5 Direct impacts on EPBC Act protected species

5.1 Introduction

There are 241 species (see Appendix 3) protected under the *Environment Protection and Biodiversity Conservation Act 1999* (Cwlth) (EPBC Act) that occur in the area of the Small Pelagic Fishery (SPF). These are comprised of:

- 10 pinniped species
- 44 cetacean species
- Dugong *Dugong dugon*
- 89 species of seabirds
- six marine turtle species
- nine seasnake species
- 13 shark and ray species
- 69 teleost species, of which 66 are syngnathids and three are other teleost fish.

The data compiled by Tuck *et al.* (2013) have been used as the primary source to inform the panel's understanding of the nature and extent of the direct interactions of mid-water trawling in the SPF with protected species to date. Tuck *et al.* (2013) report on 'interactions' with protected species but do not define 'interaction'. Since the data were compiled from Australian Fisheries Management Authority (AFMA) logbooks and observer records the panel has assumed that the interactions data reported in Tuck *et al.* (2013) reflect the definition in the memorandum of understanding (MoU) between AFMA and the Department of the Environment. As noted in Section 2.2.3, this definition excludes acoustic disturbance and behavioural changes brought about by habituation to fishing operations, which the panel includes in its definition of 'direct interactions' applied to the assessment of the Declared Commercial Fishing Activity (DCFA). As a result Tuck *et al.* (2013) understate the level of 'direct interactions'. However, in the absence of any more comprehensive assessment of historical interactions data, the panel has used the information collated by Tuck *et al.* (2013) as an indicator of the nature and extent of direct interactions with protected species by previous mid-water trawl activity in the SPF.

The panel's Terms of Reference specified the need to assess the likely nature and extent of direct interactions of the DCFA with seals and dolphins. The panel formed the view that pinnipeds and cetacean species generally warranted detailed consideration. Within each of those groups the panel identified species of particular interest (Sections 5.2 and 5.3).

The panel noted that the Department of the Environment did not ascribe a high level of uncertainty to the impacts of the DCFA on seabirds (Box 1.1). This appears to have been based largely on the fact that there would be no discharge of biological material by the DCFA, the net would remain submerged during the pumping operation and bird mitigation measures were relatively well developed and tested. The panel generally concurs with this assessment. As a result, its assessment of the impact of any direct interactions with seabirds (Section 5.4) is less extensive and less species-specific than that for pinnipeds and cetaceans. However, the panel formed the view that the potential for ecosystem effects, of any potential localised depletion arising from the DCFA, on seabirds, particularly on central place foragers (CPF), required more detailed assessment (Chapter 6).

The panel considered the need to assess direct interactions between the DCFA and protected species of dugong, turtles, seasnakes, sharks and teleosts and formed the view that this was not necessary. The rationale for this decision is provided in Appendix 3.

5.2 Pinnipeds

5.2.1 Pinniped species assessed

There are three resident pinniped species that breed in coastal areas and islands off southern Australia. These are the Australian sea lion *Neophoca cinerea*, the New Zealand fur seal *Arctocephalus forsteri*, and the Australian fur seal *A. pusillus doriferus*. All species are native to Australia, occur within the SPF and occur in sympatry (overlap in ranges) over parts of their range (Kirkwood and Goldsworthy 2013) (Figure 5.1).

In addition to these resident species, a number of vagrant species visit southern Australia irregularly (Figure 5.2). The most common is the subantarctic fur seal *A. tropicalis*. Its nearest breeding colonies are located at subantarctic Macquarie Island (Southern Ocean/South Pacific Ocean) and Amsterdam/St Paul Islands (Southern Ocean/Southern Indian Ocean). Southern elephant seals *Mirounga leonina* are also regularly sighted in southern Australia, most commonly between September and March with animals coming ashore to moult. There are a number of breeding records in southern Australia, most notably in Tasmania. Prior to European arrival in Australia this species used to breed on King Island, Bass Strait, but was eliminated by sealers by the early 1800s. The nearest breeding sites are now at subantarctic Macquarie and Heard Islands. Another regular visitor to southern Australia is the leopard seal *Hydrurga leptonyx*. Although there are records of sightings of crabeater seal *Lobodon carcinophagus*, Weddell seal *Leptonychotes weddelli*, Ross seal *Ommatophoca rossii* and Antarctic fur seal *A. gazella* in southern Australia, they are uncommon relative to the other vagrant species (Kirkwood and Goldsworthy 2013). The panel recognises that interactions between these vagrant pinniped species and SPF fishing vessels is possible but unlikely because of their irregular occurrence. Therefore, this report focuses largely on the three key resident species. The panel recognises that potential impacts from the DCFA could apply to the other seven pinniped species as well. A summary of distribution and abundance throughout the SPF, status and trends, conservation status and foraging ecology of these key species is provided below.

Australian sea lion *Neophoca cinerea* (Level 2 Productivity Susceptibility Analysis (PSA) Residual Risk – Medium)

Distribution and range

The Australian sea lion (ASL) is endemic to Australia, and restricted to South Australia (SA) and Western Australia (WA). Its extant breeding range extends from The Pages Islands (just east of Kangaroo Island) in SA to Houtman Abrolhos on the west coast of WA (Shaughnessy *et al.* 2011). Pupping has been recorded at 81 sites (islands and at several mainland sites); 47 in SA and 34 in WA (Shaughnessy *et al.* 2011, Goldsworthy *et al.* 2013b, Goldsworthy unpublished data) (Figure 5.3). Despite the large number of breeding sites, only seven sites produce more than 100 pups per breeding season, all of which are in SA. The average pup production per breeding site is just 40, with most sites (70 per cent), producing fewer than 30 pups per breeding season (Goldsworthy *et al.* 2009a, Goldsworthy unpublished data).

The population can be broadly separated into three main metapopulations, one in SA accounting for approximately 84 per cent of pup production; one on the south coast of WA accounting for approximately 10 per cent of pup production; and one on the west coast of WA accounting for approximately 6 per cent of pup production (Goldsworthy *et al.* 2009a, Goldsworthy unpublished data). All west coast WA colonies fall north of the SPF boundary (31°S) (although the southernmost colony at Buller Island is only 38 kilometres (km) to the north). Therefore, about 94 per cent of the species population occurs adjacent to the SPF area. Another 151 locations have been identified as haul-out sites (90 in SA and 61 in WA), but because records of haul-out sites are based on opportunistic observations, the actual number is likely to be higher than this (Goldsworthy *et al.* 2009a).

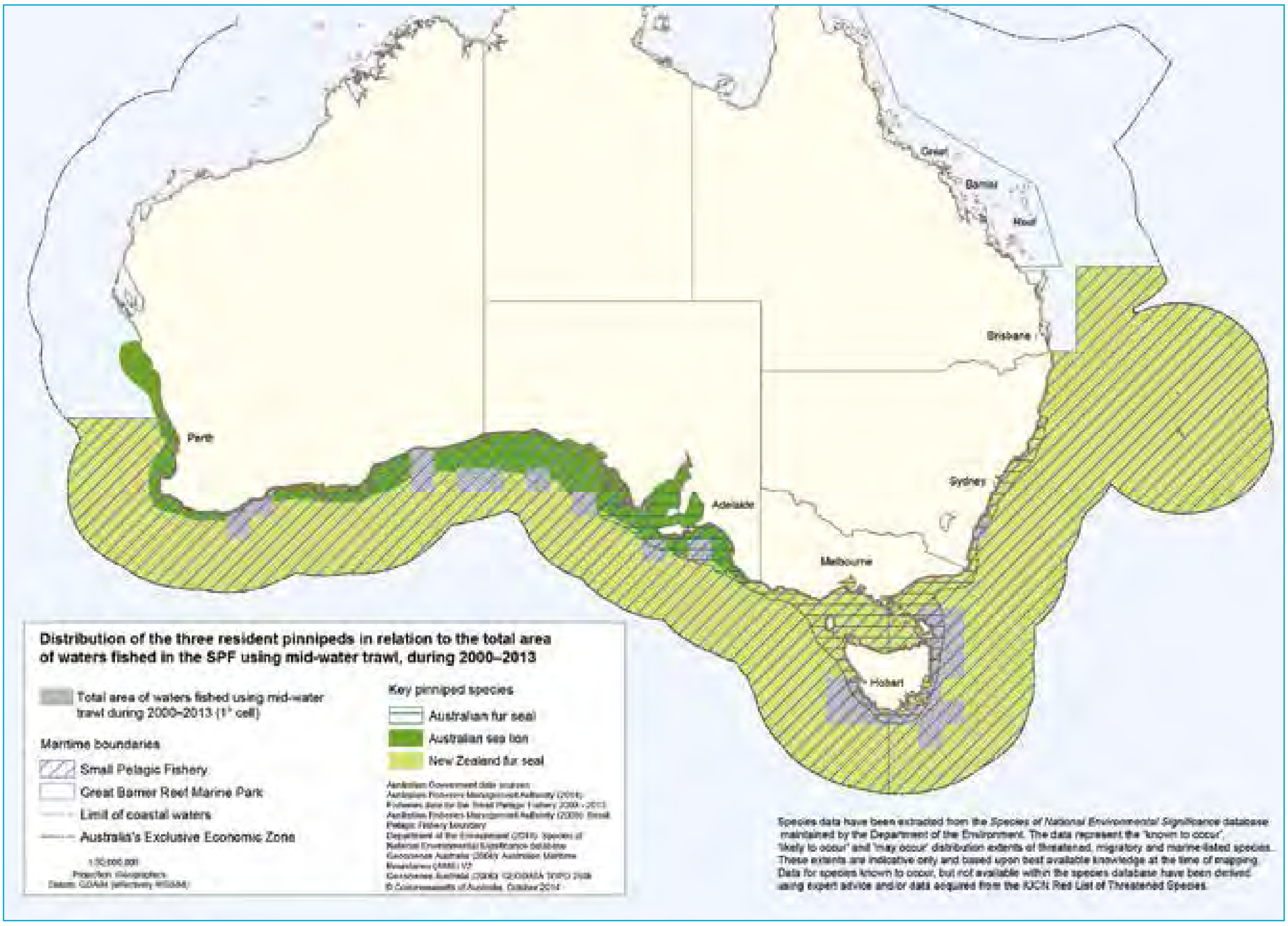

Figure 5.1 Distribution of the three resident pinnipeds in relation to the total area of waters fished in the SPF using mid-water trawl during 2000–2013. **Source:** Map produced by the Environmental Resources Information Network (ERIN), Department of the Environment using unpublished AFMA data.

Population size and trends

Total pup production is estimated to be 2691 in SA and 335 off the south coast of WA, and 182 off the west coast of WA (3208 in total) (Goldsworthy *et al.* 2009a, Shaughnessy *et al.* 2011, Goldsworthy *et al.* 2013b, Goldsworthy unpublished data, Goldsworthy in review). Pup production to total population multipliers developed for the species range from 3.83 to 4.08 (Goldsworthy and Page 2007, Goldsworthy *et al.* 2010) giving a total population estimate of approximately 12,690 (with a range of about 12,290–13,090), or approximately 12,000 (with a range of 11,590–12,350) adjacent to the SPF area.

ASL were subject to unregulated sealing in the late 18th and early 19th century (Ling 1999), resulting in a reduction in population size of unknown extent and extirpation of populations in Bass Strait and from many locations within their current range (Shaughnessy *et al.* 2011). The species has not recovered since harvesting ceased, unlike the two fur seal species in southern Australia that have undergone rapid recovery in recent years (Kirkwood *et al.* 2010, Shaughnessy *et al.* 2014).

The analysis of population trends requires consistent estimates or indices of pup production over a number of breeding seasons, and the non-annual and asynchronous breeding habits of the species have made collecting reliable time-series of pup abundance/production challenging (Goldsworthy *et al.* 2009a). Therefore, time series data from which trends in abundance can be estimated are limited (Goldsworthy *et al.* 2009a). The longest time-series data come from the three largest breeding colonies in SA, Seal Bay (Kangaroo Island), The Pages Islands (Backstairs Passage) and Dangerous Reef (Spencer Gulf). More recent time series have come from SA colonies along the Bunda Cliffs in the Great Australian Bight (GAB), at Olive Island (off Streaky Bay), Lilliput and Blefuscu Islands (Nuyts Archipelago) and two small colonies at Jones

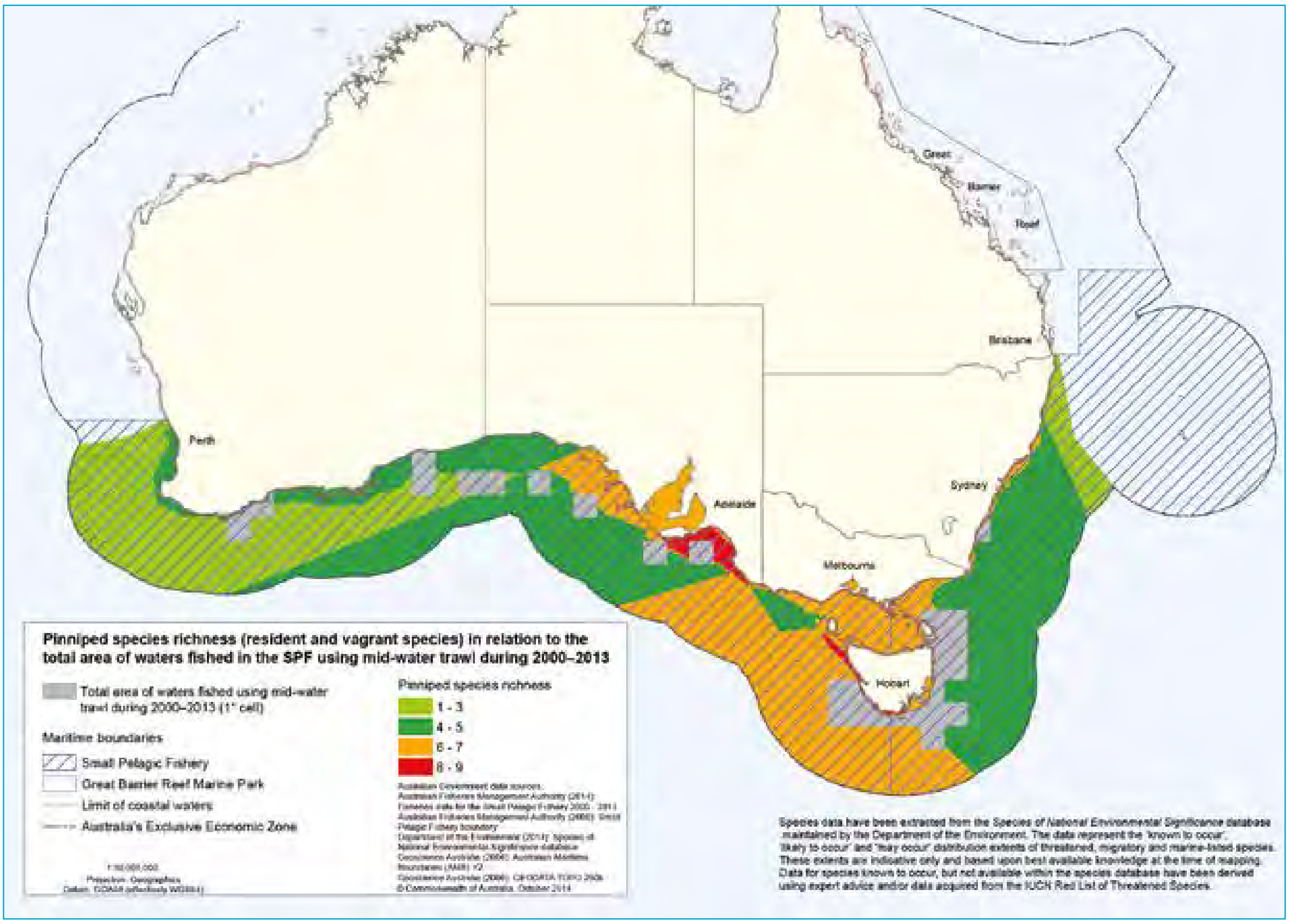

Figure 5.2 Pinniped species richness (resident and vagrant species) in relation to the total area of waters fished in the SPF using mid-water trawl during 2000–2013. **Source:** Map produced by ERIN using unpublished AFMA data.

Island (Baird Bay) and The Seal Slide (Kangaroo Island). In WA, ASL pup numbers have been surveyed in most breeding seasons since 1987 at three islands (Buller, North Fisherman and Beagle) on the central west coast of WA, but limited trend data are available for the breeding colonies off the south coast of WA (Goldsworthy in review).

Significant declines in pup numbers have been reported for Seal Bay (approximately 2 per cent decline per breeding season or about 32 per cent decline over 28 years; Goldsworthy *et al.* 2014a); colonies along the Bunda Cliffs (a 39 per cent decline in mean maximum number of pups counted per site over 19 years; Mackay *et al.* 2013); and at Olive Island (approximately 8 per cent decline per breeding season or 32 per cent decline over seven years; Goldsworthy unpublished data). Colonies that appear to be stable (no significant change in pup numbers) include The Pages Islands (surveys undertaken over 20 years; Shaughnessy *et al.* 2013); Lilliput and Blefuscu islands (data only available over five breeding seasons; Goldsworthy *et al.* 2013b); Jones Island and the Seal Slide (six and eight breeding seasons, respectively; Goldsworthy *et al.* 2013b, Goldsworthy *et al.* 2014a); and Buller, North Fisherman and Beagle islands off WA (Goldsworthy *et al.* 2009a). The only known breeding colony where pup numbers have increased is Dangerous Reef in Spencer Gulf. Here, pup numbers increased significantly between the mid-1990s and late-2000s, reaching a peak in 2006–07; since then pup numbers have declined (Goldsworthy *et al.* 2012, Goldsworthy *et al.* 2014c). It has been noted that the major period of increase in pup production at Dangerous Reef coincided with gillnet fishing effort in SA being reduced almost to zero following management changes in the fishery in 2001, which included closure of Spencer Gulf to the Commonwealth managed Gillnet Hook and Trap (GHAT) Fishery (Goldsworthy *et al.* 2007). During eight breeding seasons from 1994–95 to 2006–07 there was a significant negative relationship between gillnet fishing effort and pup abundance at Dangerous Reef (Goldsworthy and Page 2007, Goldsworthy *et al.* 2014c).

The most recent evaluation of the species' status and trends in abundance—using all available time series data on pup abundances from SA and WA subpopulations (23 subpopulations accounting for approximately 48 per cent of the species-wide pup production)—suggests the species' abundance has declined by almost 60 per cent in the past 40 years (Goldsworthy in review).

Biology and feeding ecology

ASL are unique among pinnipeds, being the only species that has a non-annual breeding cycle, with intervals between pupping seasons of approximately 17–18 months (Ling and Walker 1978, Higgins and Gass 1993, Shaughnessy *et al.* 2006, Goldsworthy *et al.* 2014a). All other pinnipeds have annual breeding seasons. Furthermore, breeding seasons are protracted in duration (six to nine months), and occur asynchronously across the species range (breeding can occur at any time of the year, Shaughnessy *et al.* 2006, Goldsworthy *et al.* 2014a). Asynchronous breeding is maintained through extremely low rates of interchange between colonies by adult females, as demonstrated by genetic studies that indicate extreme population sub-structuring of mitochondrial DNA lineages (maternally inherited), even for those separated by short distances (Campbell *et al.* 2008, Goldsworthy and Lowther 2010, Lowther *et al.* 2012). The evolutionary determinants of this unusual reproductive strategy remain enigmatic (Goldsworthy *et al.* 2009a). Pups are usually nursed for around 18 months, but this may be extended to three or more years if females do not pup in the subsequent breeding season or their new pup dies.

ASL restrict their foraging activities to continental shelf waters, with juveniles, adult females and adult males rarely exceeding depths of 90, 130 and 150 metres (m), respectively (Goldsworthy *et al.* 2010) (Figures 5.4 and 5.5). The maximum recorded dive depth for an adult male is approximately 250 m (Goldsworthy unpublished data). The maximum recorded foraging ranges of juvenile and adult female seals are 118 and 190 km, respectively (Goldsworthy *et al.* 2010). Adult males range much further and have been tracked up to 340 km from their colony. There is marked variability within and between-colonies in the foraging behaviour of juveniles, adult females and males (Goldsworthy *et al.* 2009b, Goldsworthy *et al.* 2010, Lowther and Goldsworthy 2011, Lowther *et al.* 2011). Foraging trips to sea are relatively short compared to other otariids (mean 1.1 days and maximum (max) 5.1 days in juveniles; mean 1.2 days and max. 6.2 days in adult females; mean of 2.5 days and max 6.7 days in adult males) (Kirkwood and Goldsworthy 2013). ASL are benthic foragers, they typically dive continuously while at sea and forage at all times of day. During dives they minimise the time spent during the descent and ascent phases in order to maximise foraging time on the seabed. Individual dives rarely exceed eight minutes in duration enabling animals to perform around 10 to 11 dives per hour (Kirkwood and Goldsworthy 2013).

Based on extensive satellite tracking studies, models of the spatial distribution of foraging effort are available for ASL populations in SA (Goldsworthy *et al.* 2003a, Goldsworthy and Page 2007, Goldsworthy *et al.* 2010) (Figure 5.5). Some have also been developed for WA populations based on limited data (Goldsworthy *et al.* 2003a, Campbell 2008, Hesp *et al.* 2012), although a recent study has provided significantly more satellite telemetry for the south coast WA populations (Goldsworthy *et al.* 2014b).

The diet of the ASL is poorly understood. Dietary information available is based on limited scat (faecal), digestive track (autopsied dead animals) and regurgitate analyses (Gales and Cheal 1992, Ling 1992, McIntosh *et al.* 2006), some crittercam footage (Fragnito 2013), and analyses of prey DNA recovered from faeces (Peters *et al.* 2014). Cephalopods appear to be a key component of the diet, and include octopus (Octopodidae), calamari (Loliginidae) and cuttlefish (Sepiidae) species. Key fish taxa include leatherjackets (Monacanthidae), wrasse (Labridae), flatheads (Platycephalidae), perch (Sebastidae, Serranidae), cods (Moridae), mullets (Mullidae), and nannygai/redfish (Berycidae), whiting (Siikginidae), rock-ling *Genypterus tigerinus*, stingaree/fiddler ray (Urolophidae, Rhinobatidae). Small pelagic fish including jack mackerel *Trachrus declivis*, yellowtail scad *T. novaezelandiae* and Australian sardine *Sardinops sagax* have been recorded in the diet, but are not common (McIntosh *et al.* 2006, Peters *et al.* 2014). Crustaceans have also been recorded in the diet and include crabs (stone crab), prawns, and rock lobster (*Jasus edwardsii*) (McIntosh *et al.* 2006, Fragnito 2013). Crittercam data indicate that diet and feeding behaviour can vary markedly between individual animals (Fragnito 2013).

Risks and threatening processes

A range of anthropogenic factors have been identified which may be impacting on the recovery of the ASL (Goldsworthy *et al.* 2009a, DSEWPaC 2013). The cumulative impact of many of these threats may vary across the range of the species. Fisheries bycatch (especially in gillnets) and entanglement in marine debris appear to pose the greatest threat to the Australia sea lion at present. Secondary threats include habitat degradation and interactions with aquaculture operations, human disturbance to colonies, deliberate killings, disease, pollution and oil spills, noise pollution, prey depletion and competition, and climate change (Goldsworthy *et al.* 2009a, DSEWPaC 2013).

Conservation and listing status

The ASL is listed as a threatened (Vulnerable) species under the EPBC Act; also listed as Marine (see Appendix 3). It is listed as a protected species (Rare) in SA under the *National Parks and Wildlife Act 1972*; in WA it is protected under section 14 of the *Wildlife Conservation Act 1950* and is listed as specially protected under the *Wildlife Conservation (Specially Protected) Fauna Notice 2005 (WA)*; and in Victoria the ASL is listed under the *Wildlife Act 1975* (protected wildlife; notable wildlife). Globally, the ASL is listed as Endangered under the International Union for Conservation of Nature and Natural Resources (IUCN) Red List (Goldsworthy and Gales 2008) and is listed in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES).

Summary: Australian sea lion

- *The Australian sea lion is an endemic and threatened species.*
- *Around 95 per cent of its range is adjacent to the SPF area.*
- *The most recent evaluation of the species' status and trends suggests it has declined by almost 60 per cent in the past 40 years.*
- *Interactions with fisheries are identified as a key risk.*
- *Populations are vulnerable to these interactions due to their small size, high metapopulation structure and complex breeding dynamics (non-annual/asynchronous breeding).*
- *Small pelagic fish appear to be uncommon in its diet.*
- *There is a risk of direct interactions with mid-water trawl fishing operations under the DCFA.*

Australian fur seal *Arctocephalus pusillus doriferus* (Level 2 PSA Residual Risk – High)

Distribution and range

There are two subspecies of the Afro-Australian fur seal *Arctocephalus pusillus*, the Cape or South African fur seal *Arctocephalus pusillus pusillus* and the Australian (or brown fur seal) *Arctocephalus pusillus doriferus*. The Australian subspecies was possibly derived as a consequence of late Pleistocene/Holocene (approximately 12,000 years before present) migration events from southern Africa to southern Australia via west-wind drift across the Indian Ocean (Wynen *et al.* 2001, Deméré *et al.* 2003). They are endemic to southeastern Australian waters and are found from the coast of New South Wales (NSW), Tasmania to Victoria and across to SA with the centre of their distribution in Bass Strait (Kirkwood *et al.* 2010). They have not been recorded in WA. There are 21 known breeding sites that include nine established colonies in Bass Strait, Lady Julia Percy Island, Seal Rocks, The Skerries, and Kanowna Island in Victoria; Judgment Rocks, Moriarty Rocks, Reid Rocks, West Moncoeur Island, and Tenth Island in Tasmania; eight colonies that have established in the past 10 to 15 years, which are Rag Island and Cape Bridgewater (Victoria), Wright and Double Rocks (Tasmania), Bull and Sloop rocks (Tasmania), Montague Island (NSW) and North Casuarina Island (SA); and three haul-outs, with accessional pupping at Iles des Phoques (Tasmania), Williams Island and Baudin Rocks (SA) (Kirkwood *et al.* 2010, Shaughnessy *et al.* 2010, McIntosh *et al.* 2014, Shaughnessy *et al.* 2014) (Figure 5.3). The range of the species is expanding, with the new colonies in NSW and SA all establishing in the past 10 years. Historical ranges prior to colonial sealing (pre-1800s) are unknown.

Population size and trends

Three national surveys of pup production for the species have been done at approximately five-yearly intervals since 2002–03. One undertaken in 2002–03 estimated a pup production of 19,820, another undertaken in 2007–08 estimated a pup production of 21,881, and the most recent survey undertaken in 2013–14 estimated a pup production of 15,063 (Kirkwood *et al.* 2005, Kirkwood *et al.* 2010, McIntosh *et al.* 2014). The rate of increase in pup production between 1986 and 2002–03 was estimated to be 5 per cent per year, slowing to 0.3 per cent per year between 2002–03 and 2007–08 seasons (McIntosh *et al.* 2014). It is not clear if the apparent 6 per cent per year decline between the 2007–08 and 2013–14 estimate is due to a poor pupping season in 2013–14 or represents a real decline in population over that period, as there is no colony that is monitored on an annual basis (McIntosh *et al.* 2014). Based on the 2007–08 surveys, two colonies adjacent to the Victorian coast, Seal Rocks (5660 pups) and Lady Julia Percy Island (5574 pups), account for more than half (51 per cent) the total pup production (Kirkwood *et al.* 2010). Based on these surveys the total Australian fur seal population is estimated to be 120,000 individuals (Kirkwood *et al.* 2010).

