerskirch *et al.* 1986; Croxall and Prince 1987), though a few arrive at Macquarie Island and Heard Island before. Unlike most albatrosses, the light-mantled albatross typically breeds as dispersed pairs or otherwise in small colonies to a maximum of 15 pairs.

At Macquarie Island, eggs are laid in October and November, hatch in December and January with most chicks fledging in May and June at about 140 days of age. Immature light-mantled albatrosses are extremely philopatric, and after being at sea for 7–12 years, return to their natal breeding grounds as adults (Weimerskirch *et al.* 1987). One bird banded as a chick at Macquarie Island has been observed breeding as a seven year old (Kerry and Garland 1984).

Between 1994 and 2004 mean breeding success at Macquarie Island was 51% (range = 42–62%: DPIPWE data). Mean breeding success at Possession Island is 35%, ranging from almost 0% to 78% (Weimerskirch and Jouventin 1998). Based on work at the Crozet Islands, light-mantled albatrosses fledge a chick, on average, every five years. Consequently, this species has one of the lowest reproduction rates for any species of albatross (Weimerskirch *et al.* 1987).

Foraging ecology

Using a combination of satellite and archival loggers on four albatross species, light-mantled albatrosses exhibited the shortest wet bouts at night, and spent the least amount of time on the water by night, suggesting that they may be the most aerial and nocturnally active of the four species (Phelan *et al.* 2007). They frequently scavenge, but are prone to direct competition, usually taking off quickly after seizing food (Harper 1987). This supports observations that albatrosses forage most actively during daylight, even though many of their fish and squid prey approach the surface only at night.

Light-mantled albatrosses are known to plunge to a mean maximum depth of almost 5 m, and some individuals dive to more than 12 m below the surface in pursuit of prey (Prince et al. 1994a).

At Macquarie Island, an automatic nest weighing system showed that chicks were fed on average every 1.6 days with a mean meal size of 520 g, the peak chick mass being 4.4 kg. Macquarie Island chicks were fed more frequently than conspecifics at South Georgia (Islas Georgia del Sur) (Phillips *et al.* 2005; Terauds and Gales 2006).

Global population status

There are an estimated 19 000 - 24 000 pairs of light-mantled albatrosses breeding each year (BirdLife International 2007). Pairs breed on 14 separate islands, but accurate population estimates are available for only two localities, Possession Island and Macquarie Island, each of which comprises about 5% of the estimated global population. Few colonies have been surveyed in the last ten years. At least five of the islands have less than 200 annual breeding pairs, and only three contain more than 3 000 breeding pairs (Table 3.10: reviewed in Gales 1998).

Table 3.10: Breeding populations of light-mantled albatrosses

Breeding locality	Annual no. breeding pairs	Year of census	Census reliability	Population trend
Macquarie Island	1281[1]	2005	High	Stable
Heard Island	200–500	1954	Low	?
McDonald Islands	Suspected	?	Low	?
South Georgia (Islas Georgia del Sur)	5 000–7 500	?	?	?
Prince Edward Island	40	1983–90	Low	?
Marion Island	201	1987	Moderate	?
Kerguelen Islands	3 000–5 000	1 984–87	?	?
Crozet Islands				
- Ile de la Possession	996	1 995	Low	Decreasing
- Ile de l'Est	> 900	1 981–95	?	?
- Ile aux Cochons	50–100	1 981–82	?	?
- Ile des Pingouins	30	1 981–82	?	?
- Ile des Apotres	150	1981–82	?	?
New Zealand				
- Auckland Islands	~ 5 000	1 972–73	Low	?
- Campbell Island	> 1 500	1 995	Low	?

- Antipodes Islands	< 1 000	1 969	Low	?

Data from sources cited in Gales (1998) except Macquarie Island population (DPIPWE data)
? Population trend is unknown due to a lack of recent or consistent population censuses
[1] Annual monitoring continues at 9 breeding sites. Light mantled albatross breed at intervals ranging from 2 to 4 years and not all sites are accessible; total breeding population probably ranges from around 1,000 to 2,000 breeding pairs. Population trends have been determined from the monitored sites, which have been largely stable.

Population status within areas under Australian jurisdiction

The original size of the light-mantled albatross populations breeding at Macquarie Island and Heard Island are unknown. Sealers occupied both islands during the 19th Century. It is likely that light-mantled albatrosses were exploited during this period of occupation, however the extent of this mortality is not known.

In 1992, 1 000–1 150 pairs were observed breeding on Macquarie Island, similar to the 1 281 recorded in 2005 (DPIPWE data). The population breeding at Heard Island has not been systematically surveyed since 1954. At that time between 200-500 pairs were estimated to breed annually (Downes *et al.* 1959).

Population status outside areas under Australian jurisdiction

The largest breeding population of light-mantled albatrosses is at South Georgia (Islas Georgia del Sur), containing 5 000-7 500 pairs (Thomas *et al.* 1983; P.A. Prince pers. comm., in Gales 1993). The only available information on breeding population trends and status pertains to the small population on Possession Island (Crozet Islands), which decreased by 1.7% per annum between 1966 and 1995 (Weimerskirch and Jouventin 1998).

3.1.6 Northern Giant Petrel *Macronectes halli* Mathews 1912

Jurisdiction	Breeding locality
Australia	**Macquarie Island**
France	Crozet Islands, Kerguelen Islands
New Zealand	Antipodes Islands, Auckland Island, Campbell Islands, Chatham Island
South Africa	Marion Island, Prince Edward Island
Other	South Georgia (Islas Georgia del Sur)

Distribution

The pelagic range of northern giant petrels is widespread throughout the southern oceans, mainly north of the Antarctic Convergence. In summer they occur predominantly in subantarctic to Antarctic waters, usually between 40-64°S in open oceans. Their range extends into subtropical waters (to 28°S) in the winter and early spring. Individuals banded on Macquarie Island have been recovered in South Africa, South Georgia (Islas Georgia del Sur), Chile, Argentina, Fiji and New Zealand.

Banded northern giant petrels from Macquarie Island are frequently observed in Australian waters (particularly along the southern coast) throughout the colder months, the majority of which (94%) are pre-breeding birds (Marchant and Higgins 1990; Reid *et al.* 2002).

Both northern and southern giant petrels were satellite tracked from South Georgia (Islas Georgia del Sur) and overall the foraging ecology of the two species was similar (Gonzalez-Solis *et al.* 2000). Interspecific and intersexual competition may be reduced by the limited overlap in the at-sea range, with southern giant petrels foraging further south than did northern giant petrels, and females further west than males, suggesting some spatial partitioning in foraging areas. Male northern giant petrels foraged almost exclusively on the South Georgia (Islas Georgia del Sur) coast; their strong dependence during the brooding and chick-rearing period on Antarctic fur seals, whose population has increased exponentially in recent years, may be reflected in the recent population increase of northern giant petrels at South Georgia (Islas Georgia del Sur) (Gonzalez-Solis *et al.* 2000).

At Macquarie Island, satellite transmitters were attached to northern giant petrels during early chick rearing. Most breeding birds spent almost all their time foraging within 100 km of the island. One bird undertook an oceanic trip and this might have been indicative of the foraging trips undertaken earlier in the breeding cycle. Fledglings, on the otherhand, headed almost due east from Macquarie Island and covered thousands of kilometres across the Pacific Ocean to the coast of South America over a three-week period (Trebilco *et al.* 2006).

Breeding Biology

This species is similar to the southern giant petrel and was not recognised as a separate species until the 1960s, when a detailed study of the breeding biology of the giant petrels uncovered that there were actually two distinct species breeding side by side on Macquarie Island (Bourne and Warham 1966). Unlike southern giant petrels, northern giant petrels seldom breed in colonies but rather as dispersed pairs. The nests are typically built in secluded, coastal sites.

Northern giant petrels breed annually (Voisin 1988). Breeding pairs establish their nest sites in August. The egg is laid between August and October, and hatches two months later (Burger 1978; Johnstone 1978). At Macquarie Island, eggs are laid from October 10 to October 27, and hatch from December 15 to early January (DPIPWE data).

Breeding success varies between sites. At Macquarie Island, breeding success ranges between 46 and 75% (1994-2003, DPIPWE data). A successful nest attempt sees the chick fledging at around 110 days of age, leaving for sea in late February to late April.

Northern giant petrels become reproductively mature around six years of age. However, most northern giant petrels do not commence breeding until they have reached 9–11 years of age. Adult northern giant petrels tend to be more sedentary than adult southern giant petrels (Voisin 1988).

Foraging ecology

Northern giant petrels are among the largest and most sexually size dimorphic species of seabirds, with females being only 80% the mass of males. Both sexes scavenge on seal and penguin carrion in the subantarctic ecosystem, but during the breeding season females also feed extensively on other marine food resources and show more pelagic habits than males. The outstanding sexual segregation in foraging and feeding ecology in northern giant petrels suggests that mechanisms maintaining sexual size dimorphism by ecological factors may be operating (Gonzalez-Solis 2004).

At sea, both sexes are aggressive opportunists. Most prey is taken via surface-seizing, but they are also capable of surface-diving and pursuit-plunging down to about 2 m, and have been observed swimming under water with their feet in pursuit of prey (Harper *et al.* 1985; Harper 1987). They were thought to be predominantly diurnal feeders (Brook and Prince 1991) although more recent observations show that northern giant petrels predate on penguins at night (Le Bohec *et al.* 2003).

Global population status

The global breeding population of northern giant petrels is around 11 500 breeding pairs (Table 3.11: BirdLife International 2007). This total suggests an increase of 34% since the last published estimate of 8 600 pairs (Hunter 1985). However, this apparent increase may partially reflect better monitoring in recent years.

Northern giant petrels breed at several localities, ten of which had less than 500 annual breeding pairs, and none had more than 2 200 pairs at last census (Table 3.11).

Table 3.11: Breeding populations of northern giant petrels

Breeding locality	Annual no. breeding pairs	Year of census	Census reliability	Population trend
Macquarie Island	1689[1]	2009	High	Stable
Prince Edward Island	180	1990	High	?
Marion Island	453	1997	High	Increasing
Crozet Islands				
- Ile aux Cochons	250	1981	High	?
- Ile des Pingouins	165	1981	High	?
- Ile de L'est	190	1981	High	?
- Ile des Apotres	150	1981	High	?
- Ile de la Possession	306	1994	High	Increasing
Kerguelen Islands	1,400	1985	Moderate	?
South Georgia (Islas Georgia del Sur)	2,062	1995	High	Increasing
- Bird Island	1,495	1978	Moderate	?

- Other Islands				
New Zealand Islands				
- Antipodes Islands	131[2]	2000	High	?
- Auckland Island	100	1972	Moderate	?
- Chatham Island	2,150	< 1986	Moderate	?
- Campbell Island	234[3]	1997	High	?