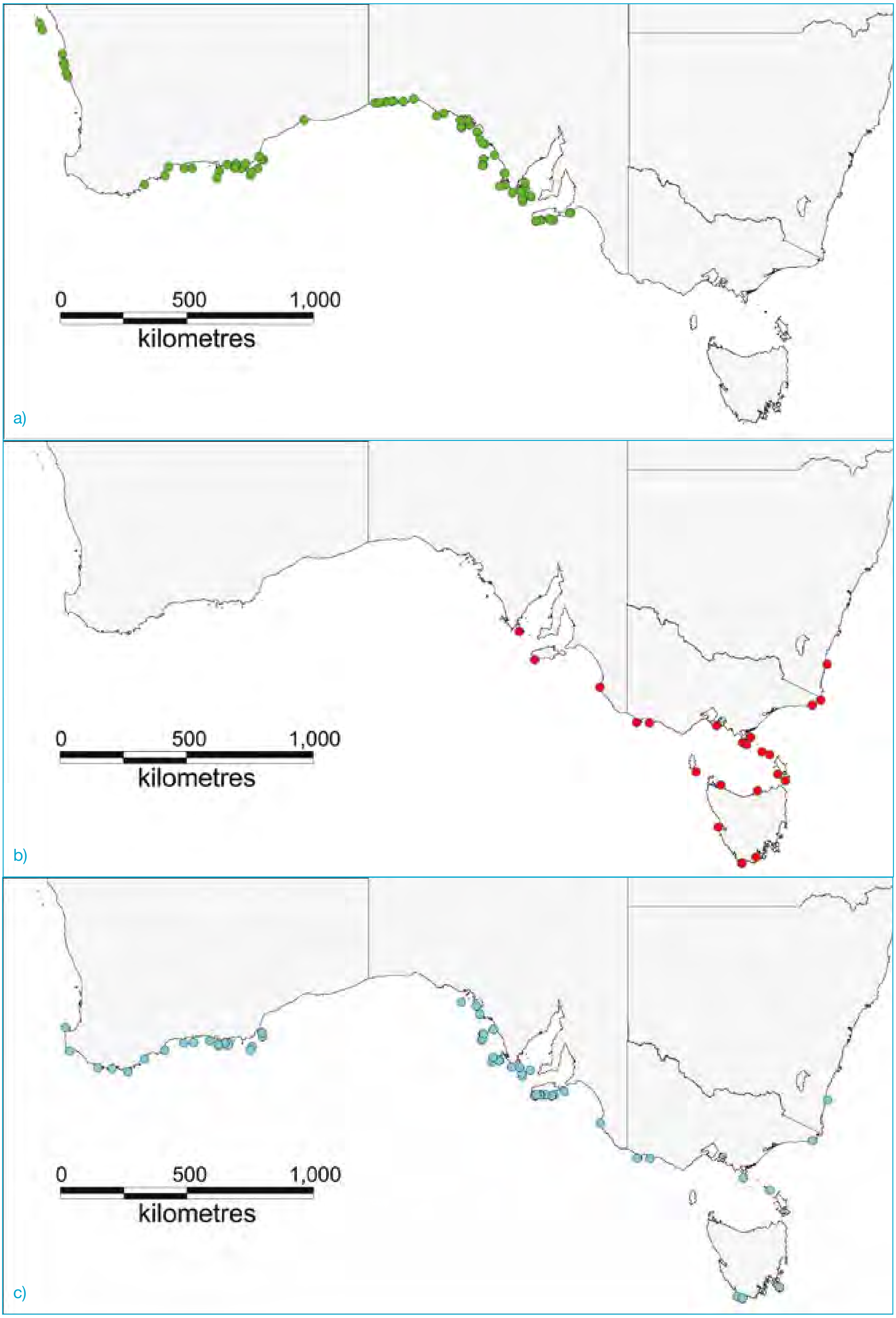

Figure 5.3 Location of known breeding sites for the Australian sea lion (a), Australian fur seal (b) and New Zealand fur seal (c) in Australian waters. **Source:** S. Goldsworthy South Australian Research and Development Institute (SARDI) unpublished data.

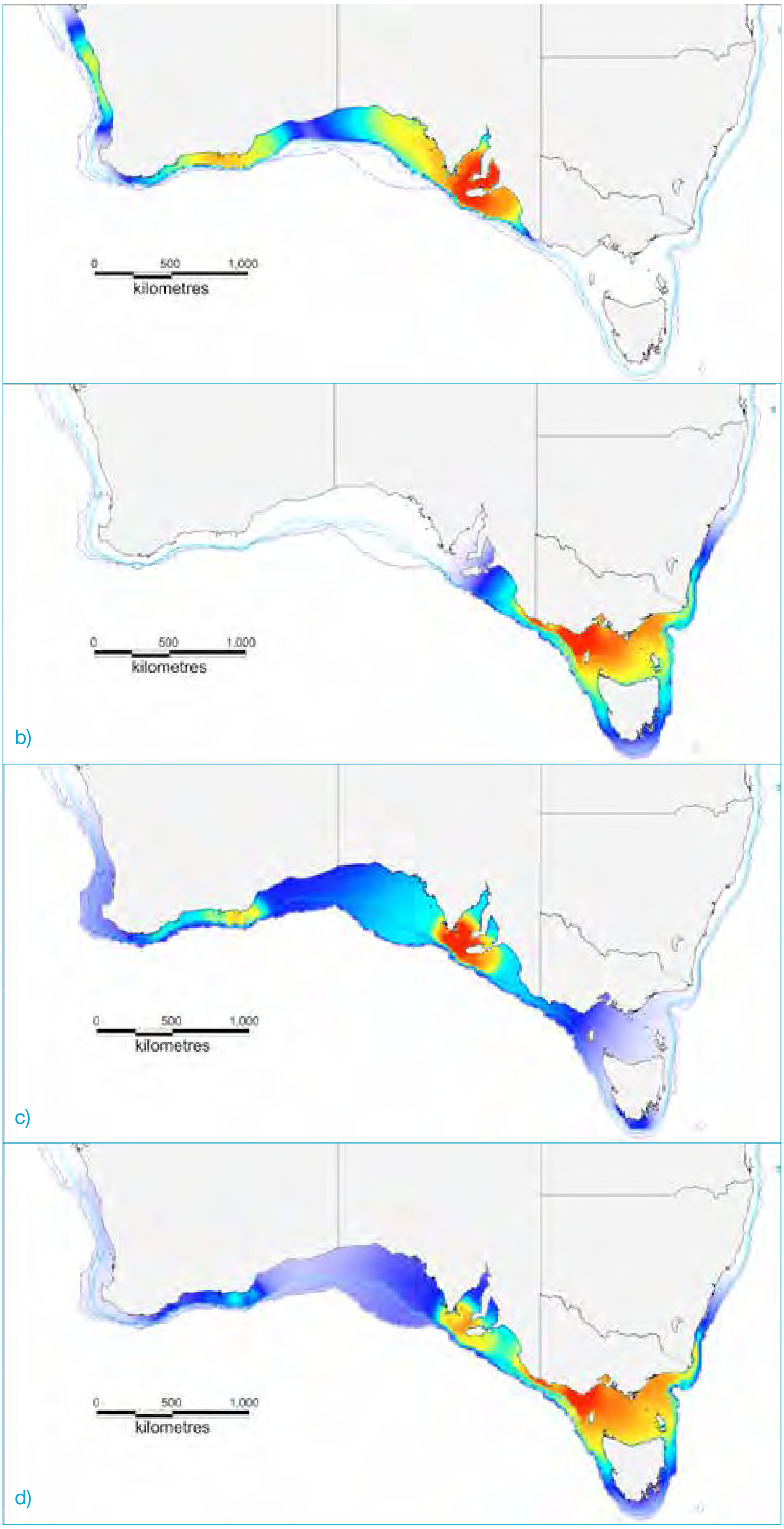

Figure 5.4 Heat plots representing the estimated spatial distribution of consumption effort by Australian sea lion (a), Australian fur seal (b) and New Zealand fur seal (c) populations, and all species combined (d). New Zealand fur seal estimates are only for consumption on shelf waters (oceanic consumption not modelled).

Source: S. Goldsworthy, SARDI, unpublished, redrawn from data presented in Goldsworthy *et al.* (2003a).

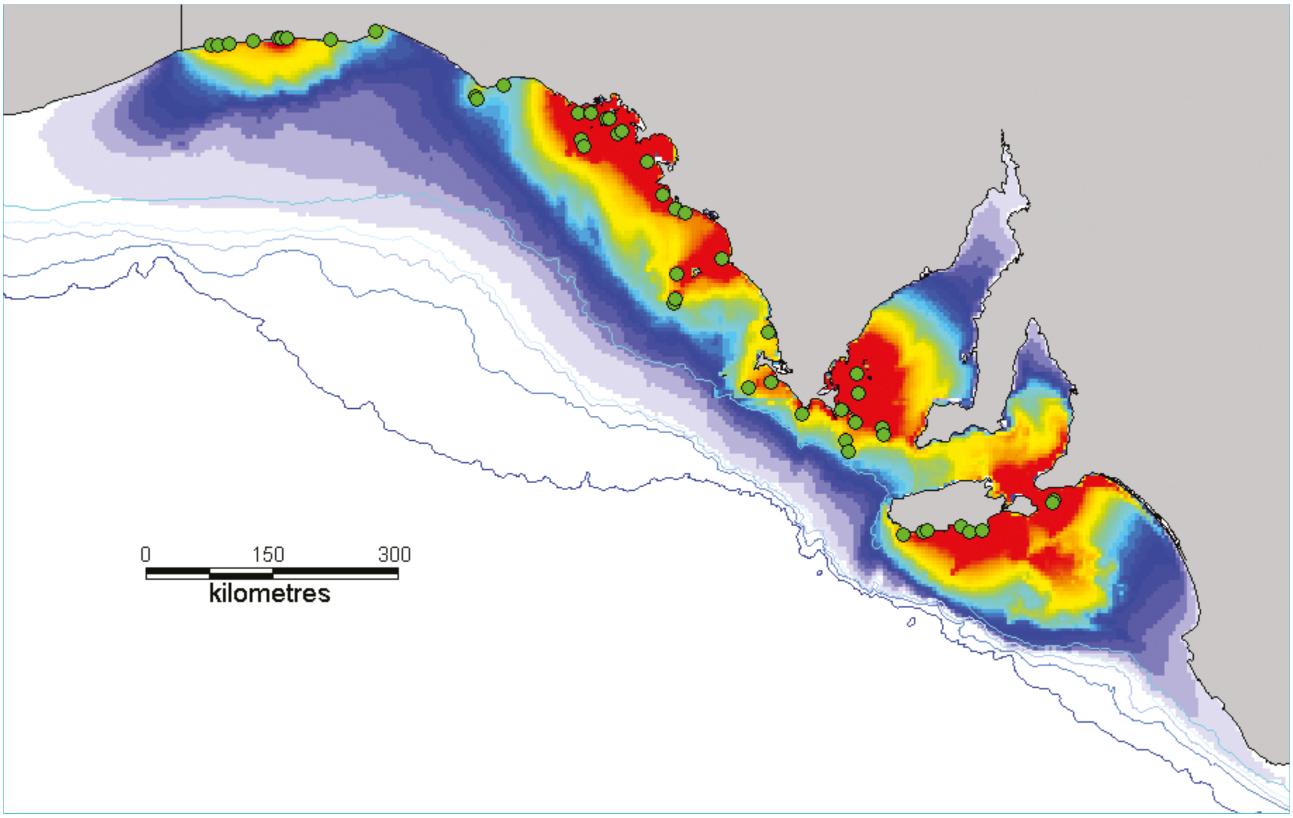

Figure 5.5 Model of the spatial distribution of foraging effort of the South Australian population of Australian sea lions including adult females, males and juveniles. The gradient from red to light blue colours indicates areas from highest to lowest foraging effort. Green dots indicate the location of breeding sites. Bathymetry lines are indicated from light to dark blue (100, 200, 500, 1000, 2000 m).

Source: Goldsworthy *et al.* (2010), reproduced with permission from SARDI – Aquatic Sciences, Simon Goldsworthy.

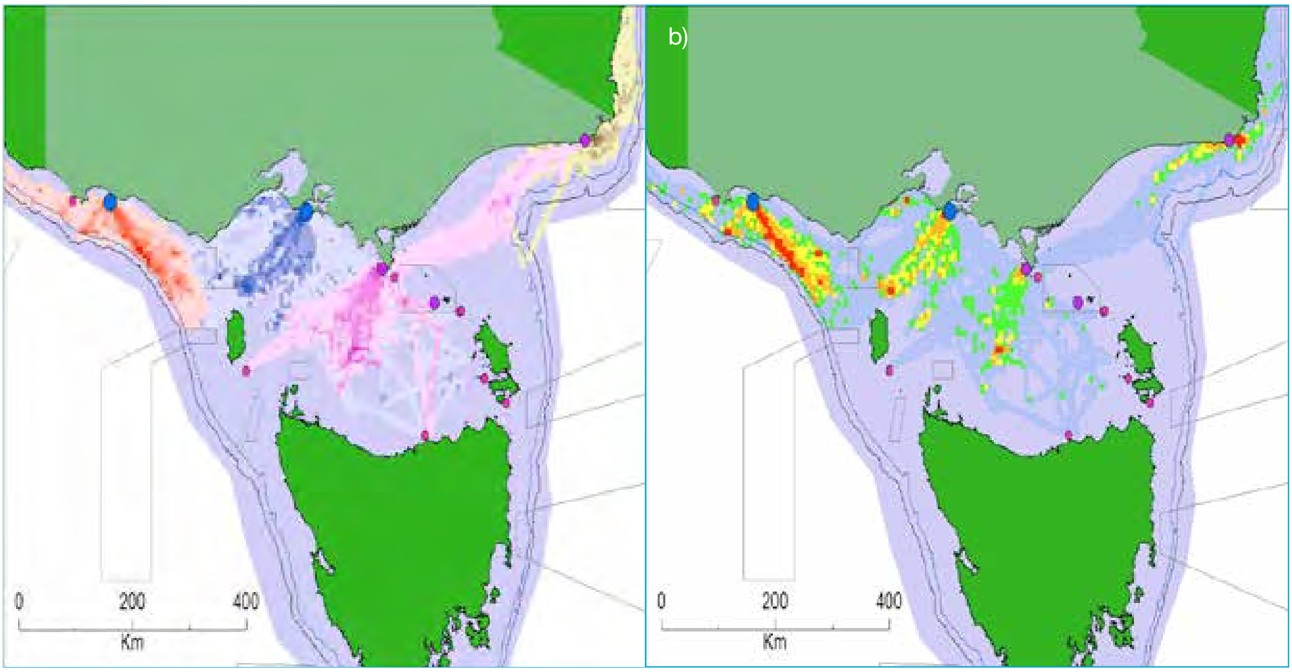

Figure 5.6 Habitat use, as time spent in 100 km^2 cells, by lactating Australian fur seals from the four main Bass Strait colonies: (a) by colony, plus 95 Kernel density polygons of locations at sea (red–Lady Julia Percy Island; blue–Seal Rocks; purple–Kanowna Island; brown–The Skerries), and (b) overall, proportional to numbers of live pups counted in 2002–03 (Kirkwood *et al.* 2005). Depth contours are 100, 200 and 1000 m, marine protected areas in the vicinity of where the females spent time at sea are included. Source: Kirkwood and Arnould (2011), reproduced with permission from CSIRO Publishing (http://www.publish.csiro.au/nid/90/paper/ZO11080.htm) and John Wiley and Sons Inc. Copyright 2005 by Society for Marine Mammalogy.

Biology and feeding ecology

Australian fur seals have an annual synchronous breeding season, with most pups born over a five-week period between early November and mid-December, with the peak in breeding usually in late November/early December (Kirkwood and Goldsworthy 2013). Most pups are weaned when they are 10–11 months old, just prior to the commencement of the next breeding season, although some may continue into a second year.

The Australian fur seal forages almost exclusively in association with the sea floor and rarely leaves the continental shelf, which reflects the benthic nature of their foraging (Arnould and Kirkwood 2008, Kirkwood and Arnould 2011, Kirkwood and Goldsworthy 2013) (Figure 5.6). Satellite tracking studies show that lactating adult females from the main breeding colony in eastern Bass Strait (The Skerries) travelled the shortest distance (20–60 km) while those in central Bass Strait (Seal Rocks, Kanowna Island) and western Bass Strait (Lady Julia Percy Island) typically forage out to 60 and 150 km from the colony (Arnould and Kirkwood 2008, Kirkwood and Arnould 2011). Foraging trip durations of lactating females last approximately six days, with most (greater than 90 per cent) time spent within 150 km of the colony (Kirkwood and Arnould 2011). Analysis of habitat use has indicated that individual seals selected areas with depths of 60–80 m, significantly more than other depths (Arnould and Kirkwood 2008). Females from colonies adjacent to productive shelf-edge waters (e.g. Lady Julia Percy Island and The Skerries) typically have shorter foraging trips, have smaller foraging ranges, forage closer to colonies and exhibit less diversity in foraging trip strategies than females from colonies more distant from the shelf-edge (e.g. Seal Rocks and Kanowna Island) (Kirkwood and Arnould 2011) (Figure 5.6). Females typically show strong fidelities to individual foraging hotspots (Arnould and Kirkwood 2008, Kirkwood and Arnould 2011).

Information on the movement of adult males comes mainly from animals satellite tracked from one colony (Seal Rock). Most foraged in western Bass Strait with many also travelling down the west coast of Tasmania to forage in southern Tasmanian waters, 500 km from Seal Rocks. One adult male travelled west of the Eyre Peninsula (SA), 1200 km from Seal Rocks (Kirkwood *et al.* 2007). A number of adult male Australian fur seals interacting with mid-water trawl gear on freezer vessels off the west coast of Tasmania in the winter blue grenadier *Macruronus novaezelandiae* fishery have also been satellite tracked (Tilzey *et al.* 2006). The tracked seals continually targeted the fishing operations, resting between foraging trips at haul-outs on Tasmania's west coast, until the fishing season ended. The seals then moved on to forage in southern Tasmania or Bass Strait (Tilzey *et al.* 2006). Juvenile Australian fur seals tracked from Lady Julia Percy Island and Seal Rocks display similar ranges to adult females (Kirkwood and Goldsworthy 2013).

The diet of Australian fur seals is reasonably well understood, with dietary studies having been undertaken across most of the species' range. In Bass Strait, southern Tasmania and SA they predominantly forage benthically but also eat a wide range of pelagic fish and cephalopod species (Goldsworthy *et al.* 2003b, Hume *et al.* 2004, Page *et al.* 2005a, Littnan *et al.* 2007, Kirkwood *et al.* 2008, Deagle *et al.* 2009). Key fish prey include redbait *Emmelichthys nitidus*, leatherjacket spp., jack mackerel, barracouta *Thyrsites atun*, red rock cod *Pseudophycis bachus* and flatheads. Cephalopods are also important prey with key species being Gould's squid (*Nototodarus gouldi)*, *Octopus* spp. and cuttlefish *Sepia apama* (Hume *et al.* 2004, Page *et al.* 2005a, Kirkwood *et al.* 2008). Most of the dietary studies have used analyses of prey hard parts recovered from faecal (scat) samples, a method that can both under and over-represent prey species. One study analysed faecal DNA from samples collected at the three main Victorian colonies (Lady Julia Percy Island, Seal Rock, The Skerries). The study confirmed, based on the prevalence of sequences from redbait and jack mackerel, the importance of these species in the seals' diet. However, blue mackerel *Scomber australasicus* was also found to be important, suggesting hard-part analyses methods may have under-represented the importance of this species in the diet (Deagle *et al.* 2009).

Kirkwood *et al.* (2008) analysed annual variation in the diet of Australian fur seals at Seal Rocks over a nine-year period (1997–2006). The importance in the diet of redbait and jack mackerel varied considerably across the period, prevalent in some years, and near absent in others when it was replaced by increased proportions of barracouta, red cod and leatherjackets (Figure 5.7). Statistical analyses indicated that annual variation in redbait prevalence in the diet was significantly related to changes in mean sea surface temperatures in western Bass Strait where the seals foraged (Kirkwood *et al.* 2008). Redbait were most prevalent in the diet in cooler years and were less important in warmer years. They found no correlation between the prevalence of redbait in the diet with fishing effort (annual fisheries catch-per-unit-effort) nor the annual mean Southern Oscillation Index (Kirkwood *et al.* 2008).

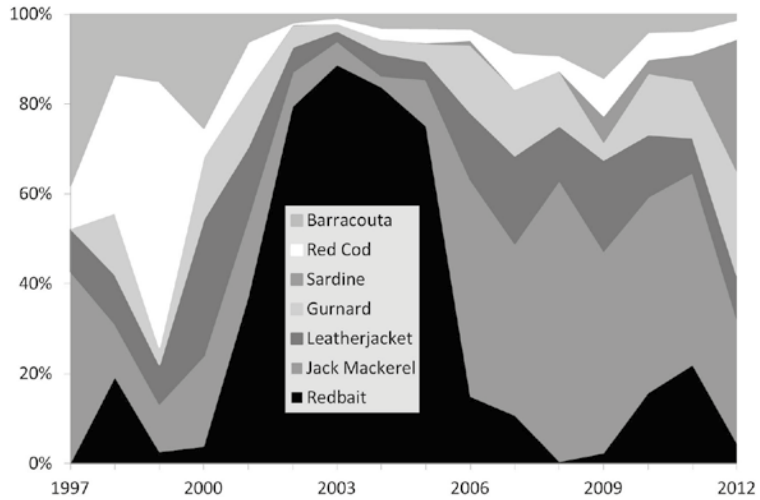

Figure 5.7 Annual variation in the diet of Australian fur seals at Seal Rocks (Victoria) based on prey hard-part analyses for scats collected at a nearly bimonthly frequency over nine years (1997–2012). Note the importance and variability of redbait and jack mackerel in the diet. **Source:** Kirkwood and Goldsworthy (2013), reproduced with permission from CSIRO Publishing (http://www.publish.csiro.au/pid/6491.htm).

Risks and threatening processes

Given that the foraging distributions of the Australian fur seal overlap extensively with commercial fishing activities, especially trawl fisheries operating in southeastern Australia (Goldsworthy *et al.* 2003b), fisheries interactions constitute the most significant risks and threatening processes to the species (Shaughnessy 1999, National Seal Strategy Group and Stewardson 2007). Australian fur seals are subject to significant and ongoing bycatch mortality associated with demersal and mid-water trawling operations, and they have constituted a significant bycatch in the mid-water trawl sector of the SPF (Knuckey *et al.* 2002, Hamer and Goldsworthy 2006, Tilzey *et al.* 2006, Lyle and Willcox 2008, Tuck *et al.* 2013). Indirect interactions, such as prey depletion from fishing, also pose a potential threat, especially in the SPF given its significant reliance on redbait, jack mackerel and blue mackerel (Goldsworthy *et al.* 2003a, Deagle *et al.* 2009) Details on fishery interactions are addressed in Section 5.2.2.

Australian fur seals interact regularly with finfish (salmon) aquaculture farms in Tasmania, where they enter net enclosures killing and damaging fish (Pemberton and Shaughnessy 1993, Hume *et al.* 2002, National Seal Strategy Group and Stewardson 2007, Robinson *et al.* 2008a). Seals are at risk of becoming entangled in nets and having their behaviour changed by becoming habituated to a predictable food source (National Seal Strategy Group and Stewardson 2007). A seal trapping and relocation program has operated since 1990, with more than 4500 individual relocations having taken place up to 2005. More than half (56 per cent) being repeat captures of previously trapped seals, with seals readily returning to the farms in southern Tasmania after release (Hume *et al.* 2002, Robinson *et al.* 2008a, b).

Other potential risks and threats to Australian fur seals include entanglement in marine debris, oil spills and disease (Shaughnessy 1999, Lynch *et al.* 2011a, Lynch *et al.* 2011b, Kirkwood and Goldsworthy 2013).

Conservation and listing status

The Australian fur seal is listed as Marine under the EPBC Act (see Appendix 3). It is protected under the Victorian *Wildlife Act, 1975* (protected wildlife; notable wildlife). And in Tasmania it is listed under Wildlife Regulations, 1999 (Schedule 1); the *Threatened Species Protection Act 1995*; and the *Nature Conservation Act 2002* (specially protected wildlife). In NSW it is protected under the *Threatened Species Protection Act 1995* (Vulnerable); and in SA under the *National Parks and Wildlife Act 1972* (Protected; Rare). Globally the species is listed as least concern under the IUCN Red List and is listed in Appendix II of CITES.

Summary: Australian fur seal

- *Australian fur seal distribution is restricted to the southeastern part of the SPF.*
- *Although the core part of its range (established colonies) may be relatively stable, the range of the species is still expanding and numbers are increasing in newly colonised areas.*
- *Its population has steadily increased over the past 30 years.*
- *Small pelagic fish (e.g. redbait and jack mackerel) are a key component of its diet.*
- *It readily interacts with a range of fisheries, particularly trawl fisheries.*
- *There is a risk of direct interactions with mid-water trawl fishing operations under the DCFA.*

New Zealand (Long-nosed) fur seal *Arctocephalus forsteri* (Level 2 PSA Residual Risk – Medium)

Distribution and range

The New Zealand (or long-nosed) fur seal is a native mammal of Australia that occurs in both New Zealand and Australian waters. Other common names include the black fur seal, Australasian fur seal, Antipodean fur seal and South Australian fur seal. The species was subject to heavy exploitation by colonial sealers between 1800 and 1830, resulting in major reductions in range and abundance (Kirkwood and Goldsworthy 2013). Numbers remained at very low levels for almost 140 years, after which they slowly began to build up and new colonies were established across their former range. In Australia, New Zealand fur seals occur in the coastal waters and on the offshore islands of South and Western Australia, from just east of Kangaroo Island, west to the south-west corner of the continent in WA, and also in southern Tasmania (Shaughnessy *et al.* 1994) (Figure 5.1). Small populations have recently been establishing in Bass Strait and Victorian and southern NSW coastal waters (Kirkwood and Goldsworthy 2013). In New Zealand, this species occurs around both the North and South Islands, with newly formed breeding colonies now established on the North Island and established and predominantly expanding breeding colonies around the entire South Island (Boren *et al.* 2006, Bouma *et al.* 2008). There are well established and expanding colonies also found on Stewart Island and all of New Zealand's subantarctic islands. Their range extends to Australia's Macquarie Island. Vagrants have been recorded in New Caledonia (Shaughnessy 1999).

The Australian population is centred off SA where more than 80 per cent of the national population occurs, with key breeding sites at Kangaroo Island, the Neptune Islands and Liguanea Island (Shaughnessy *et al.* 2014). Western Australian colonies are centred on the islands of the Recherche Archipelago with the westernmost population near Cape Leeuwin. In Tasmania, the New Zealand fur seal mainly occurs on the west and south coasts with a small number breeding on remote islands off the south coast.

Population size and trends

There are 65 known breeding sites for the species in Australia, most (86 per cent) are in South and Western Australia (SA 36; WA 20; Tasmania four; Victoria four; NSW one) (McIntosh *et al.* 2014, Shaughnessy *et al.* 2014, Campbell *et al.* in press, Department of Primary Industries, Parks, Water and Environment (DPIPWE) unpublished data). Pup production surveys were undertaken over the 2013–14 breeding season in SA, Victoria, Tasmania and NSW, and in the 2011–12 season in WA, which provide a comprehensive and current assessment of the status of the species' Australian population. In SA, total pup production was estimated to be 20,426, with most (10,133 pups) on Kangaroo Island (the largest colony in the Cape Gantheaume Wilderness Protection Area having 5333 pups); and the Neptune Islands and Liguanea Island off the southern Eyre Peninsula (9711 pups) (Goldsworthy *et al.* 2014a, Shaughnessy *et al.* 2014). Western Australian surveys estimated a total pup production of 3518 on breeding sites off the southern coast (Campbell *et al.* in press). Pup production in Victoria was estimated to be 276 pups; in Tasmania 399 pups and in NSW (Montague Island) 36 pups (McIntosh *et al.* 2014, DPIPWE unpublished data). The maximum pup production for the Australian population based on these surveys is 24,656 (about 25,000), with most pup production in SA (83 per cent) and WA (14 per cent). Based on a pup-to-total-population multiplier of 4.76 (developed by Goldsworthy and Page 2007) the Australian population is currently estimated to number approximately 117,400.

Populations of New Zealand fur seals in Australian waters appeared to begin their major recovery in the 1970s and 1980s. Between the 1989–90 and 2013–14 breeding seasons, the fur seal population in SA has increased 3.6 fold, with the average annual increase in pup production being 5.3 per cent (Shaughnessy *et al.* 2014). Recovery rates at some sites have been much greater. For example, in the Cape Gantheaume Wilderness Protection Area on Kangaroo Island, annual monitoring of pup production over a 26 year period from 1988–89 (457 pups) to 2013–14 (5333 pups), demonstrates a remarkable 11.7-fold increase at an average rate of 10 per cent per year (Goldsworthy *et al.* 2014c). In contrast, pup production at the Neptune and Liguanea islands appears to have peaked in the mid-2000s, with most of the available breeding habitat now full (Shaughnessy *et al.* 2014). The centre of population expansion is now on Kangaroo Island. The growth of New Zealand fur seal populations since the 1970s and 1980s in Australia is attributable to recovery from 19th century sealing (1800–1830) and subsequent take (Shaughnessy *et al.* 2014).

Biology and feeding ecology

New Zealand fur seals have an annual synchronous breeding season, with most pups (90 per cent) being born over a five-week period between late November and early January. On Kangaroo Island the breeding season peaks around 25–26 December (Goldsworthy and Shaughnessy 1994). New Zealand fur seal pups weigh 3-4 kilograms (kg) at birth, double their weight quickly in 60–100 days and wean at around 13–16 kg when about 10 months old (Goldsworthy 2006). Lactating females alternate between shore bouts lasting approximately 1.7 days in duration (when pups are nursed) and foraging trips to sea which increase in duration from about three to five days early in lactation, to eight to 11 days late in lactation (Goldsworthy 2006). However, foraging trips lasting more than 20 days are not uncommon (Goldsworthy 2006).

The core of Australia's New Zealand fur seal breeding distribution in SA is distributed across a relatively small geographic range characterised by narrow shelves in proximity to localised seasonal upwelling in summer and autumn (Figure 5.3). Satellite tracking studies show that early in lactation (December to March), females undertake short foraging trips to mid-outer shelf waters (70–90 km from the colony), in regions associated with localised upwelling (Page *et al.* 2006, Baylis *et al.* 2008a) (Figure 5.8). However, between April to May most females switch to foraging in distant oceanic waters associated with the Subtropical Front (STF), 700–1000 km to the south of breeding colonies, and continue foraging in these waters up until the weaning of their pups in September/October (Baylis *et al.* 2008a, Baylis *et al.* 2008b, Baylis *et al.* 2012) (Figure 5.9). These winter foraging trips last between 15 and 25 days. Once weaned, the pups head for oceanic waters south of Australia, and as juveniles, also forage in distant oceanic waters (mean maximum distance of 1095 km from the colony) (Baylis *et al.* 2005, Page *et al.* 2006)(Figure 5.5). In contrast to juveniles and adult females, adult males focus their forage efforts along the continental slope (Page *et al.* 2006).

New Zealand fur seals forage both on the shelf, where they target pelagic and bentho-pelagic prey, and off the shelf, where they target epipelagic prey that exhibit daily, vertical migrations (Kirkwood and Goldsworthy 2013). Adults can therefore forage both near or on the benthos in water depths ranging up to 200 m, and in the water column where the sea-floor might be less than 20 m or greater than 2000 m (Kirkwood and Goldsworthy 2013). The mean dive depth of adult female and male New Zealand fur seals are 41.5 m (maximum 312 m) and 52.1 m (to greater than 380 m), respectively (Page *et al.* 2005b). Mean dive durations are 2.7 minutes (maximum 9.3 minutes) for adult females and 3.6 minutes (maximum 14.8 minutes) for adult males (Page *et al.* 2005b).

Most information on the diet of New Zealand fur seals in Australia comes from studies undertaken in SA. As these seals forage both benthically and pelagically, on or off the shelf, their diet is broad. When foraging in shelf water, the main prey species include redbait, leatherjackets, western gemfish *Rexea solandri* and Gould's squid, while the main prey in the open ocean are lanternfish (Family Myctophidae) and Southern Ocean arrow squid *Todarodes filippovae* (Page *et al.* 2005b). Other important prey include jack mackerel, barracouta, Australian anchovy *Engraulis australis*, southern sea garfish *Hyporhamphus melanochir*, swallowtail *Centroberyx lineatus* and calamari squid *Sepioteuthis australis* (Page *et al.* 2005b). The diets of adult males, adult females and juveniles differ, mainly in relation to the extent to which they foraged on or off the shelf. Adult males tended to consume larger prey and were more likely than juveniles or females to consume birds (mostly little penguins *Eudyptula minor* and short-tailed shearwaters *Ardenna tenuirostris*) (Page *et al.* 2005b).

Risks and threatening processes

As the foraging distributions of the New Zealand fur seal overlap extensively with commercial fishing operations on Australian shelf waters, fisheries interactions constitute the most significant risks and threatening processes to the species (Shaughnessy 1999, National Seal Strategy Group and Stewardson 2007). Trawl and other fisheries are a source of entanglement and drowning for New Zealand fur seals (Page *et al.* 2004). It is likely that New Zealand fur seals make part of the bycatch of seals in the South East Trawl Fishery (part of the Commonwealth Trawl Sector (CTS) of the Southern and Eastern Scalefish and Shark Fishery (SESSF), but are not readily distinguished from Australian fur seals (Goldsworthy *et al.* 1997). Like all fur seals, New Zealand fur seals are vulnerable to oil spills because of their dependence on their thick pelage for thermoregulation (Gales 1991). They share most of their range with several other regularly occurring pinniped species and are at risk from transmission of infectious diseases such as morbilliviruses, brucellosis, leptospirosis and tuberculosis (MacKereth *et al.* 2005).

Conservation and listing status

The New Zealand fur seal is listed as Marine under the EPBC Act (see Appendix 3). In SA they are listed as Vulnerable under the *National Parks and Wildlife Act 1972*; in WA they are protected under the *Wildlife Conservation Act 1950* (protected, specially protected); in Victoria under the *Wildlife Act 1975* (protected wildlife, notable wildlife); in NSW under the *Threatened Species Protection Act 1995* (Vulnerable) and in Tasmania under the Wildlife Regulations 1999 (Schedule 1), *Threatened Species Protection Act, 1995, and Nature Conservation Act 2002* (specially protected wildlife, rare). Globally, they are listed as Least Concern under the IUCN Red List, and are listed in Appendix II of CITES.

Summary: New Zealand fur seal

- *The New Zealand fur seal is distributed throughout the entire SPF, but its core distribution in Australia is centred off South Australia.*
- *Although the core part of its range (established colonies) may be relatively stable, the range of the species is still expanding and numbers are increasing in recently colonised areas.*
- *Its population has steadily increased over the past 30 years.*
- *Small pelagic fish such as redbait and jack mackerel are important in its diet, as are squid.*
- *They readily interact with fisheries, including trawl fisheries.*
- *There is a risk of direct interactions with mid-water trawl fishing operations under the DCFA.*

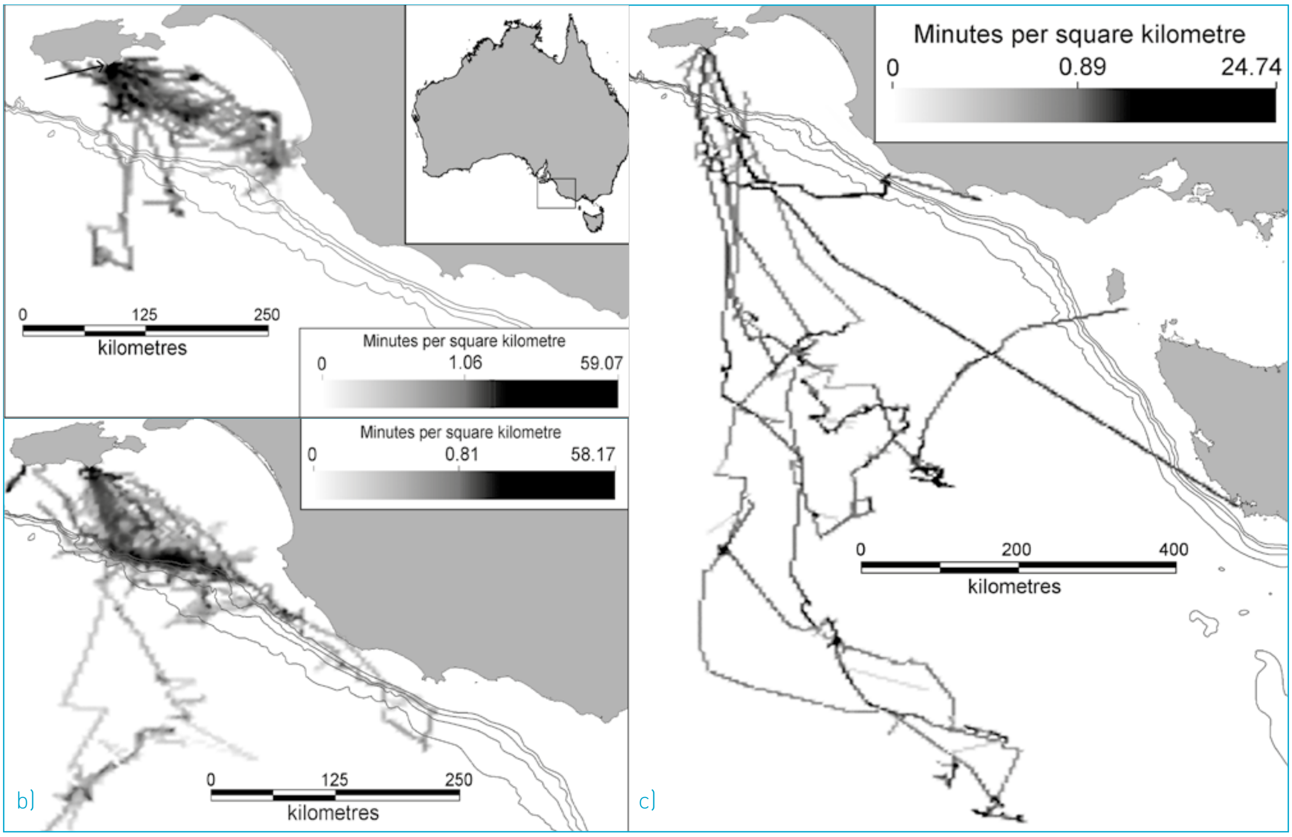

Figure 5.8 Time spent per 25 km^2 cells by (A) lactating female (n = 25), (B) adult male (n = 21) and (C) juvenile (n = 6) New Zealand fur seals, satellite-tracked from Cape Gantheaume on Kangaroo Island (SA). Location of Cape Gantheaume in relation to the continental shelf, shelf break (200, 500, 1000 and 2000 m depth contours) and pelagic waters (south of the shelf break) is shown. **Source:** Page *et al.* (2006), reproduced with permission from Marine Ecology Progress Series.

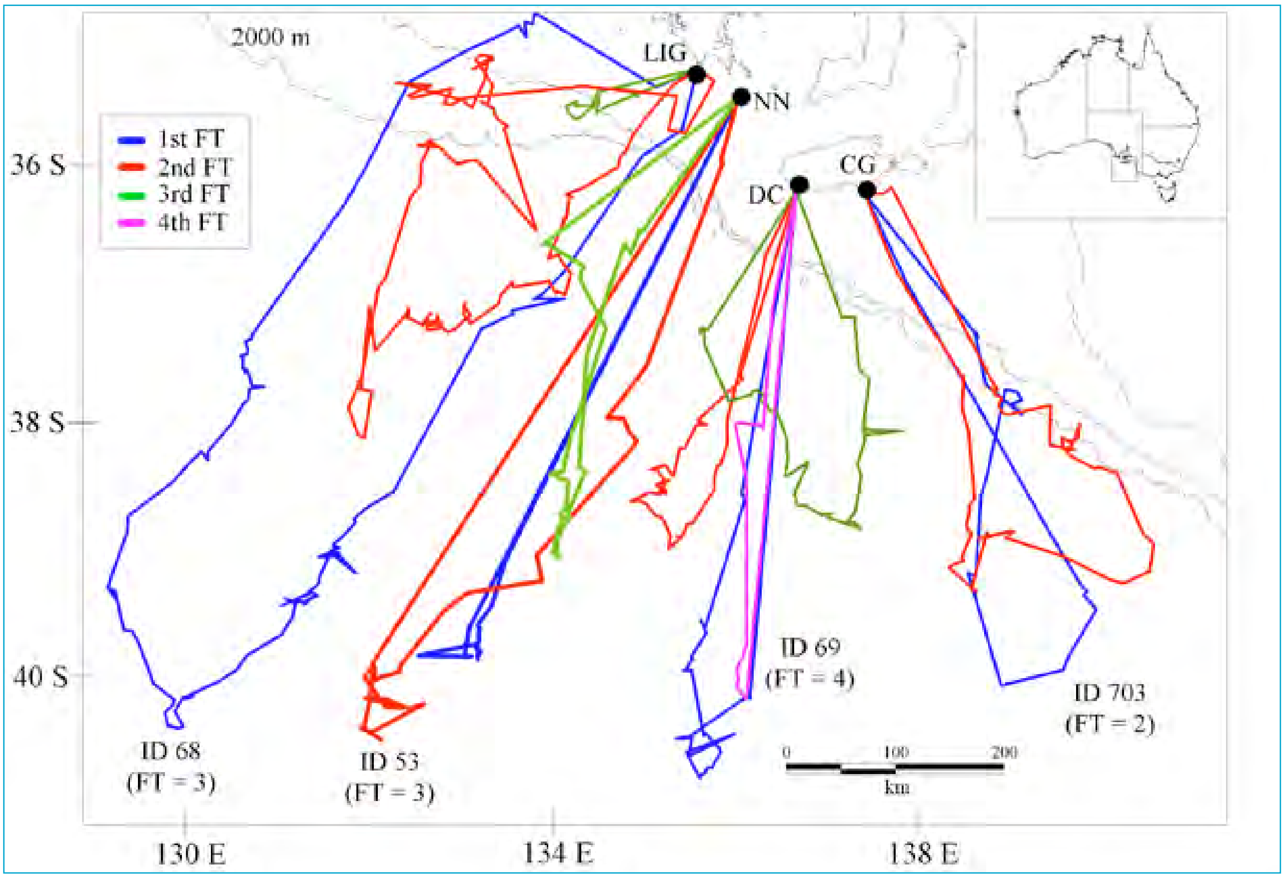

Figure 5.9 Examples of consecutive foraging trips undertaken by satellite-tracked lactating New Zealand fur seal females in oceanic waters typical of winter foraging from the four key breeding sites in SA: Cape Gantheaume, CG; Cape du Couedic, DC; North Neptune Island, NN; and Liguanea Island, LIG.

Source: Baylis *et al.* (2012), reproduced with permission from John Wiley & Sons Inc. © 2011 by the Society for Marine Mammalogy

5.2.2 Nature and extent of interactions

As detailed in Chapter 2, for the purpose of this assessment, direct interactions between the DCFA and protected species include net feeding, physical contact, acoustic disturbance, behavioural change and bycatch. For pinnipeds, acoustic disturbance is unlikely to be a significant issue (Carretta and Barlow 2011, Goetz and Janik 2013); however, net feeding, behavioural change and physical contact all contribute to bycatch interactions. Such interactions are generally not random or chance events, but occur as a direct consequence of animals deliberately interacting with fishing operations. Bycatch usually occurs as a consequence of 'net-feeding' where animals enter the net during fishing operations to feed on fish concentrated near the codend or enmeshed in the net ('stickers'). Animals that become trapped in the net while the net is being shot, or when actively fishing, will drown (the maximum dive duration of most otariid seals is less than 10 minutes); animals that become trapped in the net when it is hauled may survive until the net is retrieved, requiring release onboard the vessel (Hamer and Goldsworthy 2006, Tilzey *et al.* 2006). In general, the extent of bycatch interactions is largely a function of opportunity and the degree to which fishing operations reward seals for risky behaviour. If opportunities persist and fishing activity is predictable, then habituation of individuals or a population to fisheries interactions can result. As the number of fisheries interactions increases, so does the potential for bycatch.

Extent of trawl fishery interactions: global

Marine mammals and commercial fisheries often target the same food resource, leading to 'operational interactions' between animals and fisheries when they come into direct contact with fishing gear. Globally, the bycatch of marine mammals in fisheries is estimated to be in the hundreds of thousands of individuals per year (Read *et al.* 2006), and currently represents the dominant, recognised threat to global pinniped populations (Kovacs *et al.* 2012). Pinnipeds are readily attracted to, and interact, with trawl fisheries; they will take fish floating free from the net, stickers protruding through the net mesh, enter trawl nets to feed on fish inside the net and take discarded fish and offal (Wickens and Sims 1994, David and Wickens 2003).

Operational interactions with trawl fisheries that lead to significant levels of pinniped bycatch have been reported in most parts of the world where pinniped populations overlap with trawl fisheries. Documentation for these in many instances is limited to short-term studies where interaction rates have been reported and analysed based on independent fishery observer programs. Examples are given below by region.

South Africa

High interaction rates have been reported to occur between Cape fur seals and South African trawl fisheries (offshore demersal, inshore demersal and mid-water fisheries) where annual bycatch numbers ranged between 2524 and 3636 (Wickens and Sims 1994, David and Wickens 2003). Mortality levels were much higher in mid-water trawls (94 seals per 100 trawls), compared to inshore (4.6 seals per 100 trawls) and offshore demersal trawls (1.2 seals per 100 trawls) (Wickens and Sims 1994). This was thought to be due to a combination of factors including the wider opening of mid-water trawl nets, slower retrieval, lower buoyancy and tendency to trawl until the net reaches the vessel, which create opportunities for more seals to interact and be drowned (Wickens and Sims 1994).

South America

There is limited documentation on the level of pinniped interactions with trawl fisheries in South America. Significant bycatch of South American sea lion *Otaria flavescens* has been recorded in a small subset of observed trawls off south-central Chile conducted in September 2004, when 82 animals were caught in 69 observed trawls (1.2 seals per trawl, Reyes *et al.* 2013). In northern and central Patagonia, Argentina, based on observations from 1992 to 1994, between 175 and 602 sea lions were estimated to have been caught, mostly by factory/freezer mid-water and demersal trawl vessels (Crespo *et al.* 1997, Dans *et al.* 2003).

Antarctica

The commercial krill *Euphausia superba* trawl fishery in Antarctic waters began in the early 1970s and the prospect of a free-for-all fishery for Antarctic krill led to the signing of the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR) in 1981. Discussions on the level of Antarctic fur seal mortality associated with the krill trawl fishery first took place at the 2003 meeting of CCAMLR's Working Group on Incidental Mortality Associated with Fishing (Reid and Grilly 2014 cited in Elgin Associates unpublished (b)). Limited information is available on this interaction. In 2004, data provided to CCAMLR by the United Kingdom, as part of the CCAMLR Scheme of International Scientific Observation, indicated that 292 fur seals were caught during krill fishery trawl operations in CCAMLR Subarea 48.3 in the 2003–04 season (Reid and Grilly 2014 cited in Elgin Associates unpublished (b)).

USA

Foreign and joint venture trawl fisheries operating in the Gulf of Alaska and Bering Sea between 1966 and 1988 were estimated to have killed more than 21,000 Steller sea lions *Eumetopias jubatus* (Perez and Loughlin 1991). A particularly high level of bycatch mortality occurred in 1982 in the Gulf of Alaska (most from the Shelikof Strait walleye pollock joint venture fishery) when an estimated 1530 sea lions were killed (Perez and Loughlin 1991). Average bycatch mortality of Steller sea lions was estimated to be approximately 730 per year in the late 1960s, about 1300 per year in the 1970s, then declining to approximately 530 per year in the 1980s and declining further to between 10 and 15 per year between 1990 and 2011 (Perez and Loughlin 1991, Perez 2003, Breiwick 2013). During the 1970s and 1980s, catch rates of Steller sea lions were highest for large mid-water trawl freezer vessels targeting pollock, and lowest for small stern trawlers (Perez and Loughlin 1991). Declines in the number of Steller sea lions taken as bycatch in the 1980s were principally due to reduced fishing effort and declines in the sea lion populations (Perez and Loughlin 1991). Major declines from 1990 onwards have been attributed to spatial closures introduced around all Steller sea lion colonies in the Aleutian Island and Bering Sea in 1990 (Perez 2003). Although the Steller sea lion have been numerically the most significant pinniped subjected to bycatch mortality in the Alaskan fisheries, other pinnipeds including the northern fur seal *Callorhinus ursinus*, bearded seal *Erignathus barbatus nauticus*, harbor seal *Phoca vitulina richardsi*, ribbon seal *Histriophoca fasciata*, ringed seal *Phoca hispida hispida*, spotted seal *Phoca largha*, northern elephant seal *Mirounga angustirostris* and walrus *Odobenus rosmarus*, are also caught in Alaskan trawl fisheries (typically less than 10 each year per species, Perez 2003, Allen and Angliss 2013).

Elsewhere in the USA, pinnipeds have been identified as bycatch in the Pacific groundfish fishery (demersal trawl) operating in the North Pacific Ocean off the Washington, Oregon and Californian coasts (Carretta *et al.* 2013). Between 2004 and 2008, average annual bycatch of pinnipeds has been approximately 35 California sea lions *Zalophus californianus californianus*, around six harbor seals, approximately six Steller sea lions (Eastern Stock) (Allen and Angliss 2013, Carretta *et al.* 2013). There is also incidental bycatch of northern elephant seals (approximately one per year), Guadalupe fur seals *Arctocephalus townsendi*, and northern fur seals (Carretta *et al.* 2013). Off the Atlantic coast, grey seals *Halichoerus grypus grypus*, harp seals *Pagophilus groenlandicus* and harbor seals *Phoca vitulina concolor* are incidentally caught in the mid-Atlantic bottom and mid-water trawl fisheries; low numbers of harbor seals are taken as bycatch in the Northeast mid-water trawl fishery (approximately one per year); and low numbers of grey seals (about six each year), harp seal and harbor seal (approximately one per year) are taken as bycatch in the Northeast bottom trawl fishery (Waring *et al.* 2013).

As a global generalisation, phocid seals are most susceptible to bycatch in gillnet fisheries, whereas otariid seals are most susceptible to bycatch in trawl fisheries (Waring *et al.* 2013).

New Zealand

In New Zealand, bycatch in commercial trawl fisheries includes the New Zealand fur seal and the New Zealand sea lion *Phocarctos hookeri*. Southern elephant seals *Mirounga leonina* and leopard seals *Hydrurga leptonyx* are also caught occasionally (Thompson *et al.* 2013). A recent study by Thomson *et al.* (2013), has estimated the annual bycatch of New Zealand fur seals and New Zealand sea lions in New Zealand trawl fisheries between the 2002–03 and 2010–11, and 1995–96 and 2010–11 fishing seasons, respectively. Bycatch of New Zealand fur seals occurs in the hoki *Macruronus novaezelandiae*, southern blue whiting *Micromesistius australis*, middle depths, squid trawl, ling, hake, mackerel, scampi, deepwater and inshore trawl fisheries (Thompson *et al.* 2013). Fur seal bycatch across all trawl fisheries averages 775 per year (the maximum was 1471 in 2004–05), but has declined over the nine seasons by about 55 per cent, with an estimated bycatch of 376 seals in 2010–11 (Thompson *et al.* 2013). Bycatch rates average 0.72 seals per 100 tows, and has declined by approximately 32 per cent over the period to 0.44 seals per 100 tows in 2010–11 (Thompson *et al.* 2013). Declines in bycatch numbers correspond with an approximate 34 per cent reduction in fishing effort over the study period (Thompson *et al.* 2013). Fishing effort is greatest in July and August, with fur seal bycatch peaking in August, but also high in July and September (Thompson *et al.* 2013). The Bounty Islands and subantarctic areas had the highest bycatch rates, and distance from shore was negatively correlated with bycatch rate. Coastal areas (less than 25 km from shore) had 1.6 times the bycatch rate of areas fished between 25 and 90 km from shore, and areas fished greater than 180 km from shore had bycatch rates that were 20 per cent of those between 25 and 90 km from shore (Thompson *et al.* 2013).

New Zealand sea lions are taken as bycatch in a number of New Zealand subantarctic trawl fisheries, in the Auckland Islands (squid, scampi, non-squid/scampi) and Campbell Island southern blue whiting and Stewart-Snares shelf fisheries (Wilkinson *et al.* 2003, Thompson *et al.* 2013). The majority of bycatch mortalities recorded have been from the Auckland Island squid fishery (within management area SQU6T), where the distribution of New Zealand sea lion foraging overlaps significantly with the distribution of fishing effort (Chilvers 2008, Chilvers *et al.* 2011, Thompson *et al.* 2013). Between 1995 and 1999, approximately 100 sea lions per year were being caught as bycatch in all New Zealand trawl fisheries (Thompson *et al.* 2013). This has subsequently declined to approximately 55 per year between 2000 and 2004, about 39 per year between 2005 and 2009, and 29 in the 2010–11 fishing season (Thompson *et al.* 2013). Between 1995 and 2010, sea lion bycatch declined by approximately 80 per cent (Thompson *et al.* 2013). Part of the decline can be attributed to a 37 per cent reduction in fishing effort, but also a range of management actions have been introduced in an attempt to mitigate bycatch in these fisheries (detailed in Section 5.2.3).

Summary: global experience

- *Wherever pinniped populations and fisheries overlap, operational interactions generally follow.*
- *Direct interactions between fishing gear and pinnipeds is recognised as the dominant threat to global pinniped populations.*
- *Pinnipeds are readily attracted to and interact with trawl fisheries; they will take fish floating free from the net, 'stickers' (meshed fish) protruding through the net mesh, enter trawl nets to feed on fish inside the net and take discarded fish and offal.*
- *Globally, otariids (fur seals and sea lions) are highly susceptible to interactions with trawl fisheries. Key examples include:*
  - *Cape fur seals and South African trawl fisheries*
  - *South American sea lions and trawl fisheries off south-central Chile and factory/freezer mid-water and demersal trawl fisheries off northern and central Patagonia (Argentina)*
  - *Antarctic fur seals and Antarctic krill fisheries*
  - *Steller sea lions and mid-water freezer trawlers in US Alaskan fisheries*
  - *New Zealand sea lions and New Zealand fur seals and New Zealand mid-water and demersal trawl fisheries.*
- *Documentation and enumeration of the extent of interactions (including bycatch mortality) varies greatly. In many instances this is limited to short-term studies where interaction rates (usually only bycatch) have been reported and analysed based on independent fishery observer programs. Annual reporting and estimation of bycatch impacts is most consistent in US and New Zealand fisheries.*

Extent of trawl fishery interactions: Australia

In Australia the three main resident pinniped taxa frequently interact with a range of fisheries; Australian sea lions principally with gillnet fisheries and Australian and New Zealand fur seals mostly with trawl fisheries (Shaughnessy 1999, Knuckey *et al.* 2002, Goldsworthy *et al.* 2003b, Shaughnessy *et al.* 2003, Page *et al.* 2004, Hamer and Goldsworthy 2006, Tilzey *et al.* 2006, Goldsworthy and Page 2007, National Seal Strategy Group and Stewardson 2007, Campbell 2008, Lyle and Willcox 2008, Goldsworthy and Lowther 2010, Hesp *et al.* 2012, Kirkwood and Goldsworthy 2013, Tuck *et al.* 2013).

There are two main Commonwealth-managed fisheries that include trawl fisheries within the SPF area: the SESSF, which includes the CTS (comprising the South East Trawl (SET) and Victorian Inshore Trawl (VIT) sectors), the GAB Trawl sector (GABT) and the East Coast Deepwater Trawl sector (ECDWT); and the SPF. There are few observations (less than 10 over four years) of interactions between threatened, endangered and protected species (TEPS) in the ECDWT and none for VIT (Tuck *et al.* 2013), so these fisheries are not addressed further here.

Southern and Eastern Scalefish and Shark Fishery (SESSF)

The CTS extends from NSW state waters to the edge of Australia's exclusive economic zone (EEZ) from Barrenjoey Point southward around NSW, Victorian and Tasmanian waters to Cape Jervis in SA (Tuck *et al.* 2013). The main component of the CTS is the SET in which the main gears used are otter board trawl and Danish seine (the latter is not discussed further here). Most SET vessels are described as 'wet boats' that are small demersal trawlers (18–23 m in length) which store their catches using ice/brine with no freezing/processing capacity (South East Trawl Fishing Industry Association (SETFIA) 2009). In 1999, AFMA allowed 'factory/freezer trawlers' (using mid-water trawls) into the winter blue grenadier fishery off the west coast of Tasmania. As factory/freezer boats are processing at sea, they require on-board independent observers. In their first year of operation, 87 fur seals (assumed to be mostly Australian fur seals) (83 dead) were caught (13.1 seals per 100 tows) (Tilzey *et al.* 2006). The high levels of fur seal bycatch prompted AFMA to initiate analyses of Integrated Scientific Monitoring Program (ISMP) data (a scientific observer program that commenced in 1993 to gather information on catch composition and discarding levels), to estimate the number of seals caught in the broader 'wet boat' sector of the SET. This analysis found that annual seal bycatch rates by SET vessels varied greatly between 1993 and 2000, with the annual estimate being 720 seals per year across the fishery, and a total of approximately 5730 over the eight-year period monitored (Knuckey *et al*. 2002) (Table 5.1). This averaged about two seals per 100 tows, with the highest bycatch rates off the west coast of Tasmania (2.9 seals per 100 tows off west coast Tasmania; 1.9 seals per 100 tows off east coast Tasmania; 1.6 seals per 100 tows in central Bass Strait; two seals per 100 tows off NSW and eastern Bass Strait; and 1.3 seals per 100 tows off western Victoria, southeastern SA) (Knuckey *et al*. 2002). Most seals were caught in shots on continental shelf waters in less than 200 m (although they were also caught in deeper shots off western Tasmania and western Bass Strait), with the lowest bycatch rates occurring in summer and peaking during winter. About 68 per cent of seals caught were dead and the remainder were released alive (Knuckey *et al*. 2002).

Tuck *et al*. (2013) provided additional assessment of ISMP data across an additional four years (2005, 2006, 2009 and 2010) (Table 5.1). From these records, the interaction rates for 2005, 2006 and 2009 are an order of magnitude higher than those reported by Knuckey *et al*. (2002) (11.7–46.3 seals per 100 tows). It is possible these 'interactions' may include more than bycatch interactions, although Tuck *et al*. (2013) state clearly that these included "observations of wildlife directly interacting with fishing vessels (e.g. species entanglement in fishing gear)" (Table 5.1). This would imply higher levels of bycatch in the 2005–09 period. Seal mortalities averaged three per 100 tows across the four years, generally higher than those reported by Knuckey *et al*. (2002) throughout the 1990s (Table 5.1). In most years, details of species and sex are not reported in the ISMP data, although in the 2010 data, of the 30 recorded seal mortalities, 20 (66 per cent) were recorded as Australian fur seals and 10 (33 per cent) as New Zealand fur seals (Tuck *et al*. 2013). Tuck *et al*. (2013) indicated that ISMP sampling design has recently been re-designed to obtain effective and statistically robust coverage for recording of species identified as high risk through the ecological risk assessment (ERA) process, and to include TEPS. Tuck *et al*. (2013) highlighted difficulties associated with interpreting historical ISMP wildlife interactions data. These include inconsistent sampling effort between years; an apparent change in emphasis on a particular species group (e.g. birds or mammals) between years; and inconsistent species identification and coding. They stated that a "combination of improved reporting by industry, highly variable ISMP estimates and the introduction of various mitigation measures over the same time, means that the wildlife interaction data is impossible to interpret with any level of certainty at this stage". In the absence of any other data, based on the observed rates of bycatch mortality in the SET between 1993 and 2010, an average of approximately 597 fur seals may have died annually in the 'wet boat' sector of the SET as a consequence of fishery interactions over this period. This equates to approximately 12,000 fur seals over the past 20 years of the fishery.