? Population trend is unknown due to a lack of recent or consistent population censuses
Data from sources cited in Patterson *et al.* (in press), except [1] DPIPWE data; [2] Wiltshire and Hamilton 2003; and [3] Wiltshire and Schofield (2000).

Population status within areas under Australian jurisdiction

The original population size of northern giant petrels breeding on Macquarie Island is unknown. It is likely that many individuals were harvested for food by sealers throughout the 19th Century. In 2007, about 1 800 pairs bred on Macquarie Island (DPIPWE data). Around 1 000 pairs were estimated to be breeding there in 1970/71 (Johnstone 1977), although this figure is of low accuracy.

Population status outside areas under Australian jurisdiction

There are few recent available data on the size of northern giant petrel populations breeding on islands that are not under Australian jurisdiction. Numbers of giant petrels across their range have been compiled but the publication of the manuscript has been protracted (Patterson *et al.* in press). The very small Campbell Island population had been in decline for some years (Robertson and Bell 1984), but may have recovered slightly in recent times (Wiltshire and Schofield 2000). Nel *et al.* (2002) reports that the increase in the breeding population of northern giant petrels observed on Marion Island between 1989 and 1997 appear to be part of a global increase in this species which has been linked to an increase in Antarctic and subantarctic fur seals and also increased waste from commercial fishing operations. Population estimates for breeding giant petrels at Bird Island suggest an increase at this site, again inferred to be linked to increases in seal carrion (González-Solís *et al.* 2000).

3.1.7 Southern Giant Petrel *Macronectes giganteus* Gmelin 1789

Jurisdiction	Current breeding locations
Australia	**Macquarie Island, Heard Island, McDonald Island, Australian Antarctic Territory**
Antarctica	Antarctic Peninsula, Anvers Island, Elephant Island, Greenwich Island, King George Island, Livingston Island, Nelson Island, Robert Island, Seal Island, South Orkney Islands
Argentina	Isla de los Estados, Isla Observatorio
Chile	Islas Diego Ramírez, Isla Noir
France	Crozet Islands, Kerguelen Islands

South Africa	Prince Edward Island, Marion Island
United Kingdom	Gough Island
Other	Falkland Islands (Islas Malvinas), South Georgia (Islas Georgia del Sur), South Sandwich Islands (Islas Sandwich del Sur)
Jurisdiction	**Former breeding colonies – Now extinct**
United Kingdom	Tristan da Cunha Island
Norway	Bouvet Island

Distribution

Southern giant petrels range widely throughout the southern oceans. In summer they occur predominantly in subantarctic to Antarctic waters, usually below 60°S in the South Pacific and south-east Indian Oceans, or 53°S in the Heard Island and Macquarie Island regions. Some adults are mainly sedentary, remaining close to their breeding islands throughout the year. Nonetheless, numbers diminish at all sites over winter—the Antarctic colonies being completely abandoned. Throughout the colder months, the immatures and most adults disperse widely. The dispersal is circumpolar, extending north from 50°S to the Tropic of Capricorn and sometimes beyond. Thus, in winter they are rare in the southern waters of the Indian Ocean, and more common off South America, South Africa, Australia and New Zealand. The waters off south-east Australia may be particularly important wintering grounds (Marchant and Higgins 1990). Most (84%) southern giant petrels sighted off south-east Australia are immature birds (Reid *et al.* 2002).

The mean foraging range of breeding adults may vary markedly. Adults were observed foraging 30 km from their colony at Hawker Island in the Australian Antarctic Territory (Green 1986), 190 km from South Georgia (Islas Georgia del Sur) (Croxall and Prince 1987), and 470 km from Palmer Island (Obst 1985). One satellite-tracked adult from a breeding colony on the Antarctic Peninsula was recorded foraging in the South Pacific Ocean over 2 000 km away (Parmelee *et al.* 1985, in Marchant and Higgins 1990). It is not known if this individual subsequently returned to the nest.

At Macquarie Island, during the incubation stage southern giant petrels undertook long trips of up to 19 days south of Macquarie Island, often covering thousands of kilometres to areas south of the Antarctic Circumpolar Current (Trebilco *et al.* 2006). As the chicks hatched, the length of the foraging trips decreased and birds spent more time close to Macquarie Island, within the boundaries of the Marine Park. On leaving the nest, fledglings spent a short time relatively close to Macquarie Island before heading east and crossing the Pacific Ocean.

Southern giant petrels breeding on an island off Argentina foraged over the middle of the shelf break between 43° and 51°S. The maximum linear distance from the nest was 552 km; all three birds flew more than 400 km a day and maximum foraging trip distance was 2 540 km (Quintana and Dell'Arciprete 2001).

At South Georgia (Islas Georgia del Sur) southern giant petrels foraged further south than did northern giant petrels, and females further west than males, suggesting some spatial partitioning in foraging areas. Foraging areas of giant petrels overlapped extensively with longline fishery distribution, highlighting their susceptibility to being caught on longline hooks. Females were at higher risk during the study period since they made longer trips and foraged further west than males, into areas where local longline fisheries are more active (González-Solís *et al.* 2000).

Breeding biology

Southern giant petrels are thought to breed annually although there is some evidence to suggest they may take occasional leave from breeding (Hunter 1984; Voison 1988). The pairs return to their breeding sites in August and September, forming dispersed colonies of ten to 300 pairs. The large nests of southern giant petrels are normally built in exposed areas of open vegetation (Voisin 1988). On Macquarie Island nests are normally about 3 m apart (Warham 1962, in Marchant and Higgins 1990).

The egg is usually laid in September-October, hatching some 60 days later (Burger 1978; Johnstone 1978). At Macquarie Island, however, the egg is typically laid earlier, from August 20 to September 6. Hatching occurs from late October to mid November. The chick is brooded constantly in shifts for the first 18 days (Voisin 1988).

At Macquarie Island, between 1996 and 2004 breeding success was varied between 34–61%. If successful, the chick fledges between late January and late March at about 115 days of age (DPIPWE data). The young giant petrels then disperse for several years. Birds banded as chicks on Heard Island and the AAT have been recorded up to 12 500 km away off South America, and off Fiji, Tahiti, Easter Island and New Zealand (Downes *et al.* 1959; Orton 1963, in Marchant and Higgins 1990; Parmelee and Parmelee 1987, in Marchant and Higgins 1990; Woehler and Johnstone 1988). At 6–7 years of age the birds return to their natal colony as reproductive adults (Voisin 1988).

Foraging ecology

On land, southern giant petrels (especially the males) scavenge mainly for seal or penguin carrion. At sea, cephalopods and fish are primarily taken by surface seizing. Southern giant petrels will only very occasionally dive to shallow depths to capture prey (Harper 1987).

Gonzalez-Solis *et al.* (2002) studied foraging activity of giant petrels during the incubation period, by simultaneously deploying activity recorders and satellite transmitters on southern giant petrels at Bird Island. Satellite tracking showed the birds undertook pelagic trips, foraging at sea for marine prey or potentially scavenging on distant archipelagos e.g. South Sandwich (Islas Sandwich del Sur), Falkland (Islas Malvinas) or South Orkney Islands. Males and females exhibit clearly defined spatial segregation in their foraging areas (Quintana and Dell'Arciprete 2001, González-Solís *et al.* 2000).

Using an automatic identification system and an infrared video camera, Le Bohec *et al.* (2003) followed giant petrels tagged with micro transponders. This work showed that giant petrels predate king penguin chicks during the night. The activity of giant petrels was even slightly higher during nighttime than during the day, although southern giant petrels were less nocturnal than northern giant petrels.

Global population status

Synthesising the information in the following table, it appears that the global breeding population of southern giant petrels is around 56 000 annual breeding pairs (Table 3.12). This figure includes significant new information from the Falkland Islands (Islas Malvinas) combined with a range of other much more dated and less reliable estimates. The publication of a much awaited review (Patterson *et al.* in press) of the global status of southern giant petrels has been protracted, a comprehensive review being urgently required.

Thirty populations contain 500 or fewer annual breeding pairs. Fifteen of these localities have 50 or fewer breeding pairs. These populations are of a critically low size and hence are in danger of extinction. Many of the breeding populations have suffered serious declines although increases have been documented in recent years, especially at the largest of the populations. Southern giant petrels have been extirpated from at least two islands (Bouvet Island and Tristan da Cunha Island), and they no longer breed around Signy Island base.

Table 3.12: Breeding populations of southern giant petrels

Breeding locality	Annual no. breeding pairs	Year of census	Census reliability	Population trend
Macquarie Island	2 534[1]	2010	High	Stable
Heard Island	3 150[2]	1988	Moderate	?
McDonald Island	1 400	1979	Moderate	?
AAT				
- Giganteus Island	3-4	2007	High	Stable
- Hawker Island	45[3]	2009	High	Stable?
- Frazier Islands (total)	~250	2001	High	Increasing?
- Dewart Island	135	2001	High	Increasing?
- Charlton Island	20	2001	High	Increasing?
- Nelly Island	93	2001	High	Increasing?
Antarctic Peninsula	690	< 1997	Moderate	?
Anvers Island	634	< 1997	Moderate	?
Livingston Island	366	< 1994	Moderate	?
Greenwich Island	41	1966	High	?
Robert Island	286	< 1986	High	?
Nelson Island	912	< 1995	High	?
King George Island	3 592	< 1995	Moderate	?
Elephant Island	845	1971	Moderate	?

Seal Island	25	1971	High	?
South Orkney Islands				
- Signy Island	3 036	< 1988	Low-Mod	?
- Laurie Island	398	< 1995	Moderate	?
South Sandwich Islands (Islas Sandwich del Sur)	1 551	1996	?	?
Bouvet Island	0	1989	High	Extinct
Crozet Islands				
- Ile aux Cochons	575	1981	Moderate	?
- Ile des Pingouin	50	1981	High	?
- Ile de L'est	323	1981	High	?
- Ile des Apotres	10	1981	High	?
- Ile de la Possession	105	1994	High	?
Kerguelen Islands	3–5[4]	1987	High	?
Marion Island	1 343[5]	2008	High	Decreasing
Prince Edward Island	1 000[5]	2002	High	?
Falkland Islands (Islas Malvinas)	19 529[6]	2005	High	Increasing
South Georgia (Islas Georgia del Sur)				
- Bird Island	521	1995	High	Increasing
- Albatross Island	150	1976	High	?
- South Georgia	5 500	1978	Low	?
- Salisbury Plain	3 550	1976	Low	?
Isla Noir	1 000[5]	2004	?	?
Isla Diego Ramírez	181[5]	1981	?	?
Isla Gran Robredo	1883[7]	2005	High	?
Isla Arce	448[7]	2005	High	?
Isla Observatorio	500[8]	2004	High	?
Isla de los Estados	30	1971	Moderate	?
Gough Island	225–245[9]	2002	High	Increasing
Tristan da Cunha	0	< 1870	High	Extinct

Data from Patterson *et al.* (in press) except: [1] DPIPWE data [2] Kirkwood *et al.* (1995); [3] AAD preliminary data, [4] Weimerskirch *et al* 1989, [5] ACAP 2010, [6] Reid and Huin 2008, [7] Quintana *et al.* 2006, [8] Quintana *et al.* 2005; [9] Cuthbert and Sommer 2004, [6]
? Population trend is unknown due to a lack of recent or consistent population censuses

Population status within areas under Australian jurisdiction

Some southern giant petrel populations may have fared reasonably poorly in Australian waters. It is speculated that breeding populations may have decreased at some breeding localities, such as Heard Island. Across all southern giant petrel populations in the Australian Antarctic Territory, the extent of inter-annual variation is unknown (due to the lack of a sequence of regular counts), but could be high, and consistent count methodologies have not been used in the past, limiting inter-annual comparability and the ability to accuarately determine population trends.