Table 5.1 Estimates of annual fur seal bycatch in the SESSF-SET based on ISMP data, 1993–2010

YEAR	TOTAL SET SHOTS	OBS. SHOTS	OBS. SEAL INTERACTIONS	OBS. SEAL MORTALITIES	INTERACTIONS /100 TOWS	SEAL MORTALITY /100 TOWS	EST. SET SEAL INTERACTIONS	EST. SET SEAL MORTALITY
1993	35,779	564	10	7	1.8	1.2	374	254
1994	38,357	879	16	11	1.8	1.2	696	473
1995	37,850	603	10	7	1.7	1.1	735	500
1996	41,296	607	9	6	1.5	1.0	911	619
1997	42,652	727	12	8	1.7	1.1	804	547
1998	41,147	679	13	9	1.9	1.3	648	441
1999	42,774	947	9	6	1.0	0.6	344	234
2000	31,348	781	26	18	3.3	2.3	1222	831
2001	34,224	801	25	17	3.1	2.1	1068	726
2005	36,858	949	175	28	18.4	3.0	6797	1087
2006	30,311	855	100	5	11.7	0.6	3545	177
2009	21,488	633	293	27	46.3	4.3	9946	917
2010	22,564	706	35	30	5.0	4.2	1119	959
Sum	456,648	9731	733	178			28,209	7766
Average	35,127	749	56	14	7.6	1.9	2170	597

Source: 1993–2001 (Knuckey *et al.* 2002) 2005–2010 (Tuck *et al.* 2013). Total SET shots (AFMA 2009), 2009 and 2010 (AFMA 2012c).

AFMA have published quarterly reports of logbook interactions with TEPS on its website (AFMA 2014c). Over the past three calendar years (2011–13) a total of 688 fur seal interactions have been reported by fishermen in the SET, 521 (76 per cent) of which were bycatch mortalities. The level of reporting and the levels of species discrimination appear to be improving, but in the panel's view, the data are not a reliable indicator of the extent of bycatch interactions.

Minimising seal interactions has been a focus for the winter freezer trawler fishery for blue grenadier off western Tasmania. Seal excluder devices (SEDs) have been compulsory in this component of the SET since 2005, and modifications to fishing practices have been introduced to reduce the incidence of seal bycatch (see Section 5.2.3)(Table 5.2). Some research has been undertaken on the biology, ecology and nature of seal interactions in this fishery as a means to informing management and mitigation measures (Goldsworthy *et al.* 2003b, Hamer and Goldsworthy 2006, Tilzey *et al.* 2006). Intensive observations were undertaken on board fishing vessels to assess the relationship between seal numbers and a range of factors to do with trawling activity including on-board factors and the relationship to the proximity of other vessels, distances from seal colonies/haul-out locations and weather and sea conditions. In addition, underwater cameras were used to record seal activity in and around the codend of the net during trawling, primarily to record the timing and depths of net-entry. A complex suite of interacting parameters were found to be important in determining the number of seals present at any given time behind fishing vessels, including factors to do with the fishery (stage of fishing season, presence of other vessels), the vessel (speed), weather (barometric pressure) and the proximity to seal colonies/haul-outs. Numbers of seals increased in response to poor weather (decreasing barometric pressure/increasing swell height), increasing fishing activity (the number of nearby vessels and trawl frequency), and proximity to seal haul-outs/colonies. Numbers decreased with increasing vessel speed (Hamer and Goldsworthy 2006). Seal numbers at the surface generally increased throughout trawling operations, with brief declines during shooting and hauling phases (presumably when many seals were actively diving down to the net). This was substantiated with subsurface observations from a submersible video camera installed in the net, confirming the greatest period of seal activity within the net was during shooting and hauling (Hamer and Goldsworthy 2006). The numbers of seals observed in the net was similar during shooting and hauling; however, all seals observed to enter the net during shooting drowned; whereas most (86 per cent) that entered the net during hauling survived, with all seals entering during hauling observed to enter the net just prior to it breaching the surface and being hauled on board (Hamer and Goldsworthy 2006). The mean depth that seals were observed inside the net during shooting was 165 m (n=4) and 97 m (n=5) during hauling. The deepest recorded net entry during shooting and hauling was 190 m and 130 m, respectively (Hamer and Goldsworthy 2006).

Examination of the 87 dead seals collected over three consecutive fishing seasons indicated that all were Australian fur seals, most of them were males (94 per cent), most were between 2–13 years of age (although several exceeded 20 years) with a mean of 7.5 years (Goldsworthy *et al.* 2003b, Tilzey *et al.* 2006). This suggested that most fur seals interacting with the fishery were sub-adult males. A total of 50 stomachs recovered from dead seals were analysed. Fresh and undigested items within a stomach were categorised as 'net' feeding, indicating prey items consumed in the net immediately before drowning. Those that were somewhat digested were categorised as 'prior' feeding on prey that may have been consumed in prior trawls or independent of the fishery (Goldsworthy *et al.* 2003b). Results from dietary analysis indicate that seals feeding within the fishing ground were targeting trawling operations to feed on commercially-caught species (mostly blue grenadier and spotted/silver warehou *Seriolella punctata*). The similarities between 'net' and 'prior' samples gives strong evidence that seals attracted to the fishing grounds are there to feed principally on the contents of trawls. There was little evidence to suggest that any substantive foraging was undertaken away from trawling operations, as the predominant prey item in both 'net' and 'prior' samples was blue grenadier that could only be accessed by seals through net-feeding when brought into their diving range during trawling operations (Goldsworthy *et al.* 2003b, Tilzey *et al.* 2006). In contrast, dietary studies at Reid Rocks, the nearest breeding colony of Australian fur seals to the blue grenadier fishing grounds, found that fur seals consume mainly redbait, leatherjackets, jack mackerel and red cod (Hume *et al.* 2004).

A novel satellite telemetry study was undertaken to understand the movement patterns of seals directly interacting with freezer trawler vessels in the blue grenadier fishery. Seals were directly captured alongside fishing vessels at sea using a 'dip-net' lowered from each ship's crane, 1–2 m below the surface. Waste fish were used to lure seals into the net which was then raised onto the back of the trawl vessel where they were anaesthetised and fitted with a satellite transmitter (Goldsworthy *et al.* 2003b, Tilzey *et al.* 2006). Nine male Australian fur seals were tracked for up to seven months. All seals tracked foraged almost exclusively within the blue grenadier fishing grounds throughout the duration of the fishing season, and rested between foraging trips at either Hibbs Point or Reid Rocks (the nearest haul-out site/breeding colonies to the south and north, respectively). When leaving the fishing grounds, seals typically swam in a direct line towards haul-out sites, but on return, swam to the nearest edge of the continental shelf, possibly to enhance the likelihood of intercepting fishing vessels (Goldsworthy *et al*. 2003b, Tilzey *et al*. 2006). For seals that were tracked beyond the duration of the winter blue grenadier fishing season, there was a noticeable change in the focus of foraging effort. Most moved their foraging to areas south of the fishing ground, typically between Macquarie Harbour and Maatsuyker Island (south-west Tasmania). One seal foraged extensively over outer-shelf waters of southern Tasmania, as far north as Maria Island on the east coast of Tasmania, before returning to the west coast of Tasmania. The tracking studies clearly demonstrated the habitual nature of fur seals feeding in the fishing grounds in between resting at nearby haul-outs. The number of resights of satellite-tagged seals alongside fishing vessels (including one live capture and release in a trawl net), and the intensity of movements to and from the fishing grounds between haul-outs, suggested that the seal population interacting with the fishery may be relatively small and intransient during the period of the fishery (Goldsworthy *et al.* 2003b, Tilzey *et al.* 2006).

Table 5.2 Summary of the fishing effort and seal bycatch rates in the winter freezer trawler component of the blue grenadier fishery off the west coast of Tasmania, 1999–2004. Data for 1999 represent bycatch rates prior to introduction of seal-avoidance practices and Code of Fishing Practice (1999).

YEAR	VESSELS	NO. SHOTS	SEALS			INTERACTIONS /100 TOWS	SEAL MORTALITY /100 TOWS PER
			TOTAL	DEAD	% SURVIVAL		
1999	3	665	87	83	5	13.1	12.5
2000	2	453	53	22	58	11.7	4.9
2001	2	501	26	24	8	5.2	4.8
2002	2	557	58	37	36	10.4	6.6
2003	2	483	20	15	25	4.1	3.1
2004	1	239	12	8	33	5.0	3.3
2000–04*		2233	169	106	37	7.6	4.7

*per cent survival seals per shot and seal deaths per shot are presented as the mean of all shots, seals caught and killed between 2000 and 2004. **Source:** Tilzey *et al.* (2006).

Small Pelagic Fishery

From the commencement of mid-water trawling in the SPF in 2002, trawls were fitted with a 'soft' rope-mesh SED (Browne *et al.* 2005), and were subject to high levels of independent observer coverage (through AFMA), complemented by on-board monitoring by Tasmanian Aquaculture and Fisheries Institute (TAFI) scientists undertaking biological assessments of the target species (Lyle and Willcox 2008, Tuck *et al.* 2013). No marine mammal bycatch was observed in the fishery until 2004, when 14 dolphin mortalities occurred in two separate shots. In response to this, AFMA implemented 100 per cent observer coverage of fishing operations, and commissioned a pilot study (2005), followed by a larger project in 2006–2007 to investigate the nature and extent of marine mammal interactions, and trial and assess the performance of various exclusion devices (Browne *et al.* 2005, Lyle and Willcox 2008; discussed in detail in Section 5.2.3). Although these studies were initiated principally to assist in the development of cetacean bycatch mitigation, underwater video monitoring conducted during the pilot study identified that interactions with fur seals were far more numerous (Browne *et al.* 2005).

Between 2004 and 2010, a total of 184 seal interactions were recorded with mid-water trawl gear in the SPF, and of that, 175 interactions (95 per cent) were part of underwater video monitoring conducted during the scientific projects. The most detailed project was undertaken by Lyle and Willcox (2008) who used underwater video to monitor seal interactions in 98 trawls amounting to more than 700 hours of video footage. During the study, 151 seals (mostly Australian fur seals) were recorded inside the trawl net in the region of the SED in more than half of the monitored shots, peaking during autumn and winter months (70 per cent) and below 25 per cent at other times of the year. Most seals (87 per cent) entered the trawl net via the net mouth and exited via the SED opening (64 per cent), with a smaller percentage entering through the SED opening (13 per cent) and exiting via the net mouth (22 per cent, exit point of 14 per cent unknown) (Lyle and Willcox

2008). Seals entered the net at every stage of trawling, with the highest rates of interaction occurring during setting. However, numerically, most of the recorded net entries occurred during fishing (62 per cent), which accounted for most (73 per cent) of the trawl duration. As most fishing occurred in less than 150 m, the net was essentially available to seals at all stages of trawling (Lyle and Willcox 2008).

The overall interaction rates from the underwater video monitoring were 154.1 seals per 100 tows; with an estimated bycatch mortality rate of 19.4 seals per 100 tows, based on 19 observed mortalities (Lyle and Willcox 2008). However, Lyle and Willcox (2008) noted that for an additional eight seals the outcome of survival was uncertain; five of the seals were judged to be in very poor condition (low responsiveness) prior to being ejected from the net (four of which had been in the net more than 10 minutes); the remaining three seals were judged to be in the high risk range (submerged for more than 10 minutes) (Lyle and Willcox 2008). It is therefore possible that the mortality rate could have been as high as 27.6 seals per 100 tows.

A critical observation of the study was that without video monitoring, the extent of the bycatch issue would have gone unnoticed even with high levels of observer coverage, as all seal mortalities eventually dropped out of the net via the SED opening (Lyle and Willcox 2008). Consistent with this, no further records of seal bycatch were recorded by onboard observers in the SPF between 2007 and 2009 (Tuck *et al.* 2013).

Summary: Australian experience

- *Pinniped interactions with fishing gear appear ubiquitous in southern Australia where their populations overlap with trawl fisheries.*
- *Pinniped interactions occur predominantly with demersal trawl 'wet boats' and 'factory/freezer trawlers using mid-water trawl gear in the CTS of the SESSF and with mid-water trawlers of the SPF.*
- *The longest time series of data on bycatch interactions (1993–2010) exist for the 'wet boat' CTS where available ISMP data indicate persistent and significant ongoing bycatch mortality of fur seals. Extrapolation of these data suggests bycatch mortality in the order of 600 fur seals per year, or approximately 12,000 over the past 20 years (around 1.9 seals per 100 tows).*
- *Most research into the nature and extent of interactions (and their mitigation) has occurred in the winter factory/freezer mid-water trawl fishery for blue grenadier off western Tasmania. Results indicate a subpopulation of fur seals habitually interacting with and foraging in association with fishing operations for many months of the year.*
- *Information on the nature and extent of pinniped bycatch in the SPF mid-water trawl fishery is restricted to observations between 2006 and 2007, when underwater video monitoring of trawls and SEDs occurred. On-board observers significantly under-reported interactions because all seal mortalities were ejected from the SED opening and were undetectable by observers. Based on 151 observed interactions with a SED in place, bycatch mortality was an order of magnitude higher (19.4 seals per 100 tows) than that observed in non-SED CTS 'wet boat' vessels.*
- *Seals were observed to enter mid-water trawl SPF nets at every stage of trawling. Numerically, most net entries occurred during fishing (62 per cent), which accounted for most (73 per cent) of the trawl duration. As most fishing occurred in less than 150 m, the net was available to seals at all stages of trawling.*
- *In the US and New Zealand, annual reporting of marine mammal interactions includes routine analysis of the data on protected species interactions to provide an estimated take of these species. No such analysis is available for fisheries interacting with pinnipeds in southern Australia.*

5.2.3 Management

Existing management of operational interactions with pinnipeds globally and nationally

Management and mitigation of pinniped interactions with trawl vessels can include modifications to fishing gear (such as incorporating SEDs in the trawl net), modifications to fishing behaviour, bycatch trigger limits move-on rules, and spatial closures.

Seal excluder devices (SEDs)

Exclusion devices are widely used internationally throughout a range of trawl fisheries to mitigate bycatch of marine megafauna, including large sharks, stingrays, sea turtles, seals and cetaceans. Depending on their main function, they go by a range of names from more generic, including bycatch reduction devices (BRDs) and marine mammal excluder devices (MMED); to more specific including turtle excluder devices, cetacean exclude devices (CEDs), and sea lion excluder devices (SLEDs). In Australia, excluder/exclusion devices used to reduce the incidence of seal bycatch are usually referred to as SEDs.

Exclusion devices typically comprise an additional section of netting inserted between the entrance and the codend of the trawl net with an angled grid that directs marine megafauna to an escape hole in either the top or bottom of the net and prevents them from entering the trawl codend (Elgin Associates unpublished (b)) (Figure 5.10). Grids used to exclude the marine mammals are usually constructed of stainless steel (known as a 'hard' or 'rigid' grid) but can also be made from softer material such as fishing mesh or rope, or braided stainless wire and pipe (known as a 'soft' grid or 'semi-flexible' grid). Grids may be constructed as a single piece, or as a two or three piece unit. The spacing between the bars that form a grid, and the size, shape and location ('top' or 'bottom') of the 'escape hatch/hole' (or 'SED opening'), are dependent on the behaviour and size of the species that are intended to be excluded, and also the target species and fishing method used (e.g. demersal or mid-water trawl (Elgin Associates unpublished (b))(Figure 5.10).

Some SED openings are fitted with a 'hood' and 'kite', which consist of a forward-facing netted 'hood' with an opening held open by floats, and a panel ('kite') designed to direct water flow into the net and across the grid (Figure 5.10). These function to both minimise potential loss of commercial catch and to minimise the potential loss of dead or incapacitated megafauna so that mortalities or injuries can be detected (Elgin Associates unpublished (b)).

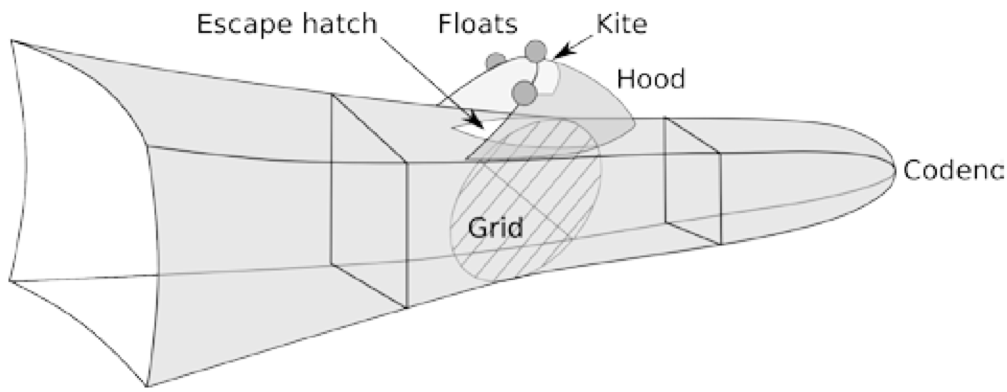

Figure 5.10 Schematic diagram of a SED consisting of a metal grid and an opening (escape hole) above it. The grid directs seals to the escape hole, enabling them to exit the net. The forward-facing hood is held open by floats, and a strip of material known as a kite. **Source:** Thompson *et al.* (2013), reproduced with permission from Ministry for Primary Industries, New Zealand.

Globally, most pinniped interactions with trawl fisheries involve otariids (fur seals and sea lions), which are predominantly a Southern Hemisphere group. Therefore, SED use in trawl fisheries is less common in the Northern Hemisphere where pinniped interactions with gillnet fisheries are generally a more significant issue (Read *et al*. 2006). Although Steller sea lions are known to interact regularly with trawl fisheries in the North Pacific Ocean and Bering Sea, SEDs do not appear to be used in commercial fisheries there, although recent trials of a MMED in a mid-water trawl net were undertaken off California (Dotson *et al*. 2010). SED use and development has been greatest in New Zealand (Auckland Islands squid trawl fishery) and Australian fisheries (CTS winter blue grenadier fishery and SPF), although some developmental work has also been undertaken in the Antarctic krill fishery. The application and effectiveness of SEDs in mitigating seal bycatch interactions in each of these fisheries is summarised below.

<u>Auckland Islands squid trawl fishery – SLED</u>

Due to high levels of bycatch of New Zealand sea lions (listed as critically threatened in New Zealand) in the Auckland Islands Squid Fishery, a SLED was developed to reduce bycatch mortality (Wilkinson *et al*. 2003). The SLED comprised an additional section of netting inserted between the lengthener and codend of a trawl net with an angled two or three panelled metal grid to guide sea lions to a top-opening escape (Hamilton and Baker 2014, see Figure 5.11).

A range of improvements to the basic design of the SLED have occurred over the past 10–15 years. These have included:

- adding a hood and kite to the top-mounted escape hole
- reducing the space between the grid bars from 26 centimetres (cm) to 23 cm (to reduce the probability of juvenile sea lions passing through the grid)
- modifying the SLED kite with additional floats on the top of the SLED hood to ensure the kites and hood operate properly in all conditions and the escape hole remains open during fishing (Ministry for Primary Industries 2012, in Elgin Associates unpublished (b)).

Since 2004–05, there has been widespread use of government-approved, standardised SLEDs in the Auckland Island Squid Fishery (Ministry for Primary Industries 2012, in Elgin Associates unpublished (b)) (Figure 5.11). Although not mandatory, the use of SLEDs is required by the current industry body, applied fleet-wide and monitored by fishery observers (Ministry for Primary Industries 2012, in Elgin Associates unpublished (b)). Following the introduction of SLEDs, the number of New Zealand sea lions captured in the Auckland Islands Squid Fishery declined from 14–142 per year (pre-SLED deployment, 1995–96 to 2001–02), to 4–31 per year (post-SLED deployment, for the period 2004–05 to 2010–11) (Thompson *et al*. 2013). SLEDs appear to be effective in enabling most sea lions to exit the trawl net, however some still drown and are retained, and there has been concern and uncertainty about the number that may drown and be ejected from the net, or escape but not survive the interaction (e.g. injuries sustained from collisions with grids). Although fisheries managers considered it unlikely that dead sea lions would fall out of a top-mounted SLED escape hole that has also been fitted with a hood (as detailed above), it has not been possible to verify this with video monitoring because of the poor visibility at fishing depth due to water turbidity, light limitations and fine debris and squid ink suspended in the water column (Hamilton and Baker 2015). Following the fleet-wide introduction of SLEDs, it has therefore been difficult to estimate the number of sea lions interacting with the fishery, and the operational effectiveness of SLEDs with respect to both the survival and mortality rates of sea lions interacting with them. This uncertainty has been exacerbated by a continued decline in the sea lion pup production at the Auckland Islands since 2004–05 (when SLEDs were in widespread use).

As a consequence, a range of research has been undertaken to assess whether SLEDs successfully eject sea lions, and if ejected sea lions survive. This research has included:

Placing cover nets over SLED openings to determine how many seal are ejected from the net: In 2001, vessels with SLEDs, and independent observers, fitted cover nets over SLED openings. In 276 tows, 33 seals were caught (12 seals per 100 tows), of which 30 were successfully ejected into the cover net by the SLED, giving an ejection rate of 91 per cent (Wilkinson *et al*. 2003). Underwater video monitoring inside the cover net of three animals (alive when they exited the SED opening), indicated that they would have likely survived if the cover net had not been in place.

Autopsies of bycatch sea lions: Examination of the retained and frozen carcasses by a veterinary pathologist concluded some of the animals exhibited severe internal trauma which, it was considered, would have led to their subsequent death (Gibbs *et al.* 2001 in Wilkinson *et al.* 2003). However, it was also acknowledged that freezing of carcasses often involved rough handling onboard fishing vessels (including dropping some animals six metres into fishing holds for storage), which may have induced changes that could be confused with true lesions. To look at effects of freezing and thawing on seal carcases, five chilled and five frozen New Zealand fur seals recovered from trawl nets without exclusion devices in the New Zealand hoki fishery were examined. Results from this study confirmed that some lesions originally thought to be caused by trauma were in fact an artefact of freezing (Roe and Meynier 2012).

Analysis of video footage of Australian fur seals interacting with SED in SPF: Because of the limited usable video footage available for New Zealand sea lion interactions/collisions with SLED grids, available footage of Australian fur seal interactions with SEDs in the SPF (Lyle and Willcox 2008) was used as a proxy to help assess the possible nature of New Zealand sea lion and SLED interactions and, in particular, the potential of head trauma injuries that may result from head-first collisions with a metal grid (Lyle 2011). Interactions with SEDs were described for 132 seals, and indicated that about one third of seals that entered via the net mouth experienced a head-first collision with the grid (usually the upper half of the SED grid) and usually the angle of the head was more or less perpendicular to the grid (Lyle 2011). Impact velocities were also estimated for these collisions.

A biomechanical study that simulated the impact of sea lions hitting the metal grid of a SLED: Ponte *et al.* (2010) used a validated method for measuring head impact injury in human pedestrians ('crash tests') with scaling and extrapolating to account for the relative head and brain mass of the New Zealand sea lion to assess the likelihood of mild traumatic brain injury (i.e. 'concussion') to a sea lion as a result of a head impact with a stainless steel SLED grid. For particular impact locations on the SLED grid, the likelihood of a brain injury, based on swim speed and effective sea lion head mass, was determined (Ponte *et al.* 2010). 'Crash test' results indicated that sea lions colliding with the grid may incur some sort of brain injury and the risk of life-threatening brain injury may be higher than 85 per cent for a female sea lion in a 10 metre per second (m/s) collision with the SLED grid at the stiffest location tested (based on trawl speed of 2 m/s and estimated burst speed of an adult sea lion of 8 m/s) (Ponte *et al.* 2010). However, this impact speed probably represents the worst case scenario, especially if Lyle's (2011) fur seal interaction speeds are considered indicative of New Zealand sea lion interactions (Hamilton and Baker 2014).

Modelling the risk of sea lions suffering mild traumatic brain injury after striking a SLED grid: Based on Ponte *et al.* (2010) and Lyle's (2011) analyses, Abraham (unpublished) developed a simulation-based probabilistic model to estimate the risk of a sea lion suffering a mild traumatic brain injury when striking a SLED grid. The estimated probability of mild traumatic brain injury from a single collision was estimated to be less than 5 per cent.

The New Zealand Ministry for Primary Industries considered that collectively, the research and assessments of SLED efficacy (summarised above), provides robust evidence that SLEDs greatly increase the survival probability of sea lions that enter a trawl net, and that the weight of evidence is that SLEDs are effective in reducing the incidental mortality of New Zealand sea lions in the Auckland Islands Squid Fishery (Ministry for Primary Industries 2012, in Elgin Associates unpublished (b)).

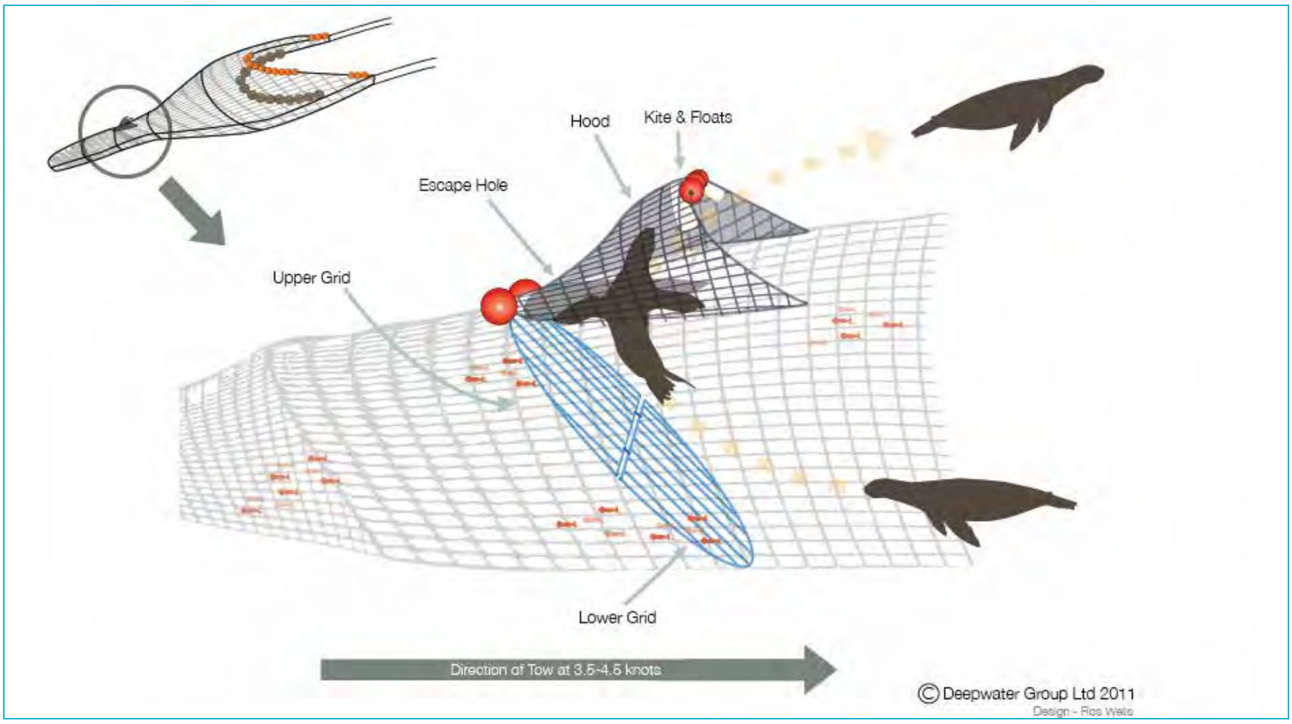

Figure 5.11 A standard SLED used in the Auckland Island Squid Fishery (SQU6T). **Source:** Reprinted from *Fisheries Research* 161 (2015) S. Hamilton and B. Baker, B. (2015). Review of research and assessments on the efficacy of sea lion exclusion devices in reducing the incidental mortality of New Zealand sea lions *Phocarctos hookeri* in the Auckland Islands squid trawl fishery. pp. 200–206. Copyright (2015), with permission from Elsevier B.V.

Antarctic krill fishery

The commercial trawl fishery for Antarctic krill is managed by CCAMLR. Large numbers (292) of Antarctic fur seal mortalities associated with the krill trawl fishery in Subarea 48.3 in the 2003–04 season (Reid and Grilly (2014) in Elgin Associates unpublished (b)), prompted the development and trialling of a range of SEDs to avoid fur seal deaths in the fishery (Hooper *et al.* 2005). Mitigation measures for fur seal bycatch were tested for krill vessels fishing around South Georgia in the 2004 fishing season (Hooper *et al.* 2005). Four approaches were trialled: physical barriers (panels of netting) excluding seals from entering the net; physical barriers (panels of netting) positioned within the net accompanied by escape channels or openings; manufactured SEDs in front of the codend that were composed of a separator grill that deflected seals to an escape opening; fishing gear configured with panels of a mesh size adequate to allow seals to escape (i.e. the forward part of the roof of the net had three large mesh panels inserted into it of mesh size 16 m and a further two panels of mesh size 4 m which appeared to allow the seals to escape alive and unharmed). It was considered that in all the above four cases, the incidence of seal entanglements during the 2004 season was either eliminated or greatly reduced (Hooper *et al.* 2005), however, the authors do not discuss the possibility that dead animals may have fallen out of the SED escape hole on hauling, and no underwater video monitoring was undertaken to confirm SED performance.

Low levels of bycatch mortality of Antarctic fur seals have since been reported in Subarea 48.3: one in the 2005–06 fishing season (from 15 per cent of the total fishing effort observed); zero in 2006–07 and six in 2007–08 (no details on observer effort) (Reid and Grilly (2014) in Elgin Associates unpublished (b)). CCAMLR has adopted a general mitigation measure (Conservation Measure (CM) 25–03) and introduced the mandatory use of marine mammal exclusion devices on trawls in the krill fisheries in Area 48 (CM 51–01), Division 58.4.1 (CM 51–02) and Division 58.4.2 (CM 51–03) (Elgin Associates unpublished (b)).

CCAMLR does not specify a standard exclusion device as there are a number of different vessels utilising different net designs, with each net design requiring a particular mitigation set up that suits the characteristics of the vessels. All exclusion device designs are included in the notification process for participation in CCAMLR fisheries and are reviewed prior to the vessels engaging in the fishery (Keith Reid, CCAMLR Science Manager, pers. comm. 7 October 2014).

SEDs in the Australian CTS, winter blue grenadier fishery

In 1999, in response to particularly high levels of incidental captures of seals on factory/freezer trawlers using mid-water trawl gear in the winter blue grenadier fishery off the west coast of Tasmania, the industry initiated a collaborative project with researchers to reduce seal bycatch (Tilzey *et al.* 2006)(Table 5.2). The project included trialling and developing a suitable SED and assessing its effectiveness in reducing seal mortalities. Tilzey *et al.* (2006) experimented with a range of different SED designs. Problems encountered included significant fish-loss via the SED escape hatch and blockage of the SED grid with larger sized target species (blue grenadier) and when catching high volumes of fish. A forward-facing 'top-hatch' SED had a significantly lower occurrence of seal bycatch than other SED designs and nets without a SED. The top opening SED was considered markedly superior to a bottom opening SED because it better facilitated both seal exit (seals more likely to swim upwards) and reduced the likelihood of seal entry via the escape hatch (Tilzey *et al.* 2006). Bycatch survival rate of seals in nets fitted with SEDs were 48 per cent compared to zero for nets without SEDs, largely because the SEDs prevented seals entering the codend where most deaths probably occur (Tilzey *et al.* 2006). However, SED performance remained largely unquantified because underwater video footage was limited and the numbers of seals interacting with the trawl net and successfully exiting the net via the SED escape hatch during this study were unknown. Obtaining significant results on SED performance by comparing replicate sets of trawl shots with and without a SED was difficult, because of the generally low level of seal bycatch and the complex suite of factors influencing seal interactions with the trawl net (Tilzey *et al.* 2006).

Hamer and Goldsworthy (2006) used underwater video monitoring to examine subsurface interactions with the SED to establish its effectiveness at reducing bycatch and mortalities. They provided details on the mortality of 13 fur seals; six entered the net during shooting, all of which died, while seven entered during hauling only one of which died. Six seals were caught while a SED was in place, three of which died, while seven seals were caught in trawls with no SED, four of which died. Based on this sample there was no significant difference in mortality rates between trawls with or without SEDs (Hamer and Goldsworthy 2006). Hamer and Goldsworthy (2006) suggested that, although a significant reduction in seal bycatch has been recorded since 1999 (see Table 5.2), attributing it to the introduction of SEDs was not supported by the evidence because all but one net-entry observed through underwater monitoring resulted in bycatch. The one animal that exited the net did so through the mouth and none were observed to exit through the SED opening. Instead, Hamer and Goldsworthy (2006) suggested that the apparent reduction in seal bycatch was due to a reduction in the incidence of seal-net interactions, as a consequence of other management measures. They also noted that seal bycatch was reduced when haul speeds were low which contradicted the recommendation in the 2007 Code of Fishing Practice, that nets should be hauled as quickly as possible to reduce the time that it remains within the diving range of fur seals (SETFIA 2007). Hamer and Goldsworthy (2006) suggested that net haul speed should be as fast as possible below the maximum dive range of seals (approximately 200 m) to reduce the length of time available to seals, but should then slow to speeds slower than the minimum average swim speed of fur seals (approximately 7.2 km per hour) to reduce the likelihood of seals becoming caught in the net in the upper water column. Further subsurface video monitoring of SED performance was recommended (Hamer and Goldsworthy 2006).

The current AFMA 'Gear Requirement' for the freezer processing vessels in the CTS of the SESSF includes a requirement that a SED is used in every trawl shot and that the SED complies with the following specifications.

- A grid is used to prevent seals from entering the codend of the trawl net, being a grid that is made of a rigid material strong enough to repel a seal (such as a 25 millimetre (mm) diameter stainless steel rod) with spacing between bars of no more than 250 mm. The grid must conform as closely as possible to the corresponding cross-section dimensions of the net.
- The escape hatch must be no smaller than 800 mm in length and 600 mm in width at its widest point and be free of obstruction and be located at the top of the net adjacent to the SED.

- The use of a 'hood' over the escape hatch is optional. If a hood is used it must be made of mesh no greater than 40 mm and have a kite attached to the leading edge of the escape hatch that ensures that the escape hatch egress is maintained.
- At least one single 20 cm diameter float is attached at the centre of the leading edge of the kite for initial flotation.

Ongoing SED performance issues in the winter blue grenadier fishery, mainly with SEDs clogging with large target species or higher fish volumes, has prompted some recent new developments with the design of the SED. AFMA has been working with the operators of the vessel *FV Rehua* to improve SED performance, for both seal exclusion and fish quality. The new design includes a hydrostatic net release, used to release a net binding after the gear has been shot away to an appropriate depth, as well as an acoustic transponder release of the SED gate (termed an 'Acoustic SED') which excludes seals from the codend during hauling (Mike Gerner, pers. comm. in Elgin Associates unpublished (b)).

Key features of the hydrostatic net release system include the net being bound with sisal, then hydrostatically released at a depth of 300 m (but could be adjusted to suit conditions/seal diving depth as required). The hydrostatic binding holds the net together very close to its mouth, preventing seals entering while the net is being shot, with the net only being opened at a depth considered below the typical diving depth of fur seals. This device was trialled in 2013 due to the observation of seals entering the mouth of the net during setting and resulting in mortalities during the 2012 season. As noted from previous underwater monitoring studies, most seal mortality occurs in this fishery during setting (Hamer and Goldsworthy 2006), so ensuring the net remains closed till reaching fishing depth (300–600 m) and below the typical dive range of fur seals, should significantly reduce the opportunity for seals to become entrapped in trawl gear.

The acoustic SED consists of a two-piece grid sewn into the net in front of the codend. The top half of the grid is hinged (the gate), while the bottom half is fixed. The top gate remains open during fishing at depths beyond the diving range of most fur seals, and provides an unimpeded path for fish flowing into the codend. While the top gate is in the open position, it covers the top opening SED escape hole, also preventing the loss of fish through the seal escape hole. Once sufficient fish have been caught, the gate can be triggered to close by an on-board acoustic transponder that sends a signal to the release device (sewn into the net), freeing the latch and allowing the gate to drop. Hauling then commences. Closing the gate opens the escape hole and closes the SED, prevents seals entering the codend and enables ejection through the SED opening.

A report on the efficacy of the acoustic SED is currently being prepared (Mike Gerner, pers. comm. in Elgin Associates unpublished (b)). At this stage the video footage has yet to be reviewed and reported on (pending funding availability).

<u>SEDs in the SPF</u>

Mid-water trawling commenced in the SPF in late 2002 (Lyle and Willcox 2008). From the commencement of this fishery, mid-water trawls were fitted with a 'soft' rope-mesh device (SED) (Browne *et al.* 2005). The mortality of 17 dolphins between 2004 and 2005, prompted AFMA to commission a pilot study (2005), followed by a larger project (2006–07) to investigate the nature and extent of marine mammal interactions and trial and assess the performance of various exclusion devices (Browne *et al.* 2005, Lyle and Willcox 2008). Lyle and Willcox (2008) trialled three SED configurations, including: (i) 'bottom opening, small escape hole', (ii) 'bottom opening, large escape hole', and (iii) 'top opening'. The bottom opening SED was composed of two panels, producing a 2.3 by 2.3 m steel grid, with 10 vertical steel bars with 21 cm spacing. The SED was angled forwards at about 15–25°, with the escape opening located at the base of the SED. The 'small escape hole' configuration, with an approximate 1 by 1 m escape opening, was trialled initially. The hole was subsequently enlarged to 1.9 m wide, producing the 'large escape hole' configuration. Escape holes were either left open, or had a flap of netting or short lengths of rope attached to the leading edge in an attempt to discourage the loss of target species while not hindering the exit of large bycatch species. The top opening SED was constructed from four panels, producing a 5 by 2.1 m grid with 23 cm spaced steel bars angled backwards at 45°. A 1.8 by 0.55 m deep escape opening was positioned on top of the net immediately in front of the SED. A cover flap of trawl netting was attached to the leading edge of the escape opening. The 'bottom opening, small escape hole' configuration was used continuously until early June 2006 when the escape opening was enlarged ('large escape hole' configuration) following several seal mortalities. The 'large escape hole' configuration was used to the end of January 2007. The 'top opening' configuration was then trialled for about a month but owing to operational problems (specifically difficulties in retrieving the SED onto the net drum), it was deemed operationally unsuitable for the vessel and replaced with the bottom opening configuration at the end of the study period (Lyle and Willcox 2008). Underwater video footage as detailed above (98 tows, 735 hours) was used to assess SED performance, however, only the two bottom opening SED configurations could be compared owing to the limited number of shots where the top opening SED was used. The 'large escape hole' SED (where the opening had been enlarged so that there was no floor in the net immediately in front of the grids) had a three-fold reduction in lethal interactions compared to the small escape hole SED (7.2 seals per 100 tows vs. 20.0 seals per 100 tows).

Summary: seal excluder devices

- *Although excluder devices are commonly used in trawl fisheries globally as a means to mitigate bycatch of marine megafauna, with the exception of one Antarctic fishery, SEDs are mostly used in New Zealand and Australian fisheries.*
- *SEDs are typically tailored to individual fisheries, fishing vessels and bycatch species because a single design is not suitable for all circumstances.*
- *A SED functioning under optimal operating conditions should reduce the incidence of bycatch mortality of pinnipeds, but will not eliminate it.*
- *SEDs leave on-board observers effectively blind to the extent of interactions and to the effectiveness of SEDs in ejecting seals in a healthy state from the net. Underwater video monitoring of SEDs is necessary to monitor interaction levels and cryptic mortality and to optimise SED design and efficacy.*
- *Innovations in SED design are emerging from the winter blue grenadier fishery. These include a hydrostatic net release, an acoustic transponder release grid gate and installation of smaller sized mesh on the hood. The acoustic SED shows promise for demersal trawling activities that take place below the normal diving range of seals. They are less likely to be effective in shallower, mid-water trawling where seals can access the net at any stage.*
- *SED trials in the mid-water trawl fishery of the SPF indicated lower seal mortality with a larger SED opening (in a bottom opening SED). Top opening SEDs were not able to be fully evaluated due to operational difficulties.*

Fishing behaviour

In many trawl fisheries, mandated or voluntary codes of practice have been developed and adopted by industry to reduce the level of interactions with seals. The most relevant to the DCFA come from those developed in New Zealand and Australian trawl fisheries. These are summarised below.

Marine Mammal Operational Procedures (MMOP) developed in New Zealand and agreed upon by quota holders are designed to reduce the risk of incidental capture of marine mammals during deepwater trawling operations (vessels greater than 28 m in length) in EEZ waters (Deepwater Group 2011). The MMOP assumes that marine mammals are most at risk when trawls are on or near the surface (less than 50 m). Fish in the nets is the key attractant to marine mammals, and any action taken to reduce the time the net is on the surface is effective in reducing this risk. All vessels must adopt the following practices (Deepwater Group 2011) to minimise incidental catches of marine mammals.

- Remove all 'stickers' (meshed fish) before shooting the trawl.
- Undertake shooting and trawling as quickly as possible.
- If large numbers (more than five) seals congregate around the vessel when the gear is hauled, the vessel should steam away from them before setting the gear again.
- Always endeavour to mend the trawl net with the whole net on deck; if this is not possible, avoid mending while hauling.
- Each vessel shall designate one or more crew member(s) to be on watch during every shoot or haul and determine if marine mammals have been captured and to organise timely humane assistance to release captured animals alive.

The procedure details that gear failures, especially when shooting or hauling (and if the trawl mouth is left open for extended periods), can create high-risk situations for marine mammals leading to multiple marine mammal capture events. In the event of a gear failure, which may delay the shooting or hauling of the gear, either of the following should occur:

- Keep the gear deep in the water, even if this means re-shooting the gear; if the gear is to remain in the water the gear headline height should be at least below 50 m and preferably below 100 m, or
- Bring the gear, or at least the ground rope and headline, on board to ensure the net mouth is closed.
- All vessels have offal management procedures and recommended fishing practices that are detailed in individual vessel management plans (Deepwater Group 2011). In support of these, the MMOP details the following actions relating to offal and rubbish disposal that will reduce the risks to marine mammals.
- Fur seals and sea lions eat fish and offal discarded from fishing vessels. These discards are likely to keep marine mammals near a vessel and this is to be avoided.
- Fish offal and waste fish must be fish-mealed where possible. If fish waste discharging is unavoidable, then do not discharge while shooting or hauling the net. Ensure a fish waste holding facility is available to allow this.
- Maritime regulations prohibit the dumping of any plastic waste and netting at sea. Marine mammals and seabirds are known to ingest such waste.

In the SET of the SESSF, the bycatch of fur seals by three freezer trawlers in 1999 prompted the development of a research program to mitigate seal bycatch in this fishery. The primary components of the program were the development of seal avoidance practices (SAPs) and SEDs both aimed at reducing the incidence of seal interactions in the fishery (Tilzey *et al.* 2006). Between 2000 and 2003 season, vessels generally adhered to the following SAPs (Tilzey *et al.* 2006).

- The vessel steamed at an average speed of 10–12 knots for at least 40 minutes prior to shooting the gear regardless of the number of seals observed.
- If seals were still present, gear deployment was delayed and the vessel continued steaming at 10–12 knots for a further 20 minutes.
- Fish meshed in the net (stickers) were removed prior to shooting the gear.
- All shooting and hauling was carried out as rapidly as possible.
- The vessel often made a sharp turn when shooting the bottom trawl to keep the net closed on descent.
- During fishing, the gear was not lifted into the top 150 m of the water column to make turns or a change in direction.
- After hauling, the vessel turned 90–180 degrees immediately after the net was on deck.
- The vessel steamed away from the hauling area at an average speed of 10–12 knots for at least 40 minutes after hauling, regardless of the estimated time of the next shot.
- When fixing the net or streaming it for cleaning, the codend was always open and the SED escape hatch closed. The mouth of the trawl was always on board at this time.
- The discarding of fish, processing offal or domestic waste on fishing grounds was rigorously avoided.

Adherence to these SAPs appeared to halve the incidence of seal bycatch in the winter fishery for blue grenadier from 1999 levels (Tilzey *et al.* 2006). Many of these measures were adopted into SETFIA's Code of Fishing Practice (SETFIA 2003, 2007), across the remainder of the 'wet boat' fleet, with some key exceptions. These exceptions included the requirement for vessels to actively steam away from seals before deploying the trawl net, and the removal of 'stickers' from the net prior to deployment (first three dot-points above) (SETFIA 2003, 2007, National Seal Strategy Group and Stewardson 2007).

The SPF's Bycatch and Discard Workplan included the development and implementation of vessel management plans (VMPs) to minimise TEPS interactions and record procedures for reporting on catch and wildlife interactions (AFMA 2011).

Spatial closures

Spatial closures, often termed 'time/area closures', are commonly used to manage interactions with both targeted and bycatch species in many fisheries (O'Keefe *et al*. 2014). Their use to mitigate bycatch typically occurs where there is a high degree of spatial and/or temporal overlap between target and bycatch species. Closures can produce simple and enforceable fisheries management outcomes. Some examples of their application in mitigating pinniped bycatch interactions are provided below.

New Zealand squid trawl fishery (Auckland Islands)

A 12 nautical mile (nm) (22 km) trawl exclusion zone was established around the Auckland Islands in 1982 in response to high levels of New Zealand sea lion bycatch mortality in the fishery (Wilkinson *et al*. 2003). This was converted to a Marine Mammal Sanctuary in 1995 (Chilvers 2008). However, the efficacy of this closure has been questioned, given that female New Zealand sea lions have been shown to forage over and utilise most of the Auckland Island Shelf, and that the Sanctuary does not include areas where the likelihood of interactions (and hence bycatch) are likely to be greatest (Chilvers 2008, Chilvers *et al*. 2011). Furthermore, since the closures were introduced, the pup production of the Auckland Island population has declined by 30 per cent (Chilvers 2008).

Alaskan fisheries

Since Steller sea lions were first listed as threatened under the US Endangered Species Act in 1990, a complex suite of time-area closures have been introduced around their breeding colonies and haul-out sites in Alaska, in order to mitigate adverse effects of fishing. Further measures were introduced after the western stock was listed as endangered in 1997 (Committee on the Alaska Groundfish Fishery and Steller Sea Lions 2003). A range of time-area closures were implemented in 1990s, the result being that most colonies are now protected by 20 nm trawl exclusion zones. Some of these are permanent closures; others are temporal to coincide with particular fishing seasons for specific target species (Committe on the Alaska Groundfish Fishery and Steller Sea Lions 2003). Although the major intent of these closures was to mitigate the potential adverse effects of localised depletion of key prey species in Steller sea lion critical habitat (Committe on the Alaska Groundfish Fishery and Steller Sea Lions 2003), it has been recognised that these closures also contributed significantly to the 90 per cent reduction in incidental bycatch mortality observed between the 1980s and 1990s (Perez 2003). This is discussed further in Chapter 6.

Australian fisheries

In 2010 and 2011, AFMA introduced spatial closures around all ASL colonies off SA, as part of a range of management measures introduced into the shark gillnet component of the GHAT Fishery (AFMA 2010a) (Figure 5.12). This followed research that integrated an on-board independent bycatch observer program on gillnet vessels, and an extensive ASL satellite tracking and spatial modelling program (Goldsworthy *et al*. 2010) (Figure 5.5). This study found a very strong positive relationship between observed sea lion bycatch rates and the underlying estimated sea lion density (i.e. lowest sea lion bycatch rates in lowest density areas; highest bycatch rates in highest density areas). As central place foragers, ASL density is typically highest in waters surrounding their colonies and haul-out areas; hence the spatial fishing closures introduced have resulted in a marked reduction in fishing effort in the high-density ASL area. Incidence of bycatch has reduced significantly since the introduction of the spatial closures and the other management actions (trigger limits; gear switching options).

Bycatch trigger limits/move-on rules

New Zealand trawl fisheries (Auckland and Campbell Islands)

Bycatch trigger limits are generally utilised to ensure that bycatch levels of protected species do not exceed a certain threshold that places the species or population at risk of further declines. The United States Marine Mammal Protection Act of 1972 (MMPA) specified that marine mammal stocks be maintained at an optimum sustainable population level (OSP). The act does not define OSP, but the National Marine Fisheries Service (NMFS) has interpreted OSP to be a population level that falls between maximum net productivity level (MNPL) and carrying capacity (K) (Moore 2013, Roman *et al*. 2013). In 1994, a new management approach was adopted under the MMPA, potential biological removal (PBR), which was specifically developed to assess marine mammal mortalities associated with commercial fisheries, and is defined as the maximum number of animals (excluding natural mortalities), that may be removed from a marine mammal stock while allowing that stock to recover to or be maintained within its OSP (Roman *et al*. 2013).

The PBR approach has been used in New Zealand to set an annual maximum allowable level of fishing-related mortality (MALFIRM). This sets the maximum limit of the number of New Zealand sea lions that can be killed incidentally in the Auckland Islands squid trawl fishery, before it is closed for the season. Since 1992, annual fishing operational plans have defined the management regime for each year including the required observer coverage to allow a statistically robust estimation of incidental captures, and steps to be taken if the estimated New Zealand sea lion mortality from squid fishing for that season approaches the bycatch trigger limit (Wilkinson *et al*. 2003). In 2003, this PBR/MALFIRM approach was superseded by the fishery-related mortality limit (FRML), developed from a more detailed Bayesian population model for the species (Breen *et al*. 2003). The fishery has been mandatorily closed by government fisheries managers in seven seasons since these bycatch trigger-limits were introduced, although the decision to close the fishery was overturned by court orders in the 2003 and 2004 fishing seasons (Robertson and Chilvers 2011).

With the introduction of SLEDs into the fishery, estimating the number of sea lion interactions has become increasingly difficult. The FRML now has to take into account the expected bycatch (strike) rate (if there were no SLEDs), and the number of animals that had they been caught would have escaped through the SLED opening (Thompson *et al*. 2013). Therefore, new models include both capture and SLED retention probabilities so that the total bycatch (observed and unobserved) can be estimated (Thompson *et al*. 2013). Adequate observer coverage has been critical in enabling a full and accurate assessment of New Zealand sea lion bycatch, and enabling assessment of risk and interaction levels in real time. Vessels are required to inform fisheries managers immediately on any sea lion capture event so the appropriate management response can be considered (Deepwater Group 2011).

<u>Australian fisheries</u>

Bycatch trigger limits have recently been utilised by AFMA to mitigate bycatch mortality in the GHAT Fishery, on Australian sea lion populations off SA (AFMA 2013d). In April 2011, AFMA put in place changes to the existing ASL Management Strategy (AFMA 2010a) to modify fishing arrangements in the GHAT fishery. AFMA determined that a bycatch rate of 1.5 per cent (of the female breeding population throughout one breeding cycle, an 18-month period, or 52 female sea lions) was likely to represent a sufficiently precautionary trigger level for ASL bycatch in the seven management zones identified in the ASL Management Strategy (AFMA 2010a). In 2012, AFMA amended the trigger-levels to take into account individual subpopulations (breeding colonies) within each of the seven management zones, several of which have been recognised as being at risk of becoming locally extinct. AFMA set an overall bycatch level of 15 animals per year, trigger limits within each zone ranging from one to five sea lions (Figure 5.12). Where zone trigger levels are met (or exceeded) the zone is closed to gillnet fishing for 18 months from the date of the last mortality. At any time, if the overall mortality number of 15 is exceeded, the entire ASL Management Zone will be closed for a period of 18 months from that time (AFMA 2013d). Given there is 100 per cent observer coverage in this fishery, mostly through electronic monitoring, there is high compliance and most sea lion bycatch is now reported in logbooks (AFMA 2013c).

The trigger limits had immediate effect, with bycatch incidents in February, March and April 2012 resulting in the closure of three fishing zones (A, B and D).

Summary: other management measures

- *Codes of practice have been used to reduce the level of interactions with seals. The most relevant elements of these include:*
  - *removing all 'stickers' before shooting the trawl*
  - *undertaking shooting and trawling as quickly as possible*
  - *suspension of trawling and moving away if seals are observed prior to trawling*
  - *no discarding of fish, offal or domestic waste on fishing grounds.*
- *Spatial closures can provide an effective means of reducing or removing fishing activity in locations or at times where direct interactions with seals are likely to be common, or present unacceptable risks to threatened or protected species' populations.*

- *Bycatch trigger limits are generally utilised to ensure that bycatch levels of protected species do not exceed a threshold that places the species or population at risk of further declines. They have been used to cap incidental mortality of the threatened New Zealand sea lion in the Auckland Island squid trawl fishery, and in Australia AFMA uses bycatch trigger limits to limit the bycatch of the threatened ASL in the GHAT Fishery.*

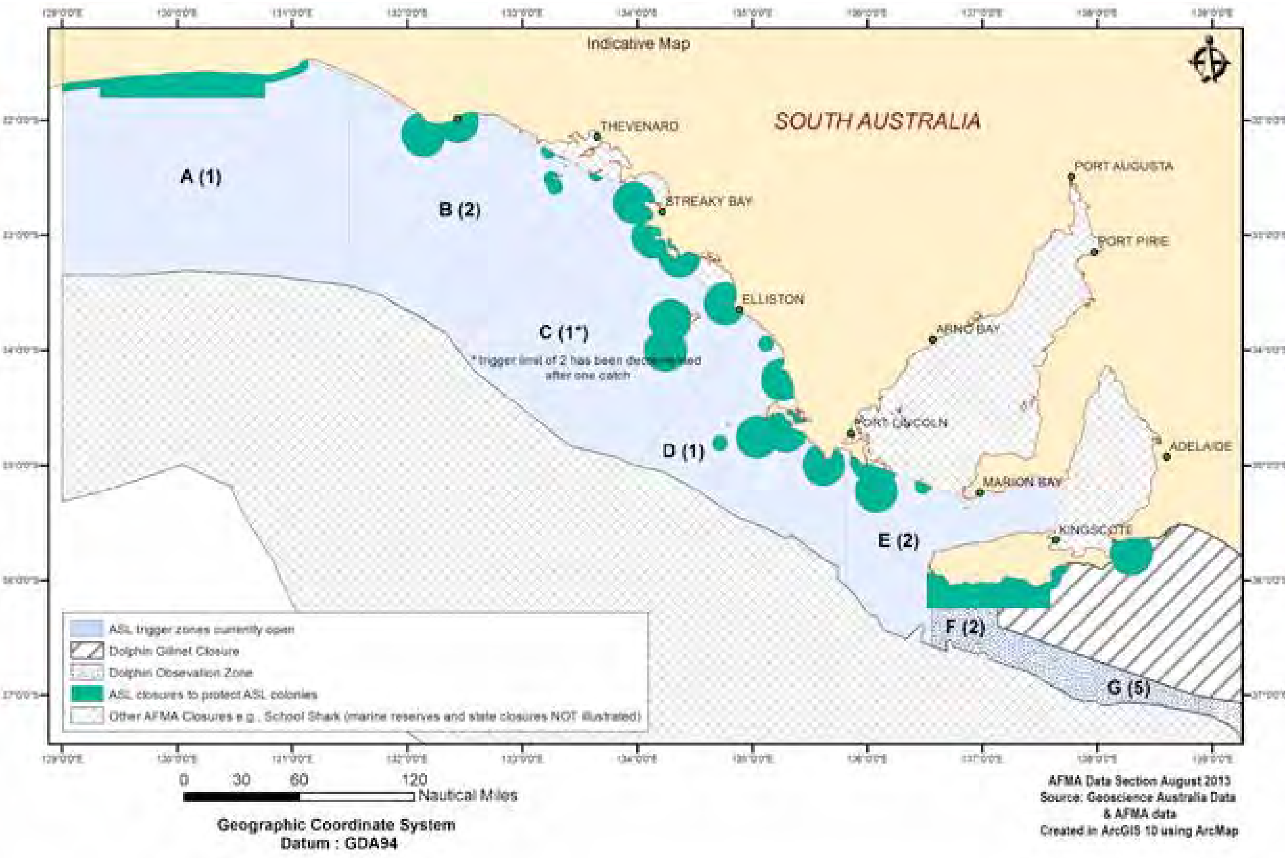

Figure 5.12 SESSF closures under the Australian sea lion management strategy, 23 August 2013. Source: AFMA (2013d).

Proposed management of direct interactions with pinnipeds in the DCFA

With respect to managing and mitigating pinniped interactions, the operations of large-scale mid-water trawl operations would have been subject to the conditions specified as 'Condition 1' in the Schedule to the accreditation of the fishery made by the Environment Minister on 3 September 2012 (see Section 3.2.4). Parts (a) and (d) of the condition relating to fur seals was to be implemented through a VMP. The VMP prepared by AFMA for managing and mitigating seal and dolphin interactions for the *FV Abel Tasman* is indicative of what was proposed (see Box 5.1).

Box 5.1 Proposed *FV Abel Tasman* Seal and Dolphin Management Plan: Boat Specific Mitigation Measures and Operational Requirements

Mandatory Gear Requirements: Exclusion Device

The concession holder must have an AFMA approved Seal and Dolphin Excluder Device installed within the net at all times while conducting fishing operations.