The population on Heard Island decreased from 5 250 pairs in the early 1950s to 3 150 pairs in 1987/88 (Kirkwood *et al.* 1995). Similarly, preliminary analysis of the available data on southern giant petrels on Macquarie Island indicates that this population may have declined by almost half over the last two decades. An estimated 4 000 pairs bred at Macquarie Island in 1970/71 although this estimate is low in reliability (Johnstone 1977). However, in 1999 about 2 300 breeding pairs remained. The population at Macquarie Island appears to have stabilised at this level with 2 570 pairs being recorded in 2007 and 2 534 pairs in 2010.

Population status outside areas under Australian jurisdiction

Southern giant petrels formerly bred on Tristan da Cunha, but they were extirpated by 1870 (Hagen 1952, in Marchant and Higgins 1990). Only one breeding pair remained at Bouvet in 1981, but none were found in 1989. The establishment of a field station at Signy Island (off the Antarctic Peninsula) led, within eight years, to the complete desertion of the colony of 200 breeding pairs and caused a decrease in the breeding population elsewhere on the island (Rootes 1988, in Marchant and Higgins 1990). Similarly, the establishment of an Antarctic research station at Dumont d'Urville saw the breeding population decrease from 69 pairs in 1969 to only two pairs in 1980 (Jouventin *et al.* 1984). A maximum of five pairs bred at the Kerguelen Islands in 1987. Previously the Falkland Islands (Islas Malvinas) populations have been seriously reduced following shooting of adults and destruction of eggs, as the giant petrels were thought to menace sheep (Woods 1975, in Marchant and Higgins 1990). However, a comprehensive survey in 2004 revealed about 19 800 pairs, a dramatic increase from all previous estimates (Reid and Huin 2005). Recently other colonies have also remained stable or increased, e.g. South Georgia (Islas Georgia del Sur), and Gough Island.

Given the changes in population trend for this species at several important breeding sites, there is an urgent need for a global assessment of the population status of this species.

3.2 Species Foraging but not Breeding in Areas under Australian Jurisdiction

This section briefly describes the breeding and non-breeding distributions, breeding biology and population status of each of the fourteen albatross species foraging but not breeding within the Australian Fishing Zone and which potentially forage there.

3.2.1 Tristan Albatross *Diomedea dabbenena* Matthews 1929

Previous name

Wandering albatross *Diomedea exulans dabbenena*

Jurisdiction	Breeding locality
United Kingdom	Inaccessible Island (Tristan da Cunha Islands), Gough Island

Endemic to territories of the United Kingdom.

Distribution

The at-sea distribution of this species has only recently been defined. Cuthbert *et al.* (2005) satellite tracked 38 breeding Tristan albatrosses and assessed the seasonal and annual at-sea distribution of these birds in relation to reported pelagic longline fishing effort. These birds ranged across the South Atlantic from 50° W to 15° E with most (97%) daytime satellite fixes between latitudes 30° S and 45° S. Considerable fishing effort occurred within the same latitudes.

Outside the breeding season it disperses to South Atlantic and South African waters, with numerous reports from Brazilian waters, and one from Australia, suggesting that birds occasionally disperse into the Indian Ocean (BirdLife International 2007). The single Australian record of this species is from a recapture off Wollongong (NSW) in September 1997. The bird had been banded as a chick on Gough Island four years prior (Leishman 1998a; L. Smith pers. comm.).

Breeding biology

Tristan albatrosses breed biennially when successful. Pairs return to the nest site in early December and most eggs are laid between late December and February. The chick fledges the following November/December (Cuthbert *et al.* 2004).

At Gough Island in September 2001, breeding success was just 27.3% (Cuthbert *et al.* 2004). However, breeding success varied considerably in different areas of the island, ranging from 17.6 to 68.0%. Most breeding failures reflected mortalities of large chicks, and over four years 75% of breeding failures occurred during the chick period. Predation by introduced house mice *Mus musculus* was the most likely cause of chick mortality. Among the small study population, birds began breeding at an average age of 9.7 years and annual adult survival from 1985 to 2001 was 92.6% (SE=1.6%).

Global population status

Tristan albatrosses once bred on the main island of the Tristan Group but were extirpated by humans by 1907 (Watkins 1987). Several hundred pairs formerly bred on Inaccessible Island. However, predation by introduced pigs devastated the colony, and by the 1940s only two or three pairs remained. This tiny population has not increased since (Ryan *et al.* 1990).

The only other breeding population is at Gough Island, with a breeding population of about 1 500 pairs each year (Ryan *et al.* 2001). Both breeding success and adult survival estimates are low in comparison with other *Diomedea* species and population modelling predicts a population decreasing at an annual rate of 2.9-5.3% (Cuthbert *et al.* 2004). Further research is needed urgently to assess whether breeding success is typical, and to confirm that mouse predation is the cause of chick mortality. The low productivity of this species will compound the negative impacts of longline fishing mortality, which are likely to be reducing adult and juvenile survival.

3.2.2 Antipodean Albatross *Diomedea antipodensis*

Previous name

Wandering albatross *Diomedea exulans antipodensis*

Jurisdiction	Breeding locality
New Zealand	Antipodes Island, Campbell Island, Pitt Island Auckland Islands

Endemic to New Zealand. Robertson and Nunn (1998) split *D. antipodensis* into *D. antipodensis* and *D. gibsoni*. However, in 2006 the ACAP Taxonomy Working Group concluded that available data do not warrant the recognition of Gibson's and Antipodean albatrosses as separate species (Burg and Croxall 2004; Brooke 2004; Double 2006).

Distribution

The Antipodean albatross disperse over the Tasman Sea and South Pacific Ocean (Marchant and Higgins 1990).

Satellite telemetry between 1994 and 2004 showed that birds from the Antipodes Island population forage mainly in the Pacific Ocean east of New Zealand, and the range of non-breeding birds was larger than that of breeders (Walker and Elliott 2006). Non-breeding males had the largest range, foraging off the coast of Chile, Antarctica and in the tropical South Pacific. They preferred to forage at the outer edge of shelves and over seamounts, particularly where there were strong currents or eddies and productivity was enhanced, as well as over deep water. Individuals of all stages of maturity preferred large foraging areas. They can travel great distances; a male flew 8 000 km to Chile in 17 days (Nicholls *et al.* 1996, 2000).

For the Auckland Island population, males and females appear to utilise different foraging areas. The females tend to frequent the Tasman Sea in the vicinity of 40°S, while the males either disperse westwards at lower latitudes or travel north-east towards the mid-Pacific Ocean (Elliot *et al.* 1995). Non-breeding male and female birds foraged westward to the south-eastern Indian Ocean but avoided Antarctic waters (Walker and Elliott 2006).

Breeding biology

The Antipodean albatross is a biennial breeder, when successful. Nests are built in very loose colonies (26 nests per 10 000 m^2: Warham and Bell 1979, in Marchant and Higgins 1990). Females lay from January at Antipodes Island and February at Campbell Island. Chicks fledge between January and March the following year (Robertson 1985). Average annual survival over 10 years was 0.96 (Walker and Elliott 2005: Walker *et al.* 2002). Productivity over 11 years (1994–2005) averaged 0.74 chicks per nesting pair.

For the Auckland Island population, pairs begin returning to the colonies from December. Females usually lay from late December to early February (Walker and Elliott 1999). The egg is incubated for 80 days before it hatches in about late March (Bailey and Sorensen 1962, in Marchant and Higgins 1990). The chick fledges the following year between January and February (Bailey and Sorensen 1962, in Marchant and Higgins 1990). Breeding success was 64% during the 1989–90 breeding season (P. Dilks pers. comm., in Gales 1993) and between 1991 and 2001, 61–78%, averaging 63% annually (Walker and Elliott 1999; Walker and Elliott 2002).

Global population status

For this biennial species, approximately 11 500 pairs of Antipodean albatrosses breed each year, which translates to a total of ca. 15 100 to 17 300 breeding pairs (Elliott and Walker 2005; BirdLife International 2007).

An estimated 5 150 pairs breed each year on Antipodes Island (Walker and Elliott 2005; Elliott and Walker 2005), less than six on Campbell Island (Gales 1998) and a single pair nested on Pitt Island in the Chatham Island group in 2005 (Miskelly in Elliott and Walker 2005). After a period of significant decline in the 1970s and 1980s the Antipodes population has been increasing at a rate of about 3% per annum (Elliot and Walker 2005). The Campbell Island population has been stable at low numbers (Taylor 2000a) for at least three decades.

For the Auckland Island group, in 1997 72 pairs bred on Auckland Island, 352 on Disappointment Island and 6 993 on Adams Island (Walker and Elliott 1999). An estimated 5 831 pairs of Gibson's albatross breed in the Auckland Islands group each year: most, > 95%, on Adams Island and the rest on nearby Disappointment and Auckland Islands (Walker & Elliott 1999). The Adams Island population was estimated as 13 000 pairs in the 1970s; however, this was a poor quality estimate only, so it is uncertain whether it represents an accurate indication of the decrease in the population (K. Walker pers. comm., in Gales 1998). In 2005 the Auckland Island population was assessed as stable (Elliott and Walker 2005).

3.2.3 Northern Royal Albatross *Diomedea sanfordi* Murphy 1917

Previous name

Northern royal albatross *Diomedea epomophora sanfordi*

Jurisdiction	Breeding Locality
New Zealand	Chatham Islands, South Island (Taiaroa Head), Auckland Islands

Endemic to New Zealand.

Distribution

Northern royal albatrosses have a circumpolar range at sea, being most common between 36°S to at least 52°S. Individuals disperse to the south West Atlantic off Argentina, the eastern South Pacific near Chile, the southern Indian Ocean and south-east Australia. Satellite tracking has revealed that whilst attending eggs and young chicks, adults remain primarily within 300 km of the colony, the range increasing with age of chick (BirdLife International 2004b). Most locations were confined to the shelf edge and slope (Nicholls *et al.* 2005).

The range of six individuals visiting the mid-shelf, shelf break and slope, and sometimes inshore, of Patagonia between January and October averaged 227 000 km^2, but each had a core area about one-tenth that of the range (Nicholls *et al.* 2005). Immature birds are also highly dispersive, and are rarely recovered (Marchant and Higgins 1990; Nicholls *et al.* 1994). While northern royal albatrosses occur infrequently in waters off NSW, they are regularly recorded throughout the year around Tasmania and South Australia at the edge of the continental shelf (Blakers *et al.* 1984).

Breeding biology

Northern royal albatrosses breed biennially if successful. Adult birds return to their breeding grounds between October and November. The female lays about a month later. The hatchling emerges after two months of incubation, and fledges some eight months later, from September to October (Robertson 1991). Mean breeding success at the South Island colony is 31% (Westerskov 1963, in Marchant and Higgins 1990). Population declines are resulting from continued very poor breeding success as a result of a significant decrease in habitat quality.

Young birds start to return to their natal colony at 4–8 years of age, and begin breeding after a minimum of nine years. Northern royal albatrosses have lived for at least 61 years in the wild (Robertson 1998).

Diet during the breeding season was mostly cephalopods and fish, and small amounts of tunicates and crustaceans, probably taken relatively close to land and not in Antarctic waters (Imber 1999).

Global population status

Most (99%) northern royal albatrosses breed at the Chatham Islands where there is an estimated breeding population of 6 500–7 000 pairs, equivalent to a projected total population of about 20 000 individual birds (Robertson 1998). The South Island Taiaroa Head population is small, with less than 28 pairs breeding each year (Taylor 2000a). This tiny colony includes five southern royal x northern royal albatross hybrids. The population, established in 1920, is slowly increasing under intensive human surveillance and management. Two hybrid pairs have also been recorded on Enderby Island, Auckland Islands (Taylor 2000a). All other northern royal albatrosses breed at the Chatham Islands. These populations are decreasing and this trend is expected to continue (C.J.R. Robertson pers comm., in Gales 1998; BirdLife International 2004a).

3.2.4 Southern Royal Albatross *Diomedea epomophora* Lesson 1825

Previous name

Southern royal albatross *Diomedea epomophora epomophora*

Jurisdiction	Breeding Locality
New Zealand	Adams Island, Auckland Island, Campbell Island, Enderby Island

Endemic to New Zealand.

Distribution

Southern royal albatrosses have a circumpolar distribution within the Southern Oceans, foraging from 36°S to 63°S, spending most of their time south of 47°S. Immature birds are especially dispersive. Breeding adults forage from the South Island southwards to the Campbell Plateau and may circumnavigate the Southern Ocean after breeding (Croxall and Gales 1998).

Satellite tracking has shown that during the incubation period foraging activity was restricted to shelf and shelf-break areas within 1 250 km of their breeding site. Foraging activity by 8 of the 14 individuals tracked was concentrated at a zone near the Snares Islands, on the Campbell Plateau (Waugh *et al.* 2002).

Southern royal albatross spent only 35% of their time sitting on the water, and made on average 2.6 takeoffs per hour. Further, royal albatross showed a similar pattern of activity during all periods of the day (Waugh and Weimerskirch 2003).

Southern royal albatrosses range over the waters off southern Australia at all times of year, but especially between July and October. They have been recorded from Byron Bay in the east to south-western Western Australia. Most records are from the shelf-break areas, especially off western and southern Tasmania and around Victoria (Blakers *et al.* 1984).

Breeding biology

Southern royal albatrosses have a biennial breeding cycle. Breeding birds return to their nesting grounds from late October to mid-November. Nests are built in dispersed colonies. The female lays her egg in November-December, which hatches in February–March after two months of incubation. The chicks fledge eight months later from October 6 to December 12. Breeding success averages 62% (46–74% at Campbell Island: Waugh *et al.* 1997).

The juveniles disperse widely without returning to their natal colony until they have reached 4–8 years of age. Southern royal albatrosses do not begin breeding until they are at least nine years old (Waugh *et al.* 1997).

Global population status

The 1996 breeding population was estimated to be 8 200–8 600 pairs (Taylor 2000a) and stable (BirdLife International 2007), possibly increasing (Robertson *et al.* 2003). These individuals are divided among four populations, three of which have fewer than 60 annual breeding pairs (P. Moore pers. comm., in Gales 1998; K. Walker pers. comm., in Gales 1998). The slowly increasing population at Enderby Island (55 pairs in 1995) represents the recolonisation of the site in 1940 after their local extirpation in the 1860s (P. Moore pers. comm., in Gales 1998).

3.2.5 Amsterdam Albatross *Diomedea amsterdamensis* Roux *et al.* 1983

Previous name

Amsterdam albatross *Diomedea amsterdamensis*

Jurisdiction	Breeding locality
France	Amsterdam Island

Endemic to Amsterdam Island (France).

Distribution

Amsterdam albatrosses nest only on Amsterdam Island in the Indian Ocean. Their pelagic range is poorly known, as a result of the similar appearance to other albatross species such as the wandering albatross, *D. exulans*. Most sightings have been of birds in the Indian Ocean, although it is likely that immature and non-breeding birds disperse much further from Amsterdam Island. Unsubstantiated sightings of this species have been recorded from both Australia and New Zealand (N. Brothers pers. comm. in Gales 1998; Shirihai 2002). Limited satellite tracking data shows that birds travel up to 2 200 km away from Amsterdam Island when foraging between incubation shifts (BirdLife International 2004b).

Breeding biology

Amsterdam albatrosses are biennial breeders. Adult birds begin arriving at Amsterdam Island in January. Eggs are laid in late February-March and hatch 79 days later in May. The chicks fledge in January-February after spending 235 days in the nest (Jouventin *et al.* 1989). Mean breeding success has been measured as 72% (Weimerskirch *et al.* 1997a). On average, each breeding pair produces one egg every 1.8 years and fledges a chick every 2.4 years (Jouventin *et al.* 1989). The offspring then range the seas for 4–7 years before returning to the island. Individuals do not begin breeding until they are nine years of age.

Global population status

There are very few Amsterdam albatrosses remaining, with only about 25 eggs laid each year (Weimerskirch 2004; ACAP 2007a). This species is among the world's rarest seabirds, and at great risk of extinction. The number of pairs breeding each year has increased from five pairs—apparently reduced to this level by longline fishing around the island from the mid 1960 to mid 1980s, when monitoring studies began (Weimerskirch *et al.* 1997a). The population is banded and has been monitored for two decades; from 1993 to 2003 it increased at a rate of 7% annually. Modelling suggests that the population could not sustain any level of incidental bycatch (Inchausti and Weimerskirch 2001).

A cause for concern is the gradual decrease in breeding success since 1983. Avian cholera has been identified as the cause for depressed productivity in conspecific Indian yellow-nosed albatross (*T. carteri*), although this disease has not yet been identified in Amsterdam albatross (Weimerskirch 2004).

3.2.6 Laysan Albatross *Phoebastria immutabilis* Rothschild 1893

Previous name

Laysan albatross *Diomedea immutabilis*

Jurisdiction	Breeding locality
Japan	Mukojima (Bonin Islands)
Mexico	Isla Guadalupe, Isla Clarion, Isla San Benedicto
U.S.A.	French Frigate Shoals, Kauai Island, Kaula Island, Kure Atoll, Laysan Island, Lisianski Island, Midway Atoll, Necker Island, Niihau Island, Pearl and Hermes Reef

Distribution

Laysan albatrosses are birds of the North Pacific Ocean. Throughout the breeding season most are located between Japan, the Aleutian Islands and Hawaii (Harrison 1990). They rapidly disperse after breeding, primarily over oceanic waters or along the continental shelf-break as far north as the Bering Sea and eastwards to the Pacific Coast of North America and Mexico. They are most numerous on the western side of the North Pacific, with the largest concentrations occurring off eastern Japan (del Hoyo *et al.* 1992). Laysan albatrosses have only ever been recorded three times south of the equator. In 1985 and 1986 a solitary Laysan albatross (presumed to be the same individual) was seen on Norfolk Island (Leishman 1998b).

Breeding biology

Adult Laysan albatrosses return to their colonies each year in late October and early November. Nesting is colonial. Egg laying occurs in November-December. Incubation lasts 65 days. The chick emerges in January-February and remains in the nest for a further 165 days before fledging in June-July (Harrison 1990). Breeding success ranges from 49%–78% (Fisher 1975, 1976; van Ryzin and Fisher 1976). Sexual maturity is reached after 5–16 years (van Ryzin and Fisher 1976).

Global population status

The current population of Laysan albatrosses approaches 437 000 breeding pairs, (BirdLife International 2007). This figure makes it the most numerous of the North Pacific albatross species. However, a decline of 32% has been recorded between 1992 and 2002 at the Northwestern Hawaiian Islands where 90% of the global population is found (BirdLife international 2007).

3.2.7 Campbell Albatross *Thalassarche impavida* (Mathews 1912)

Previous names

Black-browed albatross *Diomedea melanophris impavida*, New Zealand black-browed albatross.

Jurisdiction	Breeding locality
New Zealand	Campbell Island

Endemic to New Zealand.

Distribution

Campbell albatrosses occur in Antarctic and subantarctic waters and in the subtropical South Pacific Ocean. They breed only on subantarctic Campbell Island, south of New Zealand. Throughout the breeding season, breeding adults are generally found over the shelf waters surrounding New Zealand, specifically around the South Island and Chatham Rise, southwards to the Ross Sea (Waugh *et al.* 1999d). Non-breeding birds often forage over the continental slopes around Tasmania, Victoria and New South Wales. Their post-breeding dispersal is restricted to the temperate shelf waters of New Zealand, Australia and the South Pacific Ocean (Marchant and Higgins 1990; Moore and Moffat 1990).

Breeding biology

Pairs breed annually. Adults return to Campbell Island to begin breeding in August. Nests are built in dense colonies. The egg is laid from September 18 to October 8. The young fledge at 7–8 months of age in April-May. Breeding success between 1984 and 1994 averaged 66%. Annual adult mortality is estimated as 4.5% (Waugh 1999a).

Global population status

Population estimates by Robertson (1980, in Moore and Moffat 1990) of 74 825 pairs in 1976 appear to be over-estimates (Moore and Moffat 1990). In 1987/88 an estimated 19 000–26 000 pairs bred on Campbell Island, signalling an overall decline of 38–57% since 1942, with some colonies falling by as much as 88% (Moore and Moffat 1990). Photographic evidence also indicates that there have been significant decreases in the population in recent decades (Moore 1995, 1999). Counts in 1995-97 have recorded an estimated 23 500 pairs (P. Moore in lit. 2003 in BirdLife International 2007). The current population is listed as being stable (BirdLife International 2007).