The concession holder must ensure the escape hole on the Excluder Device is upward opening and has a hood attached to reduce any potential fish loss and to not allow any large animals to fall out of the net if immobile and the net is inverted.

Mandatory Fishing Operation Requirements

Ensure an AFMA observer is onboard the boat at all times and assist the AFMA observer to monitor fishing operations at all times.

Ensure the boat has underwater cameras operational at all times while undertaking fishing and those cameras constantly record any take of bycatch and/or the excluder device.

Allow the onboard AFMA observer access to review any footage recorded by the underwater camera at least once every 24 hours for the duration of each fishing trip.

Not deploy any trawl nets if dolphins are sighted around the boat by any crew member or onboard observers, until the dolphins have dispersed of their own accord or the boat has steamed away and are no longer in sight of the boat.

Not deploy any trawl nets if seals are sighted within 300 m of the boat by any crew member or onboard observers, until the seals have dispersed of their own accord or the boat has steamed away and are no longer within 300 m of the boat.

Ensure observers are notified prior to the deployment and/or the recovery of trawl nets, day and night, in order to allow the observers to be present to detect any seals which become enfolded or caught at the surface, so the animals can be rapidly and humanely released.

Take all reasonable steps to ensure that, as far as practicable, if a seal or dolphin is captured in a trawl net as a result of fishing operations, the mammal is released alive and unharmed.

Record any interaction with any protected species in a logbook onboard the boat and notify AFMA in writing detailing any interactions with protected species, including any mortalities, every 24 hours for the duration of each fishing trip.

Mandatory Interaction Requirements

Dolphins

If fishing operations conducted by the method of mid-water trawling result in the death of one or more dolphins in any one shot the holder must:

- suspend fishing immediately;
- notify the AFMA observer onboard of the dolphin mortalities and with the assistance of the AFMA observer review the effectiveness of mitigation measures used in fishing operations; and
- not recommence fishing within 50 nm of the event.

Seals

If fishing operations conducted by the method of mid-water trawling result in the death of a seal in any one shot the holder must:

- suspend fishing immediately; and
- notify the AFMA observer onboard of the seal mortality/ies and with the assistance of the AFMA observer review the effectiveness of the mitigation measures used in fishing operations before recommencing fishing.

If fishing operations conducted by the method of mid-water trawling result in the death of:

- three or more seals in each of three consecutive shots; or
- more than 10 seals within a 24 hour period of fishing; or
- more than 10 seals in one shot,

the holder must:

- suspend fishing immediately;
- notify the AFMA observer onboard of the seal mortalities and with the assistance of the AFMA observer review the effectiveness of mitigation measures used in fishing operations; and
- not recommence fishing within 50 nm of the event.

Source: Dr J. Findlay, AFMA *in litt.* 19 April 2013.

In addition, the operations of large-scale mid-water trawl operations would have been subject to conditions (e) to (g) specified in 'Condition 1' of the Schedule to the accreditation of the fishery made by the Environment Minister on 3 September 2012 (see Section 3.2.4). In particular, a closure for Australian sea lions would have been imposed (Figure 5.13).

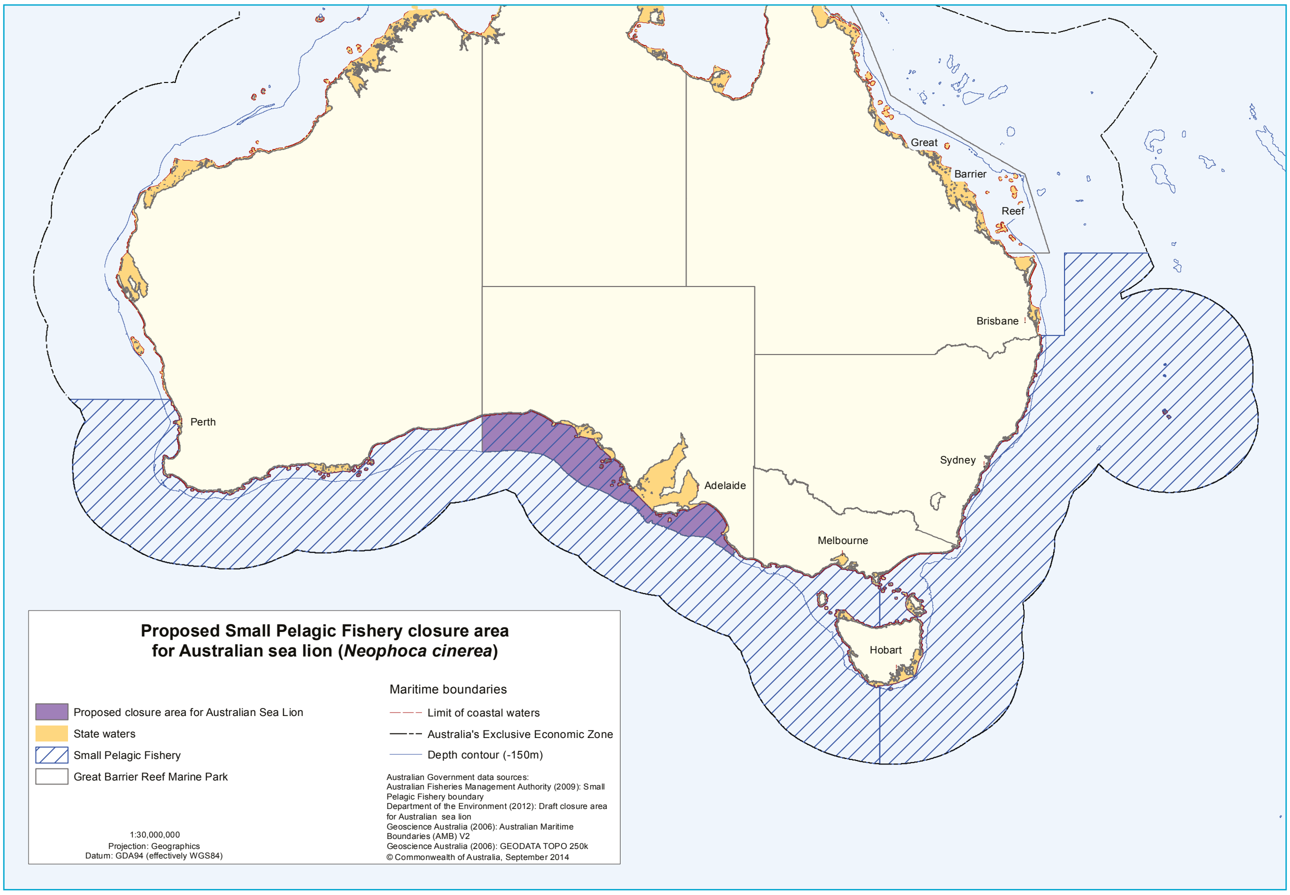

Figure 5.13 Proposed closure area for the Australian sea lion for large-scale mid-water trawl operations in the SPF. **Source:** ERIN.

5.2.4 Assessment of the likely nature and extent of direct interactions by the DCFA with pinnipeds

The panel's assessment of the likely nature of pinniped interactions with the DCFA are based on the above review and assessment of available information on the nature and extent of direct interactions between pinnipeds and trawl fisheries around the world and in Australia.

All of the breeding distribution of the Australian and New Zealand fur seals in Australia, and most of the breeding distribution of ASL, occurs within the area of or adjacent to the SPF. Seals are common marine predators in southern Australia; they are intelligent and curious animals and will be attracted to any fishing activity that occurs within their foraging range. The greater the level and frequency of fishing activity, or predictability in where and when fishing activity will occur within an area where seals forage, the greater the number of seals that are likely to be attracted to, and interact with fishing operations. This is especially the case if such interactions provide some reward. If fishing is persistent over time and fishing activities provide opportunities for seals to gain nutritional benefits, then sections of the population can become habituated to fishery interactions. This undoubtedly has happened in most trawl fisheries operating in the

SPF area, especially those in the SESSF, where persistent and ongoing bycatch interactions have been an issue for many decades (Knuckey *et al.* 2002, Hamer and Goldsworthy 2006, Tilzey *et al.* 2006, National Seal Strategy Group and Stewardson 2007, Lyle and Willcox 2008, Tuck *et al.* 2013).

The likely nature of direct pinniped interactions with the DCFA includes net feeding, entering the trawl net (during shooting, fishing and hauling), and as mentioned above, habituation to fishing activities. With these interactions, some level of bycatch mortality is inevitable and in areas of high seal abundance and/or high fishing activity, likely to be common, even with the proposed Seal and Dolphin Management Plan and a mandatory SED.

Most mid-water trawl operations that have occurred in the SPF area have been in the south-east of Australia (principally Tasmania and Bass Strait area) where the most common seals are Australian fur seals (see Figure 5.4). The major centre of the New Zealand fur seal population in Australia is off SA, with approximately 80,000 occurring in a relatively small geographic area between Kangaroo Island and the southwestern Eyre Peninsula (Figure 5.4). Any mid-water or demersal trawl fishery operating in shelf waters adjacent to these areas is likely to encounter high levels of interactions. The other main population centre of fur seals is in the Recherche Archipelago off the south coast of WA (Figure 5.4). Again, in the panel's view, seal interactions with fishing activities would be common if a trawl fishery was to operate in this region (Figure 5.4).

As summarised above, the information detailing the nature and extent of interactions between fur seals and trawl fisheries operating within the area of the SPF, indicate that where trawl fisheries and fur seals overlap, interactions will occur. However, given the limited historic and independently observed mid-water trawl activity in areas outside south-east Australia within the SPF, especially in regions off SA and WA, there is uncertainty in the likely nature and extent of interactions between Australian sea lions and the DCFA, if fishing operations were to occur there. Even though the limited dietary information suggests Australian sea lions tend not to forage on pelagic fish, like Australian fur seals, they are primarily benthic and opportunistic foragers. Furthermore, demersal foraging New Zealand sea lions readily interact with mid-water and demersal trawling operations in New Zealand. Therefore, if the DCFA were to operate in parts of the Australian sea lion range, the panel considered that some animals would interact with the fishery and that some level of bycatch is likely.

Panel assessment: likely nature and extent of direct interactions by the DCFA with pinnipeds

- *Seals occur throughout the entire area of the SPF. They are abundant and conspicuous marine predators and will be attracted to any fishing activity that occurs within their foraging range.*
- *They readily interact with all trawl fisheries in southern Australia, including the mid-water trawl fishery in the SPF.*
- *The greater the level and frequency of fishing activity, or predictability in where and when fishing activity will occur within an area where seals forage, the greater the number of seals that are likely to be attracted to, and interact with fishing operations.*
- *If fishing is persistent in an area over time, and fishing activities provide opportunities for seals to gain nutritional benefits, then sections of the population can become habituated to fishery operations and this may lead to an increase in interactions.*
- *The nature of these interactions with the DCFA would likely include net feeding, entering the trawl net (during shooting/hauling), habituation to fishing activities and bycatch.*
- *Some level of direct interactions with seals, including bycatch mortality, is inevitable and in areas of high seal abundance, likely to be common, even with current best practice mitigation devices and fishing behaviour.*
- *Historically, most mid-water and demersal trawl operations that have occurred in the SPF area have been in the south-east of Australia where most interactions are with Australian fur seals.*
- *If the DCFA were to operate in areas where threatened ASL occur, some level of direct interactions with this species, including bycatch mortality, is inevitable.*

- *There is uncertainty about the nature and extent of interactions with pinnipeds if the DCFA were to fish off SA and WA. In these regions, New Zealand fur seals and ASL are most common. Neither species has been exposed to the level of bycatch mortality from trawl fisheries experienced by Australian fur seals, so there is uncertainty about the differential impacts of bycatch on their populations. This is especially significant for the threatened ASL.*

5.2.5 Assessment of the effectiveness of proposed management measures to mitigate pinniped interactions in the DCFA

The degree to which any of the proposed management measures for a large mid-water trawl freezer vessel in the SPF would have been effective in mitigating interactions with pinnipeds is highly uncertain, largely because a vessel of the configuration of the *FV Abel Tasman* has never fished in Australian waters. As a result, the potential effectiveness of the proposed management arrangements has never been tested. The panel's assessment of the likely effectiveness, is therefore, based on a review and assessment of national and international experience documented in the literature and identified from discussions with experts on similar vessels and fishing operations around the world. The panel's assessment of each component of the proposed management arrangements is provided below.

Seal excluder device

The Seal and Dolphin VMP (Box 5.1) required the use of a top-opening SED with a hood. The SED proposed to be used on the *FV Abel Tasman* was designed by Maritiem, and was composed of a soft fibre grid made of a flexible and strong material called Dyneema twine (Maritiem 2012 in Elgin Associates unpublished (b)) (Figure 5.14). Dyneema has the same strength as steel for the same diameter and does not stretch (Maritiem 2012 in Elgin Associates unpublished (b)). A hard SED (e.g. constructed of steel bars) was not considered practical because it would not withstand the forces applied to the trawl (particularly during shooting and hauling), and because the *FV Abel Tasman* used a net drum and did not have ramp hauling and potential stowage of the SED trawl (i.e. the SED would bend out of shape if winched onto the net drum). The mesh size proposed for the soft grid was 200 by 200 mm, which previous research has indicated was adequate for preventing marine megafauna (including pinnipeds) passing through to the codend. The SED was proposed to be positioned between the intermediate (or conical) part of the trawl and the straight cylinder part of the trawl, approximately 50 m from the end of the codend (Maritiem 2012 in Elgin Associates unpublished (b)).

The proposed angle of the SED was between 15° and 25° (if parallel to the seams of the codend is 0° and perpendicular to the seams (vertical) is 90°). The small angle was chosen to increase the grid length, to improve the capacity of the grid to allow target species to pass through to the codend (Maritiem 2012 in Elgin Associates unpublished (b)). The SED was proposed to be top opening, with a cover (hood) held up by floats, which, when in operation would have an angle of approximately 45° (Maritiem 2012 in Elgin Associates unpublished (b)) (Figure 5.14). This configuration was considered optimal for preventing loss of target species as well as retaining megafauna (including seals) that do not make it out of the trawl, enabling the monitoring of mortalities (Maritiem 2012 in Elgin Associates unpublished (b)). Additional flotation was to be used around the hood so that a camera could be installed on the top of the panel to monitor SED performance. Due to the first Final Declaration, this SED was not trialled on the *FV Abel Tasman*.

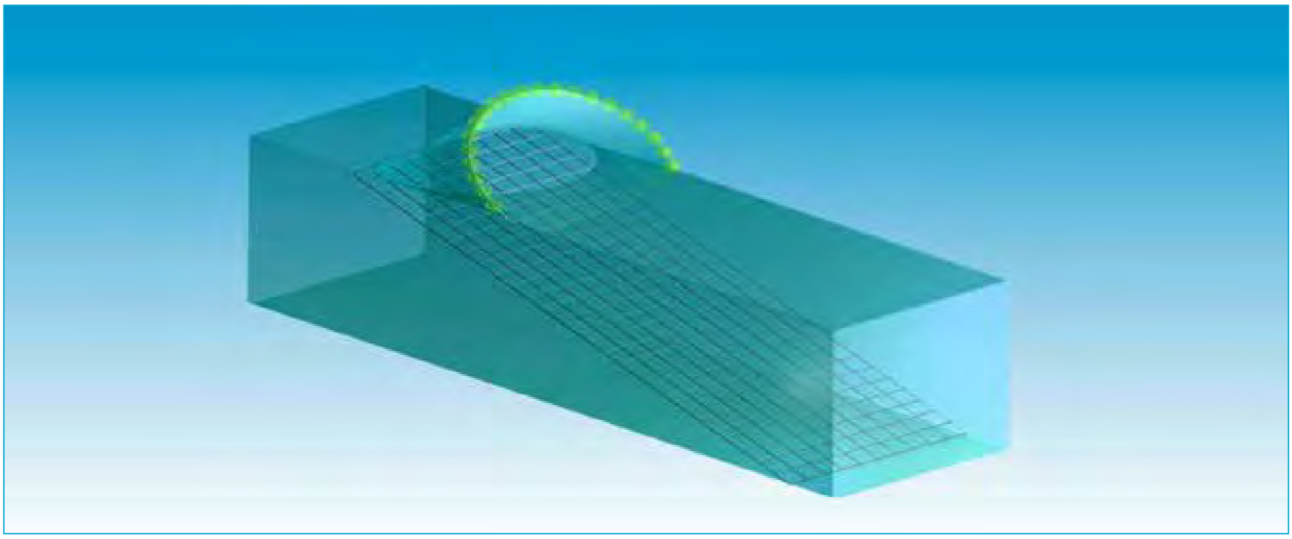

Figure 5.14 Schematic of the SED proposed to be used on the *FV Abel Tasman* in the SPF. **Source:** Maritiem 2012 in Elgin Associates unpublished (b), reproduced with permission from Seafish Tasmania Pty Ltd.

In addition to a SED, an auto-trawl system was planned to be utilised on the *FV Abel Tasman*. An auto-trawl system is controlled by telemetry and sensors maintain the shape of the trawl net when turning so that the net never closes up (Elgin Associates unpublished (b)). Ensuring the trawl mouth is always open is considered important in maintaining the effectiveness of the SED and in reducing marine mammal bycatch mortality. However, there is currently no evidence that demonstrates the efficacy of auto-trawl equipment in minimising bycatch mortality of marine mammals (Elgin Associates unpublished (b)).

The panel noted that SEDs are not commonly used internationally, and they are mainly utilised in New Zealand and Australian fisheries to mitigate interactions with otariid seals (fur seals and sea lions). SEDs need to be designed specifically for each fishery, taking into account the particular characteristics of the size of target species, gear type, fishing operation, the size and operation of gear, towing speed, the hydrodynamics of trawl set up in relation to scaling (trawl size/grid and escape hole ratios), how trawl nets are stored on the vessel, and bycatch species to be excluded (Elgin Associates unpublished (b)). For each new fishery or vessel to which a SED is being utilised, it is typical for there to be a developmental period during which SED design is modified and tailored to improve efficacy (Tilzey *et al.* 2006, Lyle and Willcox 2008).

The panel expects that had the DCFA commenced fishing operations, it is likely that there would have been operational and efficacy issues with the proposed SED design, and that, as experienced in other fisheries, a period of optimisation would have been required in order to identify a particular SED design that worked most effectively for the fishery, fishing vessel and bycatch species.

A SED functioning under optimal operating conditions will significantly reduce the incidence of bycatch mortality of pinnipeds, but will not eliminate it. The panel noted that the proposed top-opening SED with a hood enhances the escape of pinnipeds, but reduces the incidence of observed bycatch by on-board observers, because a proportion of seal mortalities may not be retained by the hood. Hoods likely enhance the retention of seal mortalities but are not 100 per cent effective, leading to potential for unobserved 'cryptic' mortality. The proposed utilisation of underwater video monitoring by the *FV Abel Tasman* would provide an essential tool to monitor SED efficacy and cryptic mortality.

The panel is aware that all trawl fisheries where SEDs have been utilised in recent years have universally used a rigid SED composed of a metal grid (New Zealand, Antarctica, Australia SET and SPF). In the one case where a soft grid was used (SPF mid-water trawl fishery), it was found to be less effective in directing seals towards the escape hole and replaced subsequently with a rigid SED design (Browne *et al.* 2005, Lyle and Willcox 2008). It is unclear if the soft-grid SED fabricated from Dyneema twine proposed to be used on the *FV Abel Tasman,* would have functioned as well as a hard metal grid in directing seals toward the escape hatch.

Fishing behaviour

The only activities detailed in the proposed seal and dolphin management plan (see Box 5.1 above) that constitutes a specific change in fishing behaviour to mitigate pinniped interactions is the requirement to "not deploy any trawl nets if seals are sighted within 300 m of the boat by any crew member or onboard observers, until the seals have dispersed of their own accord or the boat has steamed away and [sic] are no longer within 300 m of the boat".

The efficacy of this practice is uncertain. In the SPF most trawling activities commence in the evening when pelagic fish begin to school (Lyle and Willcox 2008). As most of the seal interactions in the SPF off Tasmania occurred between 1800 and 0700 hours, it is questionable whether the activity of seals within 300 m of the vessel could have been detected readily in twilight or darkness.

The panel noted that the proposed seabird vessel management plan for the large mid-water trawl freezer vessel (see Box 5.1) required that all biological material must be retained and that there should be no discard into the water while the gear is in the water. The panel noted that similar measures have been used elsewhere for this purpose (see Section 5.2.3) and considers that this requirement will also assist in reducing interactions with pinnipeds.

Bycatch trigger limits/move-on rules

A move-on rule was proposed as a key management arrangement for the large mid-water trawl freezer vessel. This required suspension of fishing and for the vessel to move at least 50 nm away if three or more seal mortalities occurred in each of three consecutive shots, or more than 10 fur seal mortalities in one shot or day. On advice from the Department of the Environment (Mr N Hanna, Department of the Environment *in litt.* 23 May 2014) the panel has concluded that this proposed restriction related only to fur seals and not to threatened Australian sea lions, noting that a closure for Australian sea lions had been proposed.

The rationale for this move-on rule and the number of fur seals permitted to be caught before it is triggered is unclear. This rule would require no change in fishing operations as long as no more than 10 fur seal mortalities occurred per day. The panel noted that three or more seals in each of three consecutive shots or 10 seals in one tow, is equivalent to a minimum bycatch rate of 300-plus seals per 100 tows. These rates would be one to two orders of magnitude higher than the mean rates observed previously in the SPF mid-water trawl fishery, in the winter blue grenadier fishery and in the wet boat sector of the SET (see Section 5.2.2). The panel considered that if such bycatch rates were occurring consistently under the DCFA, they would suggest that either the SED was ineffective, and/or that fishing activity was being conducted without due care, and/or in areas where seal density is too great to enable sustainable fishing activity to occur.

The panel considered that a permitted mortality of up to 10 fur seals per day of fishing was too high and questioned how consistent this provision is with Part 13 of the EPBC Act which requires fishing operators to take all reasonable steps to avoid killing or injuring listed marine species.

Therefore, the panel considered that any seal vessel management plan for the DCFA should include:

- *a reduction in the daily and per-shot trigger limit* to ensure that fishing operations are in compliance with the EPBC Act requirement to take all reasonable steps to avoid killing or injuring listed marine species
- *an acceptable maximum mean bycatch rate trigger limit for a fishing season and/or management zones* to ensure that average per-shot bycatch rates remain below acceptable levels and consistent with the EPBC Act requirement that fishing operators take all reasonable steps to avoid killing or injuring listed marine species by encouraging operators to move away from fishing areas of high seal density if their average bycatch rates were increasing, or close to, the maximum bycatch rate trigger limit (consideration of bycatch limits within regions/zones may be needed to prevent disproportional impacts on individual seal populations if fishing activities were concentrated in certain areas).

Further research needs to be undertaken to determine what levels of fishery-related mortality can be sustained by pinniped populations, and what the appropriate permissible level of bycatch mortality should be within the SPF (see Section 5.2.6).

The panel is also uncertain that if the trigger limits were exceeded, whether moving fishing operations at least 50 nm away is adequate or appropriate. This distance appears arbitrary and not evidence based. It is possible that in randomly moving a set distance, fishing operations could move to an area with higher seal densities where interactions are more likely than they were previously. The panel considered that a requirement to move to an area where interactions with seals are less likely, would provide a better response, but would need to be underpinned by available data on estimated at-sea density distributions (e.g. Figures 5.4, 5.5). Furthermore, if bycatch limits were applied to zones, then move-on rules would only need to be applied if zone triggers were exceeded.

As the proposed trigger limits and move-on rules only apply to fur seals, the panel noted that under the proposed management arrangements for the DCFA, there would be no restriction on fishing activity in the event of bycatch interactions with threatened ASL. Although an 'Australian sea lion closure area' in waters out to 150 m depth off SA was proposed by the Department of the Environment as a condition on the operations of the large mid-water trawl operations (see Section 5.2.3), no limit was proposed to be placed on the number of ASL that could be taken outside this closure.

In the GHAT Fishery off SA, Australian sea lion bycatch trigger limits (ranging from one to five sea lions per 18-month period), exist across seven fishing zones out to 183 m depth (AFMA 2013d). However, there would be no limit on the number of ASL permitted to be taken by the proposed large mid-water trawl freezer vessel in the 'gap' between the depth closure proposed for that vessel and the GHAT Fishery ASL closure area. Although available information indicates that most foraging occurs in waters less than 150 m depth (especially for adult females), some sea lions have been recorded foraging in depths up to 250 m.

The panel considered that the absence of trigger limits for ASL under the proposed arrangements for the large mid-water trawl freezer vessel in the SPF and inconsistencies between those arrangements and the trigger limits imposed on GHAT fishers where these fisheries overlap, are deficiencies in the proposed management measures.

Spatial closures

As noted above, Condition 1 part (e) of the Schedule to the accreditation of the fishery made by the Environment Minister on 3 September 2012, specified one spatial closure to mitigate pinniped interactions in the DCFA. The proposed 'Australian sea lion closure area' closed all shelf waters off SA, out to a depth of 150 m to fishing activity by the large mid-water trawl freezer vessel. Presumably this closure was put in place to mitigate the potential for bycatch mortalities of Australian sea lions, especially adult females, by the vessel if it fished in areas of overlap with ASL foraging effort on shelf waters off SA (see Figure 5.3). While the panel considered that this management measure would have provided significant protection for the South Australian populations of ASL it was unclear why such protection was not afforded to populations of ASL off the south coast of WA that also occur in the SPF area.

The panel noted that the proposed arrangements did not provide for any specific spatial closures to mitigate bycatch interactions with fur seals. The panel considered that central place foraging, lactating adult fur seal females may warrant similar protection.

Panel assessment and advice: assessment of proposed measures and actions to avoid, reduce and mitigate direct interactions of the DCFA with pinnipeds

Assessment: effectiveness of proposed measures

- *The proposed top-opening SED with a hood enhances the escape of pinnipeds, but reduces the incidence of bycatch observed by on-board observers. Hoods enhance the retention of seal mortalities but are not 100 per cent effective, leading to unobserved 'cryptic' mortality. The proposed utilisation of underwater video monitoring would be essential to monitor SED efficacy and cryptic mortality under the DCFA.*
- *The panel questions the effectiveness of the proposed requirement to halt deployment of the net if seals are sighted within 300 m of the vessel, given that most fishing activities in the SPF are likely to occur at night and it is questionable whether the activity of seals within 300 m of the vessel could be detected readily in twilight or darkness. In the panel's view, effectiveness is further compromised by the seals' ability to move quickly in and out of a 300 m range.*

- *The panel supports the inclusion of a prohibition on the discard of biological waste from the DCFA as a means of avoiding interactions with pinnipeds.*
- *The panel considered the requirement to suspend fishing immediately when a seal mortality is detected. However, it noted that it is likely that some mortalities will not be immediately known to the crew or the observer, but may subsequently be identified on review of video recordings which would reduce the efficacy of this requirement, i.e. by the time the mortality has been identified the vessel will no longer be in the area.*
- *The panel considered that the permitted mortality of up to 10 fur seals per day of fishing under the proposed arrangements was too high and that the 50 nm distance move-on rule was arbitrary and not evidence based.*

<u>*Advice: actions to avoid, reduce and mitigate adverse environmental impacts of the DCFA*</u>

- *Use a SED, with or without auto trawl, only after its operation has been optimised for the vessel, fishery and bycatch species under a scientific permit with the required level of performance developed in consultation with experts. For example, the panel noted that neither the soft mesh-grid, top-opening SED with hood, nor the auto trawl system proposed to be used by the FV Abel Tasman to mitigate pinniped bycatch, has undergone trials in the SPF.*
- *Use underwater video to monitor the SED efficacy and cryptic mortality.*
- *Reduce the daily and per-shot trigger limits on fur seals from the proposed limit of up to 10 per day and replace the associated 50 nm move-on rule with a requirement to move to an area where interactions with seals are less likely, based on available data on estimated at-sea density distributions*
- *Introduce a bycatch rate trigger limit for fur seals for the fishery or fishing areas, or a total mortality trigger for a fishing season and/or fishing areas.*
- *Ensure 100 per cent observer coverage of fishing operations and, if daily or per shot trigger limits are used in conjunction with move-on rules or with a requirement to review mitigation measures, provide sufficient observer capacity to ensure that underwater video footage is monitored at the end of each shot to maximise response times to mortalities.*
- *Require 'stickers' to be removed from the net before shooting, noting that this was a requirement of the proposed seabird VMP.*
- *Prohibit the discard of any biological waste (excluding the release of any protected fauna) noting that this was a requirement of the proposed seabird VMP.*
- *Implement spatial closures that mitigate bycatch interactions with fur seals, especially in regions adjacent to breeding colonies where there is high transit and foraging activity by central place foraging lactating adult females.*
- *Review the proposed Australian sea lion closure area off South Australia (out to 150 m depth) so as to provide consistency with the management arrangements for the GHAT Fishery (out to 183 m depth).*
- *Implement a similarly designed closure for the Australian sea lion colonies occurring within the SPF off Western Australia.*

5.2.6 Monitoring and research

For global pinniped populations, as for those in Australia, the most significant source of anthropogenic mortality is from fishery interactions (Shaughnessy 1999, National Seal Strategy Group and Stewardson 2007, Kovacs *et al.* 2012). In Australia, the most significant source of fishery-related pinniped bycatch is from trawl fisheries. A fishery targeting the key prey taxa of pinnipeds in their foraging grounds and within their foraging depth range will inevitably attract many animals, and potentially (as demonstrated in the mid-water trawl fishery of the SPF to date) result in significant levels of bycatch mortality. The panel has proposed a number of ways in which direct interactions of the DCFA with pinnipeds might be mitigated. The panel has also identified four key uncertainties (questions) relating to potential adverse impacts on pinnipeds resulting from the DCFA that could be addressed through further monitoring and research. They include the following.

1) What are the individual and cumulative fishery-related bycatch impacts on pinniped populations?

Seals interact with and potentially suffer incidental mortality from a range of different fisheries. A key uncertainty in assessing the potential adverse impacts resulting from any one fishery (such as the DCFA in SPF), is the extent to which that fishery contributes to the total impacts across all fisheries.

The panel considered that improved independent monitoring of pinniped bycatch and a requirement for annual reporting of estimated take of pinnipeds by all Australian fisheries is needed. This would enable the estimation of overall cumulative impacts on pinniped populations, and enable assessment of the relative contribution of individual fishery impacts.

2) What levels of fishery-related mortality can pinniped populations sustain?

Improved pinniped population models and ongoing monitoring of status and trends in abundance would provide a means to better evaluate what levels of bycatch mortality are sustainable, and reduce uncertainties about the potential for adverse environmental impacts. It would provide essential biological context to estimates of individual and cumulative fishery impacts (addressed in question one, above), and provide a direct quantitative measure to directly assess a fishery against Part 13 of the EPBC Act which requires that "the fishery does not, or is not likely to adversely impact the conservation status of protected species or affect the survival and recovery of listed threatened species".

Such information would not only inform what bycatch levels are sustainable, but also assist in apportioning and setting allowable take and maximum bycatch rate trigger limits for individual fisheries.

3) Where are the regions of critical foraging habitat for pinniped populations where the management of direct interactions with the DCFA may be most needed?

The panel considered that research to better understand the foraging distributions and critical habitat of pinnipeds could help identify regions where management of the potential adverse environmental effects of fishing may be most needed. There are two key components to such work.

a) Knowledge of the locations of key foraging areas where adult females may be particularly vulnerable to bycatch mortality in near colony waters. Adult female fur seals and sea lions spend most of their lives raising pups. The need to return regularly ashore to nurse a dependent pup requires that females make regular foraging trips to sea to forage. Bycatch of females has a disproportionate effect on populations (loss of mother, pup on teat and one in utero and future reproductive potential) compared to males. Reducing female bycatch can help reduce uncertainties about the potential for adverse impacts on pinniped populations. Such information may inform the location and timing of spatial closures to mitigate bycatch.

b) Knowledge of the locations of foraging hot-spots (areas of very high density of animals) used by one or more populations of seals could provide important information on which areas could be avoided to reduce the incidence and rate of bycatch.

4) Are there additional modifications to fishing gear and behaviour that can reduce the potential for direct interactions by the DCFA with pinnipeds?

The panel considered that additional research and fishing trials could be undertaken to optimise the proposed SED, or trial alternate SED designs appropriate to the fishing vessel and gear to be used in a DCFA. This would include testing of appropriateness of soft vs. hard grids, optimising the slope of the grid and configuration of the escape hole, hood and kites with the objective of improving the exit of healthy seals.

On-board observers should be required to monitor seal activity both on the surface and within the net, via underwater video monitoring, so that a data base can be developed to improve the understanding of the circumstances under which seal activity and interaction increase and decrease. This would help inform and promote codes of practice to further reduce interactions and maximise survival.

Panel advice: research and monitoring to reduce uncertainties

Research that addresses the following questions could reduce uncertainties about the potential for adverse environmental impacts of the DCFA on protected pinniped species.

- *What are the individual and cumulative fishery-related bycatch impacts on pinniped populations?*
- *What levels of fishery-related mortality can pinniped populations sustain?*
- *Where are the regions of critical foraging habitat for pinniped populations where the management of direct interactions with the DCFA may be most needed?*
- *Are there additional modifications to fishing gear and behaviour that can reduce the potential for direct interactions by the DCFA with pinnipeds?*

5.3 Cetaceans

5.3.1 Cetacean species assessed

A total of 47 cetacean species are recorded to occur in Australian waters (Bannister *et al.* 1996, Ross 2006, Woinarski *et al.* 2014), and of these, 44 species are known or are likely to occur in the SPF area (Appendix 3). Of these 44 species, 42 species were assessed in the Ecological Risk Assessment for the Effects of Fishing (ERAEF) process for the mid-water trawl sector of the SPF (Daley *et al.* 2007b). The two additional cetacean species recorded to occur in the SPF region (but not assessed in the ERAEF) are Omura's whale *Balaenoptera omurai* and spectacled porpoise *Phocoena dioptrica* (Woinarski *et al.* 2014). The ERAEF Level 2 PSA analysis identified a total of 20 threatened, endangered and protected cetacean species as High risk, a further 21 cetacean species as Medium risk, and one cetacean species as Low risk (Appendix 3). After Level 2 Residual Risk Guidelines were applied, seven cetacean species remained at High risk for the mid-water trawl sector of the SPF (AFMA 2010b). These are:

- Risso's dolphin *Grampus griseus*
- Fraser's dolphin *Lagenodelphis hosei*
- hourglass dolphin *Lagenorhynchus cruciger*
- southern right whale dolphin *Lissodelphis peronii*
- striped dolphin *Stenella coeruleoalba*
- Indo-Pacific bottlenose dolphin *Tursiops aduncus*
- common bottlenose dolphin *Tursiops truncatus*.

The 21 cetacean species detailed below include these seven species and 13 other cetacean species known to occur in the SPF area that are recorded to have interacted with trawl fisheries in Australia and/or internationally, and were therefore considered most relevant to assessing the risks and likelihood of interactions with large mid-water trawl vessels in the SPF (Elgin Associates unpublished (a)). In addition, the spinner dolphin *Stenella longirostris* is recorded taken as bycatch in purse seine, gillnet and trawl fisheries throughout its range, so is therefore also considered relevant to this assessment. For each of these 21 species a summary is provided of their known distribution range and overlap with the SPF area, population size and trends, relevant biology and ecology, key risks and threatening processes, and their conservation and listing status. These species' summaries are arranged in order of risk, with the seven dolphin species previously assessed as high risk described first, followed by other odontocete species then mysticete whales. The distribution of trawl effort in the SPF during 2000–2013 is shown in Figure 5.15 in relation to the pattern of species richness of the most relevant cetacean species, based on the available distribution data for these species held by the Department of the Environment.

The panel's Terms of Reference include specific mention of dolphins. Short-beaked common dolphins and *Tursiops* spp. bottlenose dolphins are the dolphin species considered most likely to interact with trawl fisheries (Elgin Associates unpublished (a)), and common bottlenose dolphins and possibly short-beaked common dolphins have previously been recorded as bycatch in mid-water trawls in the SPF (Lyle and Willcox 2008, Tuck *et al.* 2013). The panel recognised that interactions could occur between the DCFA and the other 23 cetacean species that have been recorded in the SPF area, but considered these other species to be at lower risk of interaction and therefore of less relevance to this assessment of direct interactions.

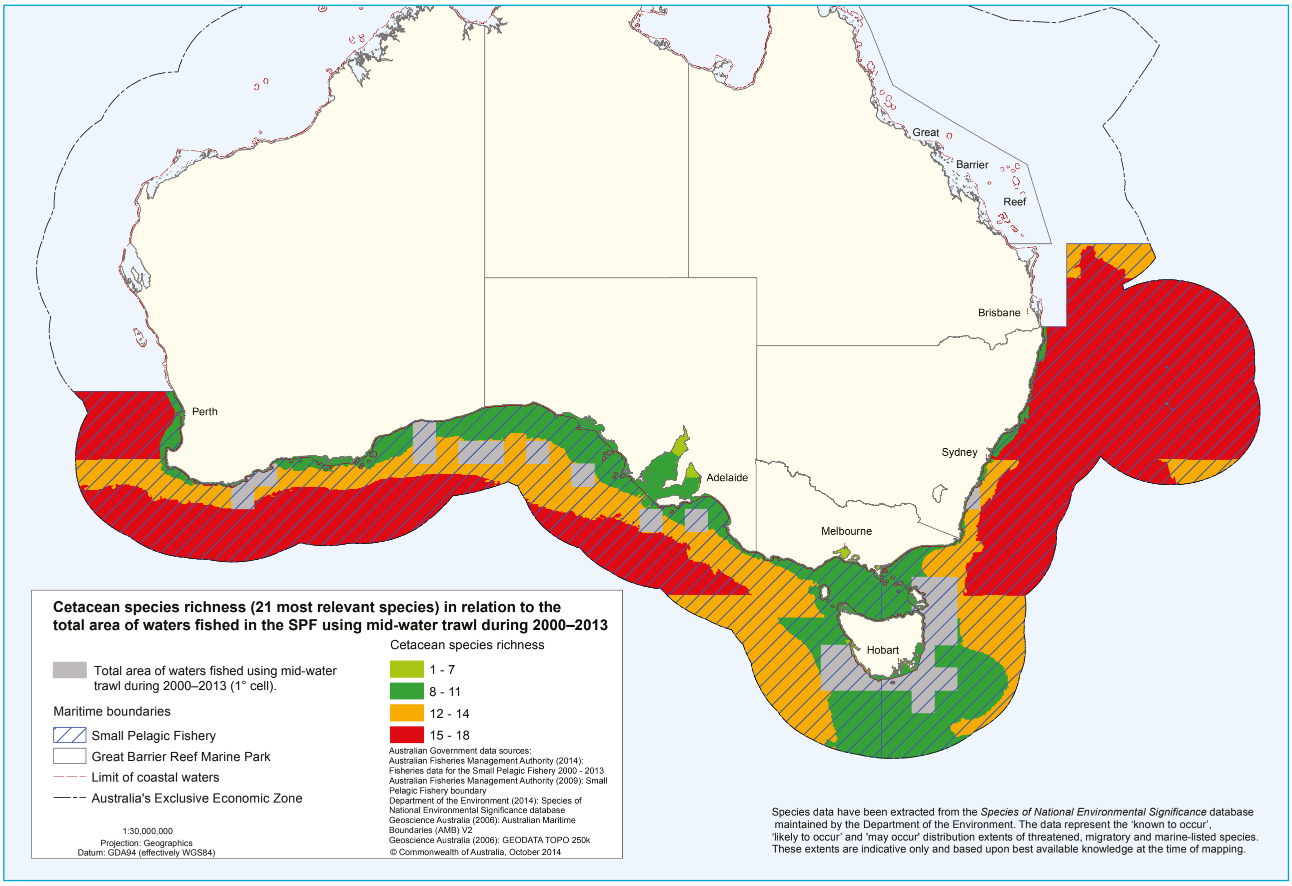

Figure 5.15 Cetacean species richness (21 most relevant species) in relation to the total area of waters fished in the SPF using mid-water trawl during 2000–2013. **Source:** Map produced by ERIN using unpublished AFMA data.

Risso's dolphin *Grampus griseus* (Level 2 PSA Residual Risk – High)

Distribution and range

Risso's dolphin is widely distributed from tropical to temperate and subantarctic regions in both hemispheres, ranging from about latitude 60°N to 60°S, but is mostly found in warmer waters within this range (Rice 1998, Baird 2009). This species has been recorded from all Australian states and Northern Territory waters, extending south to Tasmania at 43°S (Ross 2006, Warneke and Donnelly 2008). Its Australian range overlaps extensively with the SPF area.

Population size and trends

Risso's dolphins are considered to be relatively abundant throughout the main part of their Australian range (Ross 2006), but there are no estimates of the Australian or global population size or trends (Taylor *et al.* 2008a, Woinarski *et al.* 2014). Overseas, regional population estimates include about 175,000 in the eastern tropical Pacific region (Wade and Gerrodette 1993), 33,000 off the western United States coast, and 83,000 off Japan (Jefferson *et al.* 2008). There is some evidence of population structure within and between ocean basins (Baird 2009).

Biology and feeding ecology

Risso's dolphin grows to at least 3.8 m long and can weigh up to 400 kg and potentially nearer 500 kg (Jefferson *et al.* 2008). These large dolphins occur mainly in deeper water outer shelf and continental slope habitats particularly in areas of steeply sloping underwater topography and high productivity upwelling areas, and they have been sighted from inshore areas to well offshore in open pelagic habitats (Ross 2006, Jefferson *et al.* 2008, Warneke and Donnelly 2008). They occur in mostly small-to-medium-sized groups of about 4–100 dolphins, but groups of up to 4000 have been recorded (Jefferson *et al.* 2008, Warneke and Donnelly 2008). Longer-term changes in distribution patterns of Risso's dolphins off central California have been associated with oceanographic changes and movements of spawning squid (Jefferson *et al.* 2008).

These dolphins feed primarily on mid-water and bottom-dwelling squid, but also consume octopus and crustaceans, and possibly fish (Jefferson *et al.* 2008, Warneke and Donnelly 2008, Baird 2009). They appear to feed mainly at night possibly associated with diurnal vertical migrations of their prey (Jefferson *et al.* 2008, Warneke and Donnelly 2008). Diet may vary between sexes and among different age groups (Baird 2009). Females mature at about 8–10 years and males about 10–12 years, gestation is about 13–14 months, the interbirth interval is about 2.4 years, and longevity is about 34 years (Baird 2009). Generation length is estimated to be 19.6 years (Taylor *et al.* 2007).

Risks and threatening processes

Where Risso's dolphins occur in the SPF area there is a risk of incidental capture in fisheries gear (Elgin Associates unpublished (a)). Risso's dolphins have been incidentally captured in US north-east and mid-Atlantic mid-water trawl fisheries (Fertl and Leatherwood 1997, Zollett 2009, Elgin Associates unpublished (a)). Other overseas fisheries interactions include an annual drive fishery in Japan, incidental bycatch in driftnet and purse seine fisheries, and this species has been recorded taking bait from longlines, which has resulted in bycatch and instances of deliberate killing (Jefferson *et al.* 2008, Baird 2009). Other threats include ingestion of plastic, pollution resulting in high levels of contaminants in tissues, anthropogenic noise and acoustic disturbance (Bannister *et al.* 1996, Baird 2009, Woinarski *et al.* 2014). These dolphins occasionally ride bow waves of vessels which increases the risk of vessel strike, but they are considered to be mostly indifferent to vessels or avoid them (Baird 2009).

Conservation and listing status

Risso's dolphin is listed as a cetacean species under the EPBC Act, Rare in SA, Data Deficient in the Northern Territory, but is not listed in other states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014) and similarly in previous Australian status assessments (Bannister *et al.* 1996, Ross 2006). Globally, Risso's dolphin was assessed as Least Concern for the IUCN Red List in 2008 (Taylor *et al.* 2008a), and is listed in Appendix II of CITES.

Fraser's dolphin *Lagenodelphis hosei* (Level 2 PSA Residual Risk – High)

Distribution and range

Fraser's dolphin has a pantropical distribution between 30°N and 30°S (Rice 1998, Dolar 2009). Its distribution in Australian waters is poorly known, with records from NSW, Queensland and WA, and a stranding record from Corio Bay, Victoria at 38°S which is considered to be outside its normal range and possibly associated with anomalous movements of the warm East Australian Current (Bannister *et al.* 1996, Ross 2006, Dolar 2009). Its Australian range overlaps partly with the SPF area.

Population size and trends

There is no estimate of the Australian population size or trends for this species (Woinarski *et al.* 2014). Global abundance is estimated to be about 300,000, with about 289,000 individuals estimated in the eastern tropical Pacific region (Wade and Gerrodette 1993, Hammond *et al.* 2008a). The global population trend is also unknown (Hammond *et al.* 2008a).

Biology and feeding ecology

Fraser's dolphin grows to at least 2.6 m long and can weigh more than 210 kg (Jefferson *et al*. 2008). These dolphins occur mainly in deep offshore pelagic waters and along the outer continental shelf and slope, but they can occur nearer the shore where deep water occurs closer to the coast (Ross 2006, Dolar 2009). They occur in large groups containing hundreds or thousands of dolphins and are often mixed with other delphinid species (Jefferson *et al*. 2008).

Fraser's dolphins feed on mid-water myctophids and other mesopelagic fish, squid and crustaceans, and may selectively feed on larger prey (Ross 2006, Jefferson *et al*. 2008, Dolar 2009). Depth of feeding appears to vary in different regions, with records ranging from feeding near the sea surface to depths exceeding 600 m (Dolar 2009). Females mature at about five-to eight years with males maturing at 7–10 years, gestation is about 10–12 months, the interbirth interval is about two years and longevity is at least 18 years (Dixon 2008, Jefferson *et al*. 2008). Generation length is estimated to be 11 years (Taylor *et al*. 2007).

Risks and threatening processes

Where Fraser's dolphins occur in the SPF area there is a risk of incidental capture in fisheries gear (Elgin Associates unpublished (a)). Four dolphins with genetic affinities to Fraser's dolphin haplotypes were sampled in association with dolphins interacting with the Pilbara Fish Trawl Fishery off northwestern Australia (Allen and Loneragan 2010). This fishery operates between 50–100 m depth on the northwestern shelf off WA, hence these records are interesting because they indicate use of relatively shallow water shelf habitat that is unusual for this primarily deep-water species (Allen and Loneragan 2010, Jaiteh *et al*. 2013). Fraser's dolphins are hunted in Japan, Lesser Antilles and Indonesia, and have been incidentally captured in other overseas fisheries including as incidental bycatch in purse seines, gillnets, driftnets, trap nets and anti-shark nets (Jefferson *et al*. 2008, Dolar 2009). Other threats include pollution, anthropogenic noise and acoustic disturbance (Bannister *et al*. 1996, Woinarski *et al*. 2014).

Conservation and listing status

Fraser's dolphin is listed as a cetacean species under the EPBC Act, and the 'Southeast Asian population' is listed as migratory under the EPBC Act, but the species is not listed by states within its Australian range (Woinarski *et al*. 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al*. 2014) and similarly in previous Australian status assessments (Bannister *et al*. 1996, Ross 2006). Globally, Risso's dolphin was assessed as Least Concern for the IUCN Red List in 2008 (Hammond *et al*. 2008a), and is listed in Appendix II of CITES.

Hourglass dolphin *Lagenorhynchus cruciger* (Level 2 PSA Residual Risk – High)

Distribution and range

Hourglass dolphins have a circumpolar distribution restricted to higher latitudes in the Southern Hemisphere from about 33°S down to the ice edge around 67°S, with most records from 45°S to 65°S (Rice 1998, Goodall 2009). Its distribution in Australian waters is not well known, with most sightings occurring south of Australian mainland waters around subantarctic Heard Island and Macquarie Island, and at 55°S to the south of Australia (Bannister *et al*. 1996, Thiele and Gill 1999, Ross 2006). Based on the rarity of sightings, Goodall (2008) considered that Australian waters could be considered to be the extreme distribution limits for the hourglass dolphin. Most of its Australian range is thought to lie within the Australian EEZ around subantarctic Heard Island and Macquarie Island (Woinarski *et al*. 2014); hence the distribution range of this species may only overlap marginally with the SPF area south of Tasmania.

Population size and trends

There are no estimates of the Australian or global population size or trends for this species (Hammond *et al*. 2008b, Woinarski *et al*. 2014). Abundance was estimated to be about 144,000 individuals to the south of the Antarctic convergence (Jefferson *et al*. 2008).

Biology and feeding ecology

Hourglass dolphins grow to 1.8–1.9 m long and weigh up to 88–100 kg (Jefferson *et al.* 2008, Goodall 2009). These small dolphins occur mainly in deep open ocean pelagic waters with some sightings and strandings from shallower waters near islands and banks (Jefferson *et al.* 2008, Goodall 2009). Remarkably little is known about these dolphins and they are considered to be one of the most poorly known small cetacean species (Jefferson *et al.* 2008). They have been recorded in small groups containing one to eight dolphins with some larger groups up to about 60 animals, and are often seen associated with fin whales and some other baleen whales, bottlenose whales and some other delphinid species (Jefferson *et al.* 2008, Goodall 2009).

There are few records of their diet, but stomach contents indicate that these dolphins feed on myctophids and other small fish, small squid and crustaceans (Goodall 2009). Stomach contents from one dolphin indicate that it had fed in surface waters (Goodall 2009). Very little is known about reproduction and life history of this species (Jefferson *et al.* 2008).

Risks and threatening processes

If hourglass dolphins occur in the SPF area there is a risk of incidental capture in fisheries gear (Elgin Associates unpublished (a)). Hourglass dolphins appear to be attracted to ships but there are few records of ship strike (Goodall 2009). Three female hourglass dolphins were incidentally taken in a gillnet operation from New Zealand, and one dolphin was taken in an experimental drift net in the southern Pacific Ocean (Jefferson *et al.* 2008, Goodall 2009). This species has not been recorded to interact with trawl fisheries. Potential threats include fisheries impacts on prey species, pollution, and climate and oceanographic change (Bannister *et al.* 1996, Woinarski *et al.* 2014).

Conservation and listing status

The hourglass dolphin is listed as a cetacean species under the EPBC Act. This species was previously assessed as No category assigned but possibly secure in Australian waters by Bannister *et al.* (1996) and Ross (2006), but was assessed as Least Concern by Woinarski *et al.* (2014) on the basis that it is unlikely to meet or approach any criteria for listing as threatened, and there is no evidence of decreasing population size or significant threats. Globally, the hourglass dolphin was assessed as Least Concern for the IUCN Red List in 2008 (Hammond *et al.* 2008b), and is listed in Appendix II of CITES.

Given that the Australian range of this dolphin species may only marginally overlap with the SPF area and these dolphins have not been recorded interacting with trawl fisheries, and the species is not obviously threatened and is assessed as Least Concern in Australian waters and globally, the panel did not consider it to be a high risk species for direct interactions associated with the SPF mid-water trawl sector.

Southern right whale dolphin *Lissodelphis peronii* (Level 2 PSA Residual Risk – High)

Distribution and range

The Southern right whale dolphin has a circumpolar distribution in cool temperate to subantarctic waters of the Southern Hemisphere, mostly between 25–30°S and 55–65°S (Rice 1998, Lipsky 2009). In Australian waters, most records are south of 37°S and offshore south of Tasmania, the GAB and southwestern WA, with five single stranding records from eastern Tasmania and southern NSW (Bannister *et al.* 1996, Ross 2006, Warneke 2008). Its Australian range overlaps extensively with the SPF area.

Population size and trends

There are no estimates of the Australian or global population size or trends for this species (Hammond *et al.* 2008c, Woinarski *et al.* 2014). Southern right whale dolphins are considered to be fairly common off the South Island of New Zealand, in the Tasman Sea and in waters south of Australia (Van Waerebeek *et al.* 2010).

Biology and feeding ecology

Southern right whale dolphins grow to at least 3.0 m long and can weigh up to 116 kg (Jefferson *et al.* 2008). These slender dolphins are poorly known but occur mainly in deep offshore pelagic waters and along the outer continental shelf and slope, or inshore in deep water (Ross 2006, Lipsky 2009). They are highly gregarious and can form large and active groups containing up to a thousand dolphins and are often associated with other delphinid species (Jefferson *et al.* 2008).

Southern right whale dolphins feed on a variety of myctophids and other mesopelagic fish, squid and some crustaceans, with euphausiids considered a potential food source (Ross 2006, Jefferson *et al.* 2008). They are considered to be capable of diving to depths exceeding 200 m for feeding (Jefferson *et al.* 2008). Almost nothing is known about the reproductive biology of these dolphins or their subpopulation structure or status (Hammond *et al.* 2008c, Jefferson *et al.* 2008). Age at first reproduction is possibly about 12 years, and generation length is estimated to be 18.3 years (Taylor *et al.* 2007).

Risks and threatening processes

Where southern right whale dolphins occur in the SPF area there is a risk of incidental capture in fisheries gear (Elgin Associates unpublished (a)). Southern right whale dolphins have been hunted off Peru and Chile, and incidentally captured in overseas fisheries including as bycatch in gillnet fisheries (Jefferson *et al.* 2008, Lipsky 2009). Potential threats include fisheries impacts on prey species, pollution, and climate and oceanographic change (Bannister *et al.* 1996, Woinarski *et al.* 2014).

Conservation and listing status

The southern right whale dolphin is listed as a cetacean species under the EPBC Act, but the species is not listed by states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014) and similarly in previous Australian status assessments (Bannister *et al.* 1996, Ross 2006). Globally, this species was assessed as Data Deficient for the IUCN Red List in 2008 (Hammond *et al.* 2008c), and is listed in Appendix II of CITES.

Striped dolphin *Stenella coeruleoalba* (Level 2 PSA Residual Risk – High)

Distribution and range

The striped dolphin is widely distributed from tropical to warm temperate regions ranging from about 50°N to 40°S (Rice 1998, Jefferson *et al.* 2008). This species has been recorded from WA south to Augusta, and from southern Queensland and NSW (Bannister *et al.* 1996, Ross 2006). Its Australian range overlaps partly with the SPF area.

Population size and trends

There are no estimates of the Australian population size or trends for this species (Woinarski *et al.* 2014). Globally, this species is abundant with estimates of more than a million individuals in the eastern tropical Pacific region, more than 570,000 in the northwest Pacific, and more than 200,000 in the Mediterranean Sea (Jefferson *et al.* 2008, Hammond *et al.* 2008d). The global population trend is unknown (Hammond *et al.* 2008d).

Biology and feeding ecology

Striped dolphins grow to about 2.6 m long and can weigh up to 156 kg (Jefferson *et al.* 2008, Archer 2009). These dolphins occur mainly in deeper water habitats from the continental slope out to oceanic areas particularly in high productivity upwelling areas, and occur nearer the shore where deep water occurs closer to the coast (Jefferson *et al.* 2008, Archer 2009). They occur in mostly medium-to-large-sized groups of about 30–500 dolphins, but some very large groups of a few thousand animals have been recorded (Jefferson *et al.* 2008).

These dolphins feed on a wide variety of small, mid-water, benthopelagic or pelagic fish species including myctophids, cod and anchovy but they also consume squid (Jefferson *et al.* 2008, Archer 2009). They are considered likely to dive to depths of 200 to 700 m for pelagic or benthopelagic feeding, and may forage on some diurnally migrating prey at night (Ross 2006, Archer 2009). Females are sexually mature between 5–13 years and males from 7–15 years, interbirth interval is two to four years, and gestation is 12–13 months (Ross 2008, Archer 2009). Striped dolphins are thought to have a polygynous mating system (a male mates with more than one female) (Jefferson *et al.* 2008). Maximum age is recorded as 58 years (Ross 2006) and generation length is estimated to be 22.5 years (Taylor *et al.* 2007).

Risks and threatening processes

Where striped dolphins occur in the SPF area there is a risk of incidental capture in fisheries gear. One striped dolphin was recorded as incidental bycatch in the Taiwanese drift gillnet fishery in northern Australian waters during 1974–1986 (Harwood and Hembree 1987), and these dolphins may be incidentally captured in nets off WA (Ross 2006). Striped dolphins have also been incidentally captured in a wide range of fisheries gear including trawl nets throughout their range overseas, particularly in purse seine and driftnet fisheries (Fertl and Leatherwood 1997, Jefferson *et al.* 2008, Zollett 2009, Archer 2009). These dolphins are also taken in harpoon and drive fisheries in Japanese waters resulting in serious depletion of these populations (Jefferson *et al.* 2008). Directed catches of striped dolphins for food or protection of fishing gear occur in some other regions overseas (Archer 2009). Other threats include pollution resulting in high levels of contaminants in tissues, anthropogenic noise and acoustic disturbance (Bannister *et al.* 1996, Archer 2009, Woinarski *et al.* 2014). These dolphins often ride bow waves of vessels which increases the risk of vessel strike (Ross 2006), except in the eastern tropical Pacific region where they tend to rapidly move away from vessels (Jefferson *et al.* 2008).

Conservation and listing status

The striped dolphin is listed as a cetacean species under the EPBC Act, but is not listed by states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014) and similarly in previous Australian status assessments (Bannister *et al.* 1996, Ross 2006). Globally, the striped dolphin was assessed as Least Concern for the IUCN Red List in 2008 (Hammond *et al.* 2008d), and is listed in Appendix II of CITES.

Indo-Pacific bottlenose dolphin *Tursiops aduncus* (Level 2 PSA Residual Risk – High)

Distribution and range

Indo-Pacific bottlenose dolphins have a wide but discontinuous distribution from tropical to warm temperate coastal regions ranging from southern Africa to the Red Sea and eastwards to China and southern Japan, through south-east Asia and southward to New Guinea, Australia and New Caledonia (Ross 2006, Jefferson *et al.* 2008, Wang and Yang 2009). In Australian waters, this species has an extensive coastal distribution from eastern, northern and western Australian regions and some parts of southern Australia. Therefore, the southern Australian range of this species overlaps partly with the SPF area.