3.2.8 Buller's Albatross *Thalassarche bulleri* Rothschild 1893

Previous name

Buller's albatross *Diomedea bulleri bulleri*, Southern Buller's albatross

Jurisdiction	Breeding Locality
New Zealand	Snares Island, Solander Islands Chatham Islands, Three Kings Island

Endemic to New Zealand. The ACAP Taxonomy Working Group recently concluded that available data do not warrant the separation of Buller's and Pacific albatrosses at the specific level (Double 2006; see also Brooke 2004).

Distribution

Generally, adults forage between 40 and 50°S from Tasmania eastwards to the Chatham Rise, while juveniles and non breeding adults disperse across the South Pacific Ocean to the coast of South America (BirdLife International 2004b).

Satellite tracking data for birds from colonies on Snares Islands and Solander Island (south of South Island, New Zealand) suggest that the feeding grounds utilised by individuals, and the duration of their foraging trips, are dependent upon colony, sex, pairing status and developmental stage of egg/chick (Broekhuizen *et al.* 2003).

Two satellite-tracked prebreeding birds from the Snares Island group dispersed to Tasmania, Australia, from late May until at least late July (Stahl and Sagar 2006). Six older birds (five prebreeding birds, one former breeding adult) all adopted a dual strategy of short trips (mean duration 1.3 days, mean foraging range 129 km) and long trips to southern New Zealand (9.6 days, 871 km) or Tasmanian waters (22.0 days, 1 918 km). Along the Pacific coast of South America, Buller's albatrosses preferred the continental shelf and, at least in 1980 and 1995, occurred exclusively in the south (30–40°S) (Spear *et al.* 2003).

Breeding biology

Buller's albatross typically breeds annually. At the Snares and Solander Islands the adults begin returning to the colonies in mid-December, and the egg is laid in January-February. Hatching occurs in mid-March to April. The young fledge in spring from late August to late October (Warham and Bennington 1983; Sagar and Warham 1998). At the Chatham Islands, eggs are laid in November, hatch in January and chicks fledge in June (Robertson 1985; 1991).

Breeding success on Little Sister in 1994-95 has been recorded as 57-60%, lower than the breeding success of 71% recorded at The Snares from 1995-98 (Sagar *et al.* 2002). The species is highly philopatric, particularly the males (Sagar *et al.* 1998). Adult mortality has been estimated at 4–8% annually (Sagar *et al.* 2000).

Global population status

The combined breeding population is estimated at 32 000 pairs. Most birds breed on the Snares (8 713) and Solander (4 912) in the south (Sagar and Stahl 2005), and the Forty Fours (16 000) and Big and Little Sister (2 130) in the Chatham Island group, and Rosemary Rock, Three Kings Islands (20) in the north (BirdLife International 2007). Despite increases recorded at Snares and Solander Islands the propensity for these birds to interact with fishing activities is a cause for concern about future trends, and a potential decrease in the conservation status of the species (Sagar and Stahl 2005). Further, the population estimate for the Forty Fours is crude, being based upon an extrapolation of density and area (Gales 1998).

3.2.9 White-capped Albatross *Thalassarche steadi*

Previous names

Shy albatross *Diomedea cauta cauta*

Jurisdiction	Breeding Locality
New Zealand	Adams Island, Antipodes Islands (Bollons Island), Auckland Island, Disappointment Island Forty-Fours

Endemic to New Zealand. Recent data suggest shy and white-capped albatrosses are divergent and diagnosable and support recognition at the specific level (Gales *et al.* 2003; Abbott *et al.* 2006; Double 2006).

Distribution

It is difficult to know the precise distribution of this newly recognised species due to the difficulties of distinguishing it at sea from shy albatrosses and the absence of specific banding studies. Nonetheless, white-capped albatrosses are the most abundant albatross in all New Zealand shelf waters, except on the Chatham Rise and Bounty Platform (displaced by Salvin's albatross) and the Campbell Shelf (displaced by Campbell albatross). The adults are present in New Zealand and south-east Australian waters throughout the year whilst immatures are rare in New Zealand waters, being more common off south-east Australia and South Africa (Marchant and Higgins 1990).

There have been no published satellite tracking studies, so their distribution has been inferred from molecular analyses of bycatch specimens. Abbott *et al.* (2006) found that both juvenile and adult white-capped albatrosses were recovered from New Zealand, southern Australian and South African and Namibian waters (Baker *et al.* 2007a). One unpublished tracking study reported adults from Auckland Island foraging in New Zealand and also South African waters (Thompson and Sagar 2006). Throughout most of their range, juvenile and adult white-capped albatrosses are exposed to fisheries that collectively kill many thousands of these albatrosses each year (Baker *et al.* 2007a).

Breeding biology

Little is known of the breeding biology of white-capped albatrosses. Pairs nest annually and colonially. Egg laying commences in mid-November and hatching occurs in February. The young fledge in mid-August. Adults remain near the colony during the breeding season, and possibly throughout the entire year (Robertson 1985).

Global population status

The historical and present status of this newly distinguished species is not well understood. Previously, the global population has been reported as ca: 75 000 annual breeding pairs, distributed between Disappointment Island (72 000), Auckland (3 000), Adams Islands (100) in the Auckland Island group, and Bollons Island (50-100) in the Antipodes Island group (Gales 1998). One pair has also been observed breeding at the Forty-Fours, which lies 20 nautical miles east of Chatham Island, in 1991 and 1996 (Robertson & Page 1992; Robertson *et al.* 1997)

More recently, using more accurate aerial photographic monitoring methods, Baker *et al.* (2007b) report approximately 110 500 pairs and 6 500 pairs for Disappointment Island and Auckland Island, respectively, combining to provide a 2006 population estimate of 117 000 pairs for the Auckland Island group. Therefore, the global population for this species is approximately 120 000 pairs.

3.2.10 Salvin's Albatross *Thalassarche salvini* (Rothschild 1893)

Previous names

Shy albatross *Diomedea cauta salvini*, Salvin's albatross, Grey-backed albatross, Bounty albatross.

Jurisdiction	Breeding locality
France	Penguin Island (Crozet Islands)
New Zealand	Bounty Island, Snares Island

Distribution

This species is abundant throughout the year on all continental shelf areas around New Zealand (J.A. Bartle pers. comm. in Gales 1993). It roams widely in winter, moving eastwards across the South Pacific to the Humboldt Current in the waters off the west coast of South America (Chile and Peru). Here it extends north to about 5°S (Marchant and Higgins 1990). On the Pacific coast of South America wintering individuals occurred throughout the Humboldt Current but preferred the continental shelf and more northern latitudes (Spear *et al.* 2003). Small numbers of non-breeding adults regularly fly across the Tasman Sea to south-east Australian waters (Barton 1979; Blakers *et al.* 1984; Reid *et al.* 2002). It is scarce in the southern Indian Ocean, though small numbers occur around the Crozet Islands where it has been recorded breeding (Jouventin 1990). It is only a rare vagrant to the South Atlantic, though small numbers are present in the shelf waters of South Africa (Marchant and Higgins 1990).

Breeding biology

Salvin's albatross probably breeds annually. Adults return to their breeding colonies in September, with the birds at Bounty Island returning 7–10 days later than at Snares Island. The nest is built in a moderately dense colony. Eggs are laid in early October, and begin to hatch in early to mid-November. Breeding adults forage over the shelf waters around the colonies. Chicks fledge in late March to early April (Robertson and van Tets 1982; Robertson 1985).

Global population status

The status of this species is poorly known. A ground count of Salvin's albatross nests on Proclamation Island (Bounty Islands) in November 1997 found 3 062 breeding pairs (Clark *et al.* 1998), but Robertson and Van Tets (1982) estimated that there were 8 656 nests on the same island in 1978. Andrea Booth and Jacinda Amey (pers. comm. 1999 in Taylor 2000a) estimated that there were 30 752 pairs of Salvin's albatross on the Bounty Islands in 1997 using the formula of 139 780 m^2 of suitable nesting habitat in the Bounty group and an average nest density of 0.22 pairs per m^2. An alternative estimate of about 76 000 was derived from the extent of the breeding area in 1978 (C.J.R. Robertson pers. comm., in Gales 1993).

The population at Snares Island is estimated as between 650 and 'a few thousand' breeding pairs (C.J.R. Robertson pers. comm., in Gales 1998), though it has never been surveyed. The four breeding pairs on the Crozet Islands appear to be temporally stable (Jouventin 1990).

The global breeding population, is estimated as roughly 30 750 pairs, and is considered stable (BirdLife International 2007).

3.2.11 Chatham Albatross *Thalassarche eremita* (Murphy 1930)

Previous names

Shy albatross *Diomedea cauta eremita*

Chatham Island albatross

Jurisdiction	Breeding locality
New Zealand	Pyramid Rock (Chatham Islands)

Endemic to New Zealand. The recognition of *T. eremita* is sometimes controversial, although this classification is becoming widely accepted (BirdLife International 2004a; Double 2006; Onley and Schofield 2007).

Distribution

Chatham albatrosses breed only on one small island in the Chatham Islands, located to the east of New Zealand. Breeding adults forage close to their breeding sites, while non breeding birds and juveniles disperse within the South Pacific Ocean west to Tasmania and east to Chile and Peru (BirdLife International 2004b). Chatham albatrosses have also been sighted off southern Africa (Ryan 2002).

Breeding biology

Chatham albatrosses breed annually. Individuals begin arriving at Pyramid Rock in late August to form dense breeding colonies on grassy slopes (Robertson *et al.* 2000). Pairs lay in September with the egg hatching during the last three weeks of October. Chicks fledge around April (Robertson and van Tets 1982; Robertson 1985). Between 60 and 80% of chicks are fledged from eggs laid; birds have been recorded returning to the island at 4 years of age, and first breeding at 7 years, and adult survival is estimated as 86.8% per annum (Robertson *et al.* 2000, 2003; ACAP 2007b).

Global population status

Monitoring of the single population of Chatham albatrosses has been erratic and precludes a definitive assessment of population size and status. Counts of nest sites between 1999 and 2001 provided estimates of between 5 304 and 5 333. When compared to historic data it has been inferred that the population of about 11 000 breeding individuals is stable, although this assessment requires confirmation (BirdLife 2004a; ACAP 2007b).

3.2.12 Atlantic Yellow-nosed Albatross *Thalassarche chlororhynchos* Gmelin 1789

Previous name

Yellow-nosed albatross *Diomedea chlororhynchos chlororhynchos*

Jurisdiction	Breeding locality
United Kingdom	Gough Island, Tristan da Cunha Islands

Endemic to territories of the United Kingdom.