However, the full Australian range of this species is uncertain due to difficulties in identifying which species of bottlenose dolphin is present in some regions (Ross 2006, Woinarski *et al.* 2014). Smaller inshore coastal forms of bottlenose dolphins are usually regarded or identified as *T. aduncus*, whereas the larger and primarily offshore forms are referred to as *T. truncatus*, the common bottlenose dolphin (e.g. Hale *et al.* 2000, Kemper 2004, Ross 2006). Ross (2006) considered that *T. aduncus* occurs around the whole Australian mainland coast primarily in inshore waters and bays, and in parts of the northern coast of Tasmania. However, *T. truncatus* occurs sympatrically or possibly replaces *T. aduncus* in some southern Australian areas, and the taxonomic status of bottlenose dolphins in parts of Tasmania and the southern and western Australian coast remains uncertain (e.g. Hale *et al.* 2000, Kemper 2004, Ross 2006, Krützen and Allen 2008, Woinarski *et al.* 2014).

Population size and trends

Abundance estimates are available for some subpopulations of this species in Australian locations but there are no robust estimates of total population size or trends in Australian waters or globally (reviewed in Ross 2006, Hammond *et al.* 2008e, Wang and Yang 2009, Woinarski *et al.* 2014). In coastal regions within or adjacent to the SPF area, abundance estimates of subpopulations range from dozens to hundreds of dolphins (reviewed in Woinarski *et al.* 2014). For example, Indo-Pacific bottlenose dolphin abundance offshore from North Stradbroke Island in southern Queensland was estimated to be 861 (± standard error (SE) 137) in 1997 and 895 (± SE 74) in 1998 (Chilvers and Corkeron 2003).

In NSW waters, repeated surveys between 2003 and 2005 in the Byron Bay and Ballina region provided an abundance estimate of 865 (confidence interval (CI) 95 per cent 861–869) dolphins (Hawkins 2007), with average group sizes of 21 for female-calf groups and smaller adult-only groups (Hawkins and Gartside 2008). Repeated surveys from 2003 to 2006 provided abundance estimates of 34 (95 per cent CI 19–49) dolphins in the Richmond River estuary near Ballina, and 71 (95 per cent CI 62–81) dolphins in the larger Clarence River estuary further south (Fury and Harrison 2008). In Port Stephens in central NSW, minimum abundance of these dolphins was estimated to be 160 (95 per cent CI = 148–182) in 1998–99 and 143 (95 per cent CI = 132–165) in 1999–2000, with about 90 resident individuals that are genetically differentiated from adjacent coastal communities (Möller *et al.* 2002, 2007, Wiszniewski *et al.* 2010). Abundance estimates in Jervis Bay in southern NSW varied from 108 (95 per cent CI = 98–128) dolphins in 1997–98, to 61 (95 per cent CI = 58–72) dolphins in 1998–99 (Möller *et al.* 2002).

In the Port Adelaide River–Barker Inlet estuary near Adelaide in SA, 75 Indo-Pacific bottlenose dolphins were identified during surveys in 2006 and 2009–10, and about 30 dolphins are thought to be resident in this area, with some additional transient dolphins irregularly visiting the estuary (Cribb *et al.* 2013). In southern Western Australian waters, population estimates from the Bunbury region varied from 65 (95 per cent CI = 54–90) dolphins in winter 2007, to 139 (95 per cent CI: 134–148) dolphins in autumn 2009 (Smith 2012). A small resident community of about 17–18 Indo-Pacific bottlenose dolphins has been recorded from the Swan Canning Estuary adjacent to Perth (Chabanne *et al.* 2012).

Biology and feeding ecology

Indo-Pacific bottlenose dolphins grow to about 2.7 m long and can weigh up to 230 kg (Jefferson *et al.* 2008, Wang and Yang 2009). These dolphins occur mainly in shallow nearshore and inshore coastal waters less than 100 m deep, in some estuaries and bays, with some groups occurring further offshore across continental shelf habitats, while some deeper water offshore movements have also been recorded (Hale *et al.* 2000, Möller *et al.* 2002, Ross 2006, Krützen and Allen 2008, Fury and Harrison 2008, Wang and Yang 2009, Allen *et al.* 2012).

Indo-Pacific bottlenose dolphins are highly social and live in complex and dynamic fission-fusion societies where associations and group sizes vary over short-term, seasonal and longer-term timescales depending upon the numbers of resident dolphins and visitors or transient dolphins present (Connor *et al.* 2000, Möller *et al.* 2002). Smaller coastal bays and estuaries tend to have fewer resident dolphins (Möller *et al.* 2002, Fury and Harrison 2008, Cribb *et al.* 2013), whereas larger communities occur in large bays such as Moreton Bay in Queensland and Shark Bay in WA, and along some open coastal habitats (Preen *et al.* 1997, Hawkins and Gartside 2008, Ansmann *et al.* 2013). Association patterns vary among individuals and between sexes resulting in social and sexual segregation. Pairs or trios of males form long-term alliances for herding females and mating, while females form coalitions within larger social networks, and females with calves prefer shallow protected habitats (Mann *et al.* 2000, Connor *et al.* 2000, Möller *et al.* 2006, Fury *et al.* 2013).

Available global information indicates that Indo-Pacific bottlenose dolphins tend to form relatively small and localised subpopulations that are relatively isolated from each other (Wang and Yang 2009). Separate communities or genetically differentiated subpopulations with a pattern of isolation by distance are evident in some Australian coastal regions (e.g. Krützen *et al.* 2004, Möller *et al.* 2007, Bilgmann *et al.* 2007, Wiszniewski *et al.* 2010, Ansmann *et al.* 2013). Along the NSW coast, genetic analyses have demonstrated considerable genetic differentiation between most of the resident dolphin communities, with at least three genetically distinct subpopulations evident in northern NSW, Port Stephens and in southern NSW (Möller *et al.* 2007; Wiszniewski *et al.* 2010). Females in Port Stephens and Jervis Bay in NSW are relatively philopatric (remain at or return to their place of birth), whereas males exhibit greater levels of dispersal (Möller and Beheregaray 2004).

Indo-Pacific bottlenose dolphins are opportunistic feeders that prey on a wide variety of schooling, demersal, benthic and reef fish species, but there is considerable geographic variability in their diet and in some areas they also consume rays, small sharks, cephalopods and crustaceans (Ross 2006, Krützen and Allen 2008, Jefferson *et al.* 2008, Wang and Yang 2009). Prey are usually less than 30 cm long and include species from many families including Mugilidae, Belonidae, Sciaenidae, Engraulidae, Sepioteuthidae, Sepiidae, Sepiolidae, Loliginidae and Octopodidae (Wang and Yang 2009). Highly specialised foraging strategies are used by some dolphin groups to target specific prey types and these behaviours appear to be socially transmitted (Connor *et al.* 2000, Krützen *et al.* 2005, Wang and Yang 2009). Indo-Pacific bottlenose dolphins are considered to be behaviourally plastic and able to adapt to feeding in association with various fisheries. These dolphins have been observed feeding on discarded bycatch behind prawn trawlers in Moreton Bay, and large males appear to occupy optimal positions for feeding on discards behind these trawlers (Corkeron *et al.* 1990, Chilvers and Corkeron 2001).

Sexual maturity occurs from about nine to 9–15 years, gestation is about 12 months, and the average interbirth interval is about three to six years (Mann *et al.* 2000, Ross 2006, Wang and Yang 2009). Maximum age of males is about 35–40 years, whereas females live for more than 40 years and possibly more than 50 years (Ross 2006, Krützen and Allen 2008). Generation length is estimated to be 21.1 years (Taylor *et al.* 2007). These life history characteristics result in a relatively low reproductive rate and combined with high levels of philopatry and relatively small subpopulation sizes, Indo-Pacific bottlenose dolphins are likely to have a slow capacity for recovery from depletion (Woinarski *et al.* 2014).

Risks and threatening processes

Mortality from fisheries interactions and bycatch is considered to be the most serious anthropogenic threat to Indo-Pacific bottlenose dolphins (Wang and Yang 2009). An estimated 8400 Indo-Pacific bottlenose dolphins were killed from incidental bycatch in the Taiwanese drift gillnet fishery in northern Australian waters during 1974–1986 (Harwood and Hembree 1987, Ross 2006), and these dolphins are killed in shark nets and anti-predator nets around tuna feedlots in Australia and overseas (Kemper and Gibbs 2001, Kemper *et al.* 2005, Ross 2006, Jefferson *et al.* 2008). Indo-Pacific bottlenose dolphins are known to commonly interact with trawl fisheries (Elgin Associates unpublished (a)), and have been recorded feeding behind trawlers in Moreton Bay and taking food through the net mesh (Corkeron *et al.* 1990, Chilvers and Corkeron 2001). Bycatch occurs infrequently in these trawl nets, and the dolphins killed are mostly juveniles (Corkeron *et al.* 1990). One dolphin genetically identified as *T. aduncus* was sampled in association with dolphins interacting with the Pilbara Fish Trawl Fishery off northwestern Australia (Allen and Loneragan 2010).

Other threats to these dolphins (reviewed in Bannister *et al.* 1996, Ross 2006, Woinarski *et al.* 2014) include coastal development, port expansion, aquaculture and habitat loss (Watson-Capps and Mann 2005), bioaccumulation of elevated levels of persistent toxic pollutants (Evans 2003), chronic disturbance from dolphin-watching vessels and increased coastal vessel movements (Bejder *et al.* 2006, Steckenreuter *et al.* 2011), increasing anthropogenic noise and acoustic disturbance (McCauley and Cato 2003), vessel strike and intentional killing (Kemper *et al.* 2005). Potential threats include prey depletion from expanding commercial fisheries and increased recreational take of prey species (Bannister *et al.* 1996, Ross 2006), increased climate variability and altered environmental conditions including increased flood events (Fury and Harrison 2011, Woinarski *et al.* 2014).

Conservation and listing status

The Indo-Pacific bottlenose dolphin is listed as a cetacean species under the EPBC Act, and the 'Arafura/Timor Sea population' of the spotted bottlenose dolphin is listed as migratory under Appendix II of Convention on Migratory Species of Wild Animals. The species is listed as Least Concern in the Northern Territory, but is not listed in other states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014) and similarly in previous Australian status assessments (Bannister *et al.* 1996, Ross 2006). Globally, the Indo-Pacific bottlenose dolphin was assessed as Data Deficient for the IUCN Red List in 2008 (Hammond *et al.* 2008e), and is listed in Appendix II of CITES.

Common bottlenose dolphin *Tursiops truncatus* (Level 2 PSA Residual Risk – High)

Distribution and range

Common bottlenose dolphins have a cosmopolitan distribution extending from tropical to temperate coastal, shelf and offshore waters between about 55°S and 65°N (Rice 1998, Hale *et al.* 2000, Ross 2006, Jefferson *et al.* 2008, Wells and Scott 2009). These dolphins are recorded from the Pacific, Indian and Atlantic Ocean regions and also occur in most enclosed or semi-enclosed seas, with an apparently higher population density in coastal or continental shelf habitats compared with oceanic regions further offshore (Ross 2006, Jefferson *et al.* 2008). Common bottlenose dolphins are broadly distributed around much of the Australian coastal shelf area in deeper waters out to the outer continental shelf, and in some offshore habitats mostly within 1000 km of the continental coast (Ross 2006, Hale 2008) including subtropical Lord Howe Island (Hutton and Harrison 2004). Therefore, their Australian range overlaps extensively with the SPF area.

The full extent of their distribution in Australian waters is not known, and part of this uncertainty arises from difficulties in identifying which species of bottlenose dolphin is present in some regions, particularly where they are sympatric with Indo-Pacific bottlenose dolphins in some inshore environments (Ross 2006, Woinarski *et al.* 2014). Common bottlenose dolphins generally occur in deeper waters further from the coast compared with Indo-Pacific Bottlenose dolphins, but these species co-occur in parts of their range, and the taxonomic status of coastal *Tursiops* bottlenose dolphins in some parts of the southern Australian coast and from Tasmanian waters is uncertain (Hale *et al.* 2000, Kemper 2004, Ross 2006, Möller *et al.* 2008, Charlton-Robb *et al.* 2011, Woinarski *et al.* 2014).

Species identification issues are further complicated by the recent description of a putative new southern Australian *Tursiops* species, the Burrunan dolphin *Tursiops australis*, that is recorded from some coastal waters of Victoria (including Port Phillip Bay and Gippsland Lakes), eastern Tasmania and SA west to St Francis Island (Charlton-Robb *et al.* 2011). The Society for Marine Mammalogy (Committee on Taxonomy 2014) has not included *T. australis* in the global list of recognised marine mammal species and subspecies, and considered the validity of this putative species to be uncertain. Some ongoing genetic and morphological research indicates that Burrunan dolphin specimens fall within the range of Common bottlenose dolphin *T. truncatus* specimens (M. Jedensjö *et al.*, pers. comm. in Woinarski *et al.* 2014). Therefore, the Burrunan dolphin was not evaluated in the recent assessment of the conservation status of marine mammals in Australia by Woinarski *et al.* (2014), and is not separately evaluated in this report. However, of relevance to the assessment of dolphins and matters of national environmental significance for the DCFA, if the Burrunan dolphin is considered a distinct species, Charlton-Robb *et al.* (2011) noted that it 'would qualify for listing as a threatened species' given its small range and area of occupancy, with only two known small resident populations that occur close to disturbed coastal urban and agricultural areas.

Population size and trends

There are no robust estimates of total population size or trends in Australian waters (Hale 2008, Woinarski *et al.* 2014). Hammond *et al.* (2008f) suggested a minimum global abundance estimate of 600,000 common bottlenose dolphins based on a summation of estimates from parts of their range. The global population trend is unknown but some populations are declining, and one subspecies and two subpopulations are assessed as threatened (Hammond *et al.* 2008f). Groups of up to 100 dolphins have been recorded in deeper waters off the coast of NSW and Queensland (Hale 2008), and 151 individuals were photographically identified foraging in association with a trawler off northwestern Australia in 2011 (S. Allen pers. comm. in Woinarski *et al.* 2014). Two small resident communities of bottlenose dolphins identified as Burrunan dolphins *T. australis* occur in Port Phillip Bay (about 80–100 dolphins) and in the Gippsland Lakes, Victoria (Charlton-Robb *et al.* 2011, Howes *et al.* 2012), which may also be relevant to assessment of *T. truncatus*.

Biology and feeding ecology

Common bottlenose dolphins grow to about 1.9–3.8 m long and can weigh up to 650 kg, but most are considerably smaller and there is considerable geographical variation in size (Jefferson *et al.* 2008, Wells and Scott 2009). Although these dolphins have been extensively studied in some other regions overseas and are considered to be one of the best known cetacean species (reviewed in Leatherwood and Reeves 1990, Wells and Scott 2009), relatively little information is available from the Australian region. Around Australia, common bottlenose dolphins mostly occur further offshore and in deeper water greater than 30–100 m habitats across the continental shelf and near the shelf edge compared with inshore Indo-Pacific bottlenose dolphins. Although, as noted above, the ranges of these species overlap in some coastal and inshore areas (Hale *et al.* 2000, Ross 2006, Hale 2008).

These dolphins are highly social and live in dynamic fission-fusion societies where group size and composition change over time (reviewed in Connor *et al.* 2000, Wells and Scott 2009, Möller 2012). Group size tends to be smaller in inshore bays and estuaries while larger groups occur in offshore waters, and group size ranges from about 2–15 dolphins up to aggregations of several hundred to more than 1000 dolphins (Hale 2008, Wells and Scott 2009). Community and subpopulation sizes vary over time corresponding to changes in the numbers of resident dolphins, occasional visitors and transient dolphins present. Strong social bonds exist between mothers and calves and between some related females, some dolphins form nursery groups or mixed sex groups of juveniles, some males remain solitary while others form long-term bonds with other males (Connor *et al.* 2000, Wells and Scott 2009, Möller 2012). Some coastal dolphin communities exhibit long-term residency within their home ranges over many decades, while dolphins in other regions undertake seasonal migrations or larger-scale movements over hundreds or thousands of kilometres (Connor *et al.* 2000, Jefferson *et al.* 2008, Wells and Scott 2009).

Common bottlenose dolphins are generalist feeders that forage in a range of habitats and prey on a wide variety of benthic and pelagic fish (often Sciaenidae, Scombridae and Mugilidae) and squid, and sometimes eat crustaceans (Hale 2008, Jefferson *et al.* 2008, Wells and Scott 2009). However, diets vary among individuals within and between populations and regionally (Wells and Scott 2009). Lactating females with calves tend to feed closer to shore, adolescents feed further away from the coast, and non-breeding females and males feed further offshore (Wells and Scott 2009). Coastal subpopulations forage in shallower areas including rocky reefs and seagrass habitats, whereas offshore subpopulations forage in deeper habitats ranging from about 50–200 m and up to 500 m depths (Hale 2008, Wells and Scott 2009, Gibbs *et al.* 2011, Dunshea *et al.* 2013). Analysis of the diets and feeding ecology of coastal common bottlenose dolphins and inshore *Tursiops* sp. bottlenose dolphins with overlapping ranges in South Australian waters revealed strong evidence of niche partitioning; common bottlenose dolphins feed at a higher trophic level than the inshore bottlenose dolphins (Gibbs *et al.* 2011). The diet of common bottlenose dolphins from the GAB region includes small percentages of Australian sardine, jack mackerel and blue mackerel (Table 4.2 in Section 4.2). Stomach contents of a common bottlenose dolphin that drowned in a fish net in Tasmania included cephalopod beaks and remains of fish including jack mackerel (Gales *et al.* 1992). Some groups of common bottlenose dolphins actively seek or become associated with fishing vessels and regularly forage on discarded catch or remove fish from the nets (Broadhurst 1998, Svane 2005, Allen and Loneragan 2010, Jaiteh *et al.* 2013, Allen *et al.* 2014).

Females become sexually mature at five to 13 years and males at nine to 14 years. Gestation is about 12 months, and the average interbirth interval is about three to six years (Connor *et al.* 2000, Wells and Scott 2009, Möller 2012). Maximum age of males is up to 48 years, while females remain reproductive until about 48 years and can live up to 57 years (Wells and Scott 2009). Generation length is estimated to be 21.1 years (Taylor *et al.* 2007). These life history characteristics result in a relatively low reproductive rate hence common bottlenose dolphin populations are likely to have a slow capacity for recovery from depletion (Woinarski *et al.* 2014).

Risks and threatening processes

Hale (2008) considered that successful conservation of common bottlenose dolphins in Australian waters and elsewhere will depend primarily on the success of minimising incidental bycatch in fishing gear. These dolphins are known to interact with various fisheries throughout their range and incidental bycatch has been recorded in gillnets, trawl nets, purse seine nets, shark nets and from hook and line gear (Paterson 1990, Fertl and Leatherwood 1997, Jefferson *et al.* 2008, Zollett 2009, Reeves *et al.* 2013). In Australian waters, mortality of common bottlenose dolphins has been recorded from bycatch in mid-water trawls from Zone A (pre-2009) of the SPF (Lyle and Willcox 2008, Tuck *et al.* 2013). Three *T. truncatus* dolphins and 14 other dolphins that may have been *T. truncatus* or common dolphins *Delphinus delphis* were recorded as bycatch mortality in 2004 (Lyle and Willcox 2008, Tuck *et al.* 2013). A further eight dolphins that were not identified to species level were recorded as bycatch mortality in Zone A, east Tasmania in 2005 (Lyle and Willcox 2008, Tuck *et al.* 2013).

An estimated 150–350 common bottlenose dolphins were caught in the Pilbara Fish Trawl Interim Managed Fishery (PFTIMF) between 2003–2009 (Allen and Loneragan 2010, Jaiteh *et al.* 2013). Observer-reported bycatch rates were about double the rates reported by trawler skippers (Allen *et al.* 2014). The rate of dolphin bycatch was reduced by about 45 per cent after BRDs were introduced; bycatch rates have not declined further since the introduction of BRDs in 2006 (Allen *et al.* 2014). Allen *et al.* (2014) concluded that modified BRDs to include a top-opening escape hatch might be more effective in reducing dolphin bycatch. Common bottlenose dolphins are known to forage on discarded catch or remove fish from the trawl net codend or sometimes from within trawl nets, which greatly increases the risk of incidental bycatch (Broadhurst 1998, Svane 2005, Allen and Loneragan 2010, Jaiteh *et al.* 2013). Subsurface behaviour of common bottlenose dolphins interacting with trawl nets in the PFTIMF showed very high rates of interaction during most trawls, with dolphins occurring inside nets in 29 of 36 tows recorded (Jaiteh *et al.* 2013). A total of 29 individual dolphins were identified within the nets, with seven of these repeatedly returning to feed within and between tows and during different fishing trips (Jaiteh *et al.* 2013). These results indicate that feeding within trawl nets occurs frequently and may be a specialised form of behaviour used by a subset of dolphins that associate with trawlers (Jaiteh *et al.* 2013).

Broadhurst (1998) noted that common bottlenose dolphins regularly associate with fish and prawn trawlers off the NSW coast and remove bycatch from codends during retrieval of trawl nets, and scavenge discarded catch during sorting. Furthermore, underwater video records showed common bottlenose dolphins manipulating prawn-trawl codends during trawling off northern NSW to remove and eat juvenile whiting *Sillago* spp. and other catch (Broadhurst 1998). The feeding patterns observed indicated that this was a well-established feeding behaviour by these dolphins (Broadhurst 1998).

A range of other threats are known to affect these dolphins (reviewed in Bannister *et al.* 1996, Ross 2006, Woinarski *et al.* 2014) including bioaccumulation of elevated levels of persistent toxic pollutants (Vetter *et al.* 2001, Evans 2003, Wells *et al.* 2005), cetacean morbillivirus infection (Stone *et al.* 2011), habitat degradation caused by coastal development, port expansion, aquaculture and associated increased vessel activity, increasing anthropogenic noise and acoustic disturbance (McCauley and Cato 2003), chronic disturbance from dolphin-watching vessels (Constantine *et al.* 2004, Lusseau *et al.* 2006, Howes *et al.* 2012), and vessel strike (Wells and Scott 2009). Potential threats include prey depletion from expanding commercial fisheries and increased recreational take of prey species (Bannister *et al.* 1996, Ross 2006), and increased climate variability and altered environmental conditions (Woinarski *et al.* 2014).

Conservation and listing status

The common bottlenose dolphin is listed as a cetacean species under the EPBC Act, and is listed as Least Concern in the Northern Territory, but is not listed in other states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014) and similarly in an earlier Australian status assessment by Bannister *et al.* (1996). Ross (2006) recommended that this species be classified as No category assigned but possibly secure. Globally, the Indo-Pacific bottlenose dolphin was assessed as Least Concern for the IUCN Red List in 2008 (Hammond *et al.* 2008f), and is listed in Appendix II of CITES.

Common dolphin *Delphinus delphis* (Level 2 PSA Residual Risk – Medium)

Distribution and range

The short-beaked common dolphin is widely distributed in continental shelf and pelagic waters from tropical to cool temperate regions in the Pacific and North Atlantic Oceans, and is possibly absent from most of the South Atlantic and Indian Oceans (Rice 1998, Jefferson *et al.* 2008, Perrin 2009a, Amaral *et al.* 2012). This species has been recorded from all Australian states and Northern Territory waters, including subtropical Lord Howe Island off NSW and southwestern Australia, with few records from northwestern Australia (Bannister *et al.* 1996, Chatto and Warneke 2000, Bell *et al.* 2002, Hutton and Harrison 2004, Kemper *et al.* 2005, Kemper 2008). There appear to be two main locations in Australian waters with one cluster occurring in the southern southeastern Indian Ocean and another in the Tasman Sea (Woinarski *et al.* 2014). Its Australian range overlaps extensively with the SPF area.

Population size and trends

Short-beaked common dolphins may be the most numerous dolphins in Australian waters and are often reported in coastal waters of southern Australia (Kemper 2008), but there are no robust estimates of the Australian population size or trends (Woinarski *et al.* 2014). The estimated size of the subpopulation of these dolphins in preferred habitat areas of the Gulf St Vincent in SA was about 2000 individuals (Filby *et al.* 2010). Substantial genetic differentiation has been recorded between short-beaked common dolphin subpopulations in SA and those in eastern Australia including Tasmania, with finer levels of subpopulation substructuring along the southeastern and southern Australian coasts possibly associated with spatial variation in oceanographic currents, upwellings or fish distributions (Bilgmann *et al.* 2008, 2014, Möller *et al.* 2011, Amaral *et al.* 2012). At least six different management units of these common dolphins have been identified and this population substructuring is of considerable significance for managing these populations, particularly in relation to managing mortality from fisheries bycatch in the purse seine fishery for sardines off SA and the gillnet fishery for gummy sharks off southern Australia (Bilgmann *et al.* 2008, 2014, Hamer *et al.* 2008).

Globally, this species is considered to be very abundant (Jefferson *et al.* 2008), but there is no robust estimate of global population size and population trends are unknown (Hammond *et al.* 2008g). Overseas, regional population estimates include about 3,000,000 in the eastern tropical Pacific region, and about 370,000 from the western United States coast (Jefferson *et al.* 2008).

Biology and feeding ecology

Short-beaked common dolphins can grow up to about 2.2–2.7 m long and can weigh up to 200 kg, but adult size and colouration varies geographically (Jefferson *et al.* 2008). These dolphins occur in open ocean habitats and over the continental shelf and in some regions they prefer areas of steeply sloping underwater topography and high productivity upwelling areas (Ross 2006, Jefferson *et al.* 2008, Perrin 2009a). They have been sighted from nearshore areas to thousands of kilometres offshore in open pelagic habitats (Jefferson *et al.* 2008). These dolphins are gregarious and form core groups of about 20–30 individuals but can form large aggregations of many thousands of dolphins, with aggregations of up to 100,000 dolphins observed from Australian waters (Bannister *et al.* 1996, Kemper 2008). Schools may be segregated by sex and age, and in the eastern tropical Pacific they can be associated with yellowfin tuna resulting in bycatch in the purse seine fishery (Jefferson *et al.* 2008). In some regions these dolphins appear to undergo seasonal movements and inter-annual migrations in response to changing oceanographic conditions and occurrence of prey (Perrin 2009a).

Short-beaked common dolphins feed primarily on small schooling fishes and squid, including small epipelagic schooling species from families Scombridae and Clupeidae (Perrin 2009a). In South Australian waters these dolphins feed mainly on southern calamari and fish from the Clupeidae and Carangidae families (Kemper 2008). The diet of short-beaked common dolphins from the GAB region includes a high proportion of Australian anchovy and Australian sardine, with smaller amounts of jack mackerel and blue mackerel (Table 4.2 in Section 4.2). In some regions these dolphins feed on mesopelagic species associated with the deep scattering layer that migrates into shallower waters at night (Jefferson *et al.* 2008, Perrin 2009a). These dolphins have been recorded on foraging dives to 200 m, and may dive to at least 280 m depth (Kemper 2008, Perrin 2009a).

Females mature at about six to eight years and males about 7–12 years, gestation is about 10–11.7 months, and the interbirth interval varies regionally from one to three years (Perrin 2009a). Maximum age estimates range from about 22–30 years (Kemper 2008, Perrin 2009a). Generation length is estimated to be 14.8 years (Taylor *et al.* 2007). These life history characteristics indicate that this species has a relatively low lifetime reproductive capacity, therefore populations are susceptible to adverse impacts from relatively low levels of fisheries bycatch mortality (Hamer *et al.* 2008).

Risks and threatening processes

Short-beaked common dolphins are known to commonly interact with trawl fisheries, and bycatch mortality in pelagic trawl, purse seine, gillnets and other fisheries gear has been reported throughout their global range (Fertl and Leatherwood 1997, Jefferson *et al.* 2008, Zollett 2009, Perrin 2009a). In Australian waters, bycatch mortality of short-beaked common dolphins is frequently recorded in fisheries nets, shark nets and in anti-predator nets around tuna feedlots (Paterson 1990, Kemper and Gibbs 2001, Shaughnessy *et al.* 2003, Hamer *et al.* 2008, Bilgmann *et al.* 2014). A total of 14 dolphins that may have been short-beaked common dolphins or *T. truncatus* common bottlenose dolphins were recorded as bycatch mortality in mid-water trawls from Zone A (pre-2009) of the SPF in 2004 (Lyle and Willcox 2008, Tuck *et al.* 2013). In 2005, a further eight dolphins that were not identified to species level were recorded as bycatch mortality in Zone A from east Tasmania (Lyle and Willcox 2008, Tuck *et al.* 2013). An estimated 337 short-beaked common dolphins were killed in the South Australian Sardine Fishery (SASF) between November 2004 and June 2005, with subsequent mitigation measures leading to a substantial reduction in bycatch mortality (Hamer *et al.* 2008). During 2011, a total of 33 short-beaked common dolphins were killed in fisheries netting interactions in Australian waters (Cusick *et al.* 2012). Some of these dolphins have been reported killed for bait or as perceived competition with fishers in Australian waters (Bannister *et al.* 1996, Kemper *et al.* 2005, Ross 2006).

Short-beaked common dolphins have also been recorded interacting with mid-water and other trawls in many regions of the world, with New Zealand trawl interactions include herding fish