Distribution

Little is known about the oceanic distribution of the Atlantic yellow-nosed albatross. It is most common between 15°S and 50°S in the southern Atlantic Ocean, over both pelagic and inshore waters (del Hoyo *et al.* 1992). Adults may forage far from their southern breeding grounds. Post-breeding adults and juveniles disperse to become abundant off the east coast of South America and the west coast of southern Africa. Atlantic yellow-nosed albatrosses are known to be killed in fishing operations off the coasts of Brazil and Uruguay (Olmos *et al.* 2001; Stagi *et al.* 1998). This species is rarely seen in Australian waters (Marchant and Higgins 1990; Adams 1992).

Breeding biology

The annual breeding cycle of the Atlantic yellow-nosed albatross commences in late August-early September when adults return to their breeding colonies. Pairs may nest solitarily, or in loose colonies to large colonies. The egg is laid in September-October, and hatches in November-December. A few clutches (up to 1%) are of two eggs, but these seem to either be the result of egg dumping or inexperience and are rarely, if ever, successful (Ryan *et al.* 2007). The chicks fledge in April to early May, after spending 130 days in the nest (Elliot 1957). Average breeding success (67–69%) and breeding frequency (66–65%) were similar on Gough and Tristan da Cunha Islands (Cuthbert *et al.* 2003).

Atlantic yellow-nosed albatross are extremely philopatric. Of the 72 birds banded as adults, not one has been recovered away from the island. Adults banded on Tristan da Cunha in 1938 were still alive in 1982 (Hagen 1982). On Gough Island immature and adult annual apparent survival averaged 88% and 92%, respectively, and apparent survival from fledging to age 5 averaged 31% (Cuthbert *et al.* 2003). On Tristan da Cunha, apparent adult survival averaged only 84% and was negatively correlated with longline fishing effort in the South Atlantic Ocean (Cuthbert *et al.* 2003).

Global population status

The global breeding population is estimated to be between 26 600 and 40 600 annual breeding pairs. Cuthbert and Sommer (2004) estimated the population at Gough Island to be 5 300 with a nesting density of 5 pairs per hectare. The population at Tristan da Cunha was estimated to be 21 600 to 35 600 in the 1980s (Fraser *et al.* 1988). Population modelling predicts annual rates of decrease of 1.5-2.8% on Gough Island and 5.5% on Tristan da Cunha (Cuthbert *et al.* 2003).

3.2.13 Indian Yellow-nosed Albatross *Thalassarche carteri* Mathews 1912

Previous name

Yellow-nosed albatross *Diomedea chlororhynchos bassi*

Jurisdiction	Breeding locality
France	Amsterdam Island, Crozet Islands, Kerguelen Islands, St. Paul Island
South Africa	Prince Edward Island

Distribution

Indian yellow-nosed albatrosses predominantly occur within the southern Indian Ocean. They are found over both pelagic and inshore waters between 15°S and 50°S (del Hoyo *et al.* 1992). Even during the breeding season adults can be found foraging at subtropical latitudes. Post-breeding birds are abundant off the southern and eastern coasts of South Africa (Adams 1992), though they have not been recorded any further west (Marchant and Higgins 1990). Indian yellow-nosed albatrosses are the most common albatross in the Great Australian Bight and central Bass Strait. They also occur in the waters east of Tasmania and the Australian mainland north to Coff's Harbour (Barton 1979; Woods 1992; Reid *et al.* 2002).

Breeding Indian yellow-nosed albatrosses largely remained within 1 500 kms of their colonies, foraging in pelagic subtropical waters (Pinaud *et al.* 2005; Pinaud and Weimerskirch 2007).

Breeding biology

The annual breeding cycle of Indian yellow-nosed albatrosses lasts eight months, beginning mid-August. Nests may be built in dispersed pairs or in large colonies. The egg is laid in September-October, and incubated for 78 days. Chicks hatch in November-December, and fledge 115 days later, between March 20 and April 17 (Jouventin *et al.* 1983; Weimerskirch *et al.* 1986). Breeding success averages 24.5% (range = 0–67%) at Amsterdam Island (Weimerskirch and Jouventin 1998). The breeding success at Pointe d'Entrecasteaux has only twice exceeded 20% in the years between 1990 and 2002. Avian cholera has been identified as the cause of this depressed productivity (Weimerskirch 2004).

Global population status

In 1998 the global population was an estimated 36 500 annual breeding pairs; this corresponds to roughly 160 000–180 000 individuals in total (Gales 1998). The Amsterdam Island population, which represents over 70% of the global population, has halved from 37 000 to 18 000 pairs between 1984 and 2003 (ACAP 2007c). In 2002 the population on Prince Edward Island, South Africa, was estimated with a high level of confidence at 7 500 pairs (ACAP 2007c). The population at St. Paul Island, was three pairs from 1993–2005. The current status of other populations of Indian yellow-nosed albatrosses remains unknown (ACAP 2007c).

3.2.14 Sooty Albatross *Phoebetria fusca* Hilsenberg 1822

Previous name

Sooty albatross *Phoebetria fusca*

Jurisdiction	Breeding locality
France	Amsterdam Island, Crozet Islands, Kerguelen Island, St. Paul Island
South Africa	Marion Island, Prince Edward Island
United Kingdom	Gough Island, Tristan da Cunha Island

Distribution

Post-breeding and non-breeding sooty albatrosses disperse widely between about 30°S and 60°S in the southern Atlantic and Indian Oceans, from Argentina east to the New South Wales coast (del Hoyo *et al.* 1993). They occur in small numbers from Western Australia across to Tasmania, particularly beyond the continental shelf (Reid *et al.* 2002) and are a vagrant to New South Wales and Queensland (Marchant and Higgins 1990). While breeding, some sooty albatrosses travel over 350 km from their colony to foraging grounds (Cooper and Klages 1995).

Breeding biology

Adults begin to return to their breeding grounds in mid-July to early September. The nests can be solitary or in loose association, with nesting density varying according to the steepness of the terrain. Most eggs are laid in October. The egg is incubated for 65–75 days, hatching in mid-December. The nestling period lasts from 145–178 days, with most chicks fledging in mid-May to early June (Berruti 1979; Weimerskirch *et al.* 1986, 1987). Breeding success at Possession Island ranged from 10% to 85% (1966–1995, mean 58%: Weimerskirch and Jouventin 1998). Most unsuccessful pairs (83%) attempt to breed in the following year (Jouventin and Weimerskirch 1984).

Immatures spend at least eight years foraging over subtropical seas before returning to their natal colony. The average age at first breeding is 12 years, with a minimum of nine years. Non-breeding immature birds often join the breeding colony in December, leaving again in June (Jouventin and Weimerskirch 1991).

Global population status

Gough Island (5 000 pairs), Marion and Prince Edward Islands (1 539 pairs) and the Crozet Islands (2 620 pairs) hold between 48%–73% of the total estimated breeding population of 12 500–19 000 pairs (BirdLife International 2007). The decrease in numbers of sooty albatrosses breeding on Gough Island (an estimated annual decrease of around 3% over 28 years) corresponds closely with decreases observed at other monitored breeding sites (Weimerskirch and Jouventin 1998; Nel *et al.* 2002; Crawford *et al.* 2003). Together, these results suggest that the global population of the species has decreased from 21 000–26 000 pairs in 1972 to around 10 000–14 000 pairs in 2000. This equates to a decline of around 74%–75% over three generations (90 years), justifying the recent upgrading of the conservation status of the sooty albatross to Endangered (Cuthbert and Sommer 2004).

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APPENDIX 1: LIST OF ABBREVIATIONS AND ACRONYMS

ACAP	Agreement on the Conservation of Albatrosses and Petrels
AAD	Australian Antarctic Division, Commonwealth Department of Sustainability, Environment, Water, Population and Communities
AAT	Australian Antarctic Territory
AFMA	Australian Fisheries Management Authority
AFZ	**Australian Fishing Zone**
ANARE	Australian National Antarctic Research Expeditions
ASAC	Antarctic Science Advisory Committee
CCAMLR	Convention on the Conservation of Antarctic Marine Living Resources
CCSBT	Convention for the Conservation of Southern Bluefin Tuna
CMS	Convention on the Conservation of Migratory Species of Wild Animals
COFI	FAO Committee on Fisheries
DPIPWE	Tasmanian Department of Primary Industries, Parks, Water and Environment
EEZ	Exclusive Economic Zone
EPBC Act	*Environment Protection and Biodiversity Conservation Act 1999*
FAO	Food and Agriculture Organisation of the United Nations
HIMI:	Heard Island and McDonald Islands
IAATO	International Association of Antarctica Tour Operators
IMAF	Incidental Mortality Arising from Fishing Working Group of CCAMLR
IOTC	Indian Ocean Tuna Commission
RFMO	Regional Fisheries Management Organisations
STWG	Status and Trends Working Group, ACAP
TAC	Total Allowable Catch
TAP	Threat Abatement Plan for the incidental catch (or bycatch) of seabirds during oceanic longine fishing operations
VMS	Vessel Monitoring System
WCPFC	Western and Central Pacific Fisheries Commission

APPENDIX 2: THE CRITICAL HABITAT OF SPECIES BREEDING WITHIN AREAS UNDER AUSTRALIAN JURISDICTION

Under the EPBC Act (1999: Section 207A), a Recovery Plan must identify the habitat that is regarded as critical to the survival of the threatened species. Albatrosses and giant petrels utilise two broad categories of habitat: breeding habitat (remote islands) and foraging habitat (southern oceans). This section describes the habitat that is critical to their survival within areas under Australian jurisdiction.

A2.1 Breeding Habitats

Albatrosses and giant petrels breed at only six localities under Australian jurisdiction. These are:

- Macquarie Island (including Bishop and Clerk Islets)
- Albatross Island
- Pedra Branca
- The Mewstone
- Heard and McDonald Islands
- Australian Antarctic Territory (Giganteus Island, Hawker Island and the Frazier Islands (Nelly Island, Dewart Island and Charlton Island)).

These remote islands constitute the only suitable breeding habitat under Australian jurisdiction and should be regarded as habitat that is critical to the survival of albatrosses and giant petrels in Australian waters. Shy albatrosses breed only within Australia, and hence the breeding habitats of this species (Albatross Island, Pedra Branca and the Mewstone) comprise its entire breeding habitat. Macquarie Island, Heard and McDonald Islands and the Australian Antarctic Territory (AAT) host several species of albatrosses and giant petrels. Many of these populations are very small and are critical for maintaining the genetic diversity necessary to ensure the viability of these species.

There are no other islands within areas under Australian jurisdiction that are considered to be potential or former breeding habitat for albatrosses or giant petrels.

A brief description of each albatross and giant petrel breeding location within Australian jurisdiction, including information on their protection status, geography, flora, fauna, and the effects of human occupation, is provided below.

A2.1.1 Macquarie Island (and Bishop and Clerk Islets): 54°18’S, 158°35’E

Species breeding on Macquarie Island (and Bishop and Clerk Islets)

- Wandering albatross
- Black-browed albatross
- Grey-headed albatross
- Light-mantled albatross
- Southern giant petrel
- Northern giant petrel

Protection status

- Designated a Biosphere Reserve by UNESCO in 1977
- Designated a Nature Reserve by DPIPWE in 1978
- Macquarie Island Management Plan implemented in 1991
- Designated a World Heritage Area in 1997

Geography

Lying in the Southern Ocean, 40 km from the Antarctic Convergence, subantarctic Macquarie Island is the exposed crest of the Macquarie Ridge. The island is 32 km long by 5 km wide at its broadest point and 12 785 ha in area. It rises abruptly from the ocean to form an undulating plateau, usually between 200-300 m above sea level, with a maximum altitude of 433 m. The north-western portion of the island is fringed by a raised beach terrace 15 m above sea level and up to 1 km wide.

Bishop and Clerk Islets lie 37 km to the south of Macquarie Island while Judge and Clerk Islets lie 14 km to the north. These small islands are poorly known because of difficulty of access. They are mostly barren rock less than 50 m high and are geologically similar to the main island (Selkirk *et al.* 1990).

Flora

There are no trees on Macquarie Island. However, there are 45 species of vascular plants as well as numerous moss and lichen species. These species are often associated to form one of five vegetation communities: feldmark, grasslands, herbfield, fen and bog.

Three plant species are endemic to Macquarie Island: the Cushion Plant *Azorella macquariensis*, an orchid *Corybas dienemus* and a salt tolerant species *Puccinellia macquariensis*. Five introduced plant species have become naturalised (Selkirk *et al.* 1990).

The cushion-like *Colobanthus muscoides* is the only vascular plant on Bishop and Clerk Islets (*Macquarie Island Nature Reserve and World Heritage Area Management Plan 2006*).

Fauna

Macquarie Island is inhabited by a large variety of wildlife. About 86 500 southern elephant seals *Mirounga leonina* and fur seals *Arctocephalus spp*, and around 3.5 million seabirds breed on the island (Selkirk *et al.* 1990). Seventy-two bird species have been recorded on Macquarie Island. Twenty seabird species breed on Macquarie Island, notably king penguins *Aptenodytes patagonicus*, royal penguins *Eudyptes schlegeli*, rockhopper penguins *E. chrysocome* and gentoo penguins *Pygoscelis papua*. Royal penguins are endemic to Macquarie Island. Over a million birds attend a rookery at Hurd Point at the southern end of the island during the breeding season (Selkirk *et al.* 1990).

Albatross and giant petrel breeding locations

Wandering albatrosses at Macquarie Island usually nest in moderately wind-exposed areas of the plateau edge up to an altitude of 250 m. Nests have been recorded along the western side of the island, extending around to the southern side at Petrel Peak and the northern side at Handspike Corner. Nests have also been recorded on the raised beach terrace areas, from the north-western corner to Aurora Cave.

Black-browed albatrosses breed in small numbers on South-West Point. Three small colonies and several solitary nests are located in this area. A larger population of black-browed albatrosses breed on Bishop and Clerk Islets (N. Brothers pers. comm.).

The grey-headed albatross breeding population is confined to the slopes on the southern side of Petrel Peak, West Rock and the slopes opposite West Rock. The majority of birds breed on the steep, tussocky southern slopes of Petrel Peak.

The light-mantled albatross has the largest breeding distribution of all the albatrosses on Macquarie Island. Nests are found at the northern end of the island around Bauer Bay, North Head and Sandy Bay. Nests are also found in the south around Caroline Cove, Hurd Point and Lusitania Bay.

Southern giant petrels tend to form breeding colonies on the coastal plateau or headlands, or on exposed flats, hillsides or ridge tops (Voisin 1988). Most of the adult birds roost communally on the coastal beaches and around lakes (Gales and Brothers 1996).

The northern giant petrels establish their solitary nests at low altitudes among dense tussock-grass on the coastal flats around the island (Gales and Brothers 1996).

Introduced species

Fifteen species of vertebrates have been introduced to Macquarie Island since its discovery. Six of these species are still present on Macquarie Island (Table A2.1). Five plant species have become established on Macquarie Island.

Table A2.1: Animal species introduced to Macquarie Island

Introduced species still present on Macquarie Island	Introduced species no longer present on Macquarie Island
European rabbit Oryctolagus cuniculus	Cat *Felis catus*
European starling *Sturnus vulgaris*	Cow *Bos taurus*
House mouse *Mus musculus*	Dog *Canis familaris*
Mallard *Anas platyrhynchos*	Donkey *Equus asinus*
Redpoll *Carduelis carduelis*	Goat *Capra hircus*
Ship's Rat *Rattus rattus*	Horse *Equus caballus*
	Pig *Sus scrofa*
	Sheep *Ovis aries*
	Weka *Gallirallus australis*

Source: *Macquarie Island Nature Reserve and World Heritage Area Management Plan 2006*

Effects of human occupation

Macquarie Island has a long history of human impact. Seal and penguin oil harvesters occupied the island from 1810 to 1920 (Cumpston 1968; Townrow 1988). Albatrosses and giant petrels were harvested for food throughout this time, particularly in the early years (Cumpston 1968).

Whilst the number of albatrosses and giant petrels taken by the early settlers is unknown, it is likely to have been excessive, given the degree to which other species were exploited. For example, sealers killed over 80 000 southern elephant seals within the first 20 years of occupation (Hindell and Burton 1988). In addition, fur seals had been completely eliminated from Macquarie Island following 25 years of exploitation. This species began to re-colonise the island in 1964 (Rounsevell and Brothers 1984).

Feral cats and rodents were recorded on the island by the 1820s and 1880s respectively. Recent pest control programs ensured cats were eradicated by 2002 (see Section 5.5), but rabbits and rodents are still present on the island. Wekas were introduced to Macquarie Island by the sealers as a source of food. These aggressive birds preyed upon penguin chicks, burrow-nesting petrels and invertebrates. An eradication program for wekas began in 1985, and ended when the last weka on the island was shot in 1988 (Copson 1995).

The introduction of rabbits to Macquarie Island in the 1870s has modified the distribution of vegetation alliances, particularly the grasslands (Rounsevell and Brothers 1984). Once rabbit control (myxomatosis) commenced in 1978, numbers declined from in excess of 150 000 to an estimated 3 300 animals (Parks and Wildlife Service 2006). This resulted in rapid recovery of most plant communities (Copson & Whinam 1998, 2001). However, from a low in the 1980s the rabbit population has increased to over 100 000 rabbits, which may be attributable to the combined effect of the eradication of cats, warmer drier weather and the possible reduced effectiveness of biological control methods (Parks & Wildlife Service 2006). Eradication measures are currently the only option for effective control of rabbits on Macquarie Island.

Few historical structures remain on the island. The modern station, located on the Isthmus, is comprised of over 40 buildings and structures for scientific and tourism purposes. There are also some field huts located elsewhere on the island (*Macquarie Island Nature Reserve and World Heritage Area Management Plan 2006).*

The *Macquarie Island Nature Reserve and World Heritage Area Management Plan 2006* provides guidelines preventing activities likely to impact upon wildlife on the island.

A2.1.2 Albatross Island: 40°23’S, 144°39’E

Species breeding on Albatross Island

- Shy albatross

Protection status

- Designated a Nature Reserve in 1981
- Albatross Island Management Plan is currently in preparation

Geography

Albatross Island is located in western Bass Strait, 30 km north of the north-west corner of Tasmania. The small island is only 1 100 m long, by 200 m wide, comprising an area of only 33 ha. The rocky island rises steeply from the surrounding sea to a height of about 35 m. A deep ‘gulch’ runs through the short axis of the island near its northern end (Green 1974).

Flora

Twenty-three plant species, including two small shrub species, have been found on the island.

Fauna

Albatross Island once contained a large population of fur seals before sealers exterminated the population. Fur seals now occasionally haul out on Albatross Island. Shy albatrosses, fairy prions *Pachyptila turtur*, little penguins *Eudyptula minor*, short-tailed shearwaters *Puffinus tenurostris* and silver gulls *Larus novaehollandiae* breed in large numbers on the island. Numerous other birds are occasionally seen on the island. In addition, at least two species of skinks (*Leiolopisma pretiosum* and *L. metallicum*) are found on the island (Green 1974).

Introduced species

Common starlings (*Sturnus vulgaris*) and European blackbirds (*Turdus merula)* have colonised the island.

Albatross breeding locations

The shy albatrosses nest on the top of the island. Colonies have formed in four areas: in the north-east, east, south-east and western edges of the island. These remnant colonies were formerly interconnected, except for the northern and southern sectors.

Effects of human occupation

The first European sighting of the shy albatross colony on Albatross Island was by George Bass in 1798. About 20 000 breeding pairs are thought to have nested on the island annually. By 1909, however, plume and egg hunters had decimated the colony to only 250-300 nests (Johnstone *et al.* 1975).

A2.1.3 The Mewstone: 43°44'S, 146°22'E

Species breeding on the Mewstone

- Shy albatross

Protection status

- Incorporated within the Southwest National Park.
- The Southwest National Park was designated a Biosphere Reserve in 1978.
- Incorporated within the Tasmanian World Heritage Area in 1989.

Geography

The Mewstone is located 22 km south of Tasmania. The tiny island is 450 m long and only 150 m wide, comprising 6.8 ha. The island rises precipitously from the sea to a height of 133 m. A ridge consisting of loose boulders and numerous rock crevices runs in a south-east direction. The only flat tracts on the island occur along the summit of the ridge. The steep sides of the ridge are occasionally interspersed with gently sloping ledges.

Flora

Only seven species of plants occur on the island (*Senecio leptocarpus, S. lautus, Carpobrotus rossii, Poa poiformis, Asplenium obtusatum, Chenopodium glaucum, Salicornia quineflora*). These small plants grow opportunistically in crevices or cavities where soil has accumulated (Brothers 1979a).

Fauna

Shy albatrosses and fairy prions nest on the island. Other birds recorded on the island include the common diving-petrel *Pelecanoides urinatix*, the black-faced shag *Leucocarbo fuscescens* and the silver gull. The Australian fur seal *Arctocephalus pusillus* occurs in moderate numbers, and a skink *Leiolopisma pretiosa* is abundant (Brothers 1979a).

Introduced species

None

Albatross breeding locations

Loose nesting colonies occur along the summit and on the rock ledges on both sides of the island. Some nests are located only 15 m above sea level, but most are at higher levels. Two-thirds of the nests are built on the western side of the island (Brothers 1979a). The rocky habitat ensures that the opportunities for entrapment in crevices are great, causing many albatrosses to die as a consequence.

Effects of human occupation

The tiny island has never been inhabited. In 1927 Lord reported that the Mewstone was "…swarming with birds. That albatrosses breed there we know." However, Lord also recounted stories that shy albatross eggs were taken for sale. The number of shy albatrosses destroyed in this manner is unknown. The island's remote location ensures that direct human interference is minimal (Brothers 1979a).

A2.1.4 Pedra Branca: 43°52'S, 146°58'E

Species breeding on Pedra Branca

- Shy albatross

Protection status

- Incorporated within the Southwest National Park.
- The Southwest National Park was designated a Biosphere Reserve in 1978.
- Incorporated within the Tasmanian World Heritage Area in 1989.

Geography

Pedra Branca lies 26 km south-southeast of Whale Head, the south-eastern extremity of Tasmania. Only 2.5 ha in area, the island is a mere 270 m long and 100 m wide. The island is essentially a rock mass emerging from the surrounding sea. The east and west slopes rise steeply to meet at a central ridge less than 60 m in height, running in a north-south direction.

Flora

Salicornia blackiana is the only plant species on the island. This species occurs sparsely and is confined to cracks among the rocks.

Fauna

Shy albatrosses, Australasian gannets *Morus serrator*, black-faced shags, silver gulls and fairy prions all breed on Pedra Branca. Australian fur seals inhabit the island, as does the endemic Pedra Branca skink *Pseudemoia palfreymani* (Brothers 1979b). The skink is regarded as endangered and a Recovery Plan has been prepared (Anon. 2001).

Introduced species

None

Albatross breeding locations

The main shy albatross colony is located on the south-eastern section of the island above 25 m above sea level where the sheer slope begins to level out making conditions suitable for nesting. Numbers gradually decrease northwards from the main colony (Brothers 1979b).

Effects of human occupation

The tiny island has never been inhabited. It is not known for certain whether humans exploited shy albatrosses on Pedra Branca in the past.

In 1938 S. Fowler visited Pedra Branca. Although he did not land on the island, he did indicate that an albatross colony existed there. In 1947, A.E. Palfreyman became the first recorded European to make a landing on Pedra Branca, however he made no record of the albatrosses. In October 1978, Brothers (1979a) landed on the island and located 97 active shy albatross nests.

Due to the island's remote location and the extreme difficulty of access, human interference is unlikely.

A2.1.5 Heard Island and the McDonald Islands: 53°04'S, 73°12'E

Species breeding on Heard Island and the McDonald Islands

- Black-browed albatross
- Light-mantled albatross
- Southern giant petrel

Protection status

- *Antarctic Marine Living Resources Conservation Act 1981*
- Listed on the Register of National Estate in 1983
- Environment Protection and Management Ordinance 1987
- Heard Island Wilderness Reserve Management Plan implemented in 1996
- Designated a World Heritage Area in 1997

Geography

The Territory of Heard Island and the McDonald Islands consists of a remote group of islands lying close together in the Indian Ocean sector of the Southern Ocean. The subantarctic island group lies south of the Antarctic Polar Front, over 4 100 km to the south-west of Fremantle, and 1 500 km north of Antarctica.

Heard Island is 20 km wide, 43 km long and has a total area of 368 km^2. It is dominated by Australia's tallest mountain (outside the Australian Antarctic Territory), 'Big Ben', an active volcano 2 745 m tall. To the north-west is a subsidiary volcanic cone, Anzac Peak (715 m). Glaciers cover eighty percent of the island. The remaining ice-free areas are mostly narrow coastal flats at the north-western and eastern ends of the island and along some northern beaches.

McDonald Island lies 43.5 km to the west of Heard Island. McDonald Island rises to 230 m above sea level, with a total area of about 2.5 km^2 and, over the last decade or so, has significantly increased in size due to volcanic activity. The McDonald group also includes the smaller Flat Island and Meyer Rock. All islands have high, cliff-lined coasts and rocky shoals.

Flora

The vegetation of the island group is typically subantarctic comprised predominantly of bryophytes, lichens, mosses, liverworts and tussock grasses. Eleven species of vascular plants are known to occur on Heard Island and five on the McDonald Islands. Six major higher plant communities dominate the islands: tussock grassland, meadow, herbfield, pool complex, cushion carpet and felfield. The islands are void of woody plants (Heard Island Wilderness Reserve Management Plan 1995).

Fauna

Southern giant petrels, black-browed albatrosses and light-mantled albatrosses breed on Heard and McDonald Islands. Fifteen other avian species nest on the islands. The Heard Island sheathbill *Chionis minor nasicornis* is a strongly defined subspecies endemic to the Heard and McDonald Island group. Four species of burrow-nesters breed in tens of thousands on Heard Island (Antarctic prions *Pachyptila desolata*, fulmar prions *P. crassirostris*, South Georgia (Islas Georgia del Sur) diving-petrels *Pelecanoides georgicus* and common diving-petrels). Other birds breeding in large numbers include cape petrels *Daption capense,* Wilson's storm-petrels *Oceanites oceanicus*, subantarctic Skuas *Catharacta lonnbergi* and kelp gulls *Larus dominicanus*.

Vast colonies of Macaroni penguin *Eudyptes chrysolophus* (with over one million breeding pairs) occur on both Heard Island and McDonald Island. There are also large numbers of southern rockhopper penguins, gentoo penguins, and king penguins.

Three seal species breed on the islands; namely the southern elephant seal, the Antarctic fur seal *Arctocephalus gazella* and the subantarctic fur seal *A. tropicalis*.

127 species of terrestrial invertebrates (many of which are endemics) have been found to occur on the islands (Heard Island Wilderness Reserve Management Plan 1995).

Introduced species

The introduced grass *Poa annua* is present on Heard Island, as well as several non native invertebrate species. It is thought that the *Poa* was introduced by natural processes, probably by skuas from Iles Kerguelen where it is widespread, because the grass was initially recorded in 1987 in two recently deglaciated areas where human visitation had not occurred (AAD 2005).

Albatross and giant petrel breeding locations

The ice-free areas of Heard Island are mostly confined to the narrow coastal flats at the north, north-western and eastern ends of the island. These are the principle breeding areas for light-mantled and black-browed albatrosses and southern giant petrels. There have been occasional sightings of wandering albatross on Heard Island. In 1980, a male wandering albatross (originally banded at Macquarie Island) was observed brooding a small chick at Cape Gazert. Two old nest mounds were also present nearby, suggesting that breeding had been attempted in previous years as well (Johnstone 1982).

Effects of human occupation

Heard Island was first sighted in 1833 and became the focus of a major sealing industry from 1855 to 1929. It is likely that the albatross and giant petrel populations were exploited for food throughout this period (Anon. 1996).

In 1947 Heard Island and the McDonald Islands were transferred from Britain to Australia. Australia used Heard Island as a meteorological base until 1954. There has since been no protracted stay on Heard Island other than ANARE scientific programs in the summers of 1985 – 1989, and an over-wintering expedition in 1992 (Anon. 1996).

The first recorded landing on the McDonald Islands occurred as recently as 1971. The only other landing on McDonald Island was in 1980 when a team of Australian scientists visited the islands (Anon. 1996).

Over the last decade or two, the number of tourists to Heard Island has increased but is at a very low level. An increase in uncontrolled tourist and scientific activities may reduce breeding success of seabirds on the islands, however all visits to the islands and the surrounding territorial seas require prior permission from the administering authority (the AAD), and visitors are not permitted to disturb seabirds or other animals.

Global warming is having a dramatic impact on the island group. Glaciers that were once at sea level have now retreated to above 1 600 m above sea level and vegetation and lagoons now exist where once there were sea-front, glacier snouts. The surrounding ocean has increased its mean temperature by 0.75° C since 1947 (Anon. 1996).

A2.1.6 Australian Antarctic Territory: Hawker Island 68°39’S, 77°52’E; Frazier Islands 66°14’S, 110°10’E; Giganteus Island 67°37’S, 62°33’E

Species breeding on the Australian Antarctic Territory

- Southern giant petrel

Protection status

- Antarctic Treaty (1961)
- Agreed Measures for the Conservation of Fauna and Flora (1964)
- Commission for the Conservation of Antarctic Marine Living Resources (1981)
- Madrid Protocol to the Antarctic Treaty on Environmental Protection (1991)

Geography

The Australian Antarctic Territory (AAT) covers 5 896 500 km^2, or 42 % of Antarctica. Less than 0.2 % of the continent is permanently ice-free. It is the driest, coldest and windiest continent on earth. It is also the highest continent on earth, with an average elevation of 2 300 m.

Flora

Over 500 species of algae have been found in continental Antarctica, along with 125 lichen species and 30 mosses.

Fauna

Millions of crabeater seals *Lobodon carcinophagus* and southern elephant seals breed around the rocky Antarctic coastline and offshore islands. Antarctic fur seals, Weddell seals *Leptonychotes wedellii* and leopard seals *Hydrurga leptonyx* also breed on Antarctica. Ten seabird species breed within the AAT; namely southern giant petrels, southern fulmars *Fulmarus glacialoides,* south polar skuas *Catharacta maccormicki*, Antarctic prions *Pachyptila desolata*, Adelie penguins *Pygoscelis adelie*, emperor penguins *Aptenodytes forsteri*, Antarctic petrels *Thalassoica antarctica*, cape petrels, snow petrels *Pagodroma nivea* and Wilson's storm-petrels (Soper 1994).

Introduced species

None.

Giant petrel breeding locations

Less than 0.2 % of the AAT is permanently ice-free, but it is these ice-free areas that are the principal breeding, roosting and moulting sites for southern giant petrels, as it is for all seabirds breeding within the AAT (Woehler 1993).

Southern giant petrels breed at only four sites around the coastline of Antarctica and on the Antarctic Peninsula. Three of these breeding sites are within the AAT; namely Giganteus Island, Hawker Island and the Frazier Islands. Colonies are established on open gravel areas and rocky outcrops, usually towards the periphery of the islands (Woehler *et al.* 1990).

Effects of human occupation

During the 1820/21 summer, two sealing masters working from the South Shetland Islands (discovered only two years prior) independently landed on the Antarctic Peninsula. By 1892, over 1 100 sealing ships had visited Antarctic regions (Headland 1993).

Australia's record of involvement with Antarctic exploration dates back to 1886 when the Australian Antarctic Committee was founded. The first research expedition to winter on the Antarctic continent occurred 12 years later. There are now 40 permanent scientific research stations in Antarctica, most of which are located on the Antarctic Peninsula. Australia has three permanent scientific research stations within the AAT; namely Mawson Station (near the Rookery Islands), Davis Station (near Hawker Island) and Casey Station (near the Frazier Islands).

The habitat loss and disturbance to nesting sites associated with construction and operations of research stations have directly affected at least two species, snow petrels and Wilson's storm-petrel. There are also data suggesting regular visits to colonies of Adelie penguins and southern giant petrels may disturb breeding birds, causing colonies to decrease (Woehler 1993), although this interpretation for some of these data for southern giant petrels is disputed (Wienecke *et al* 2009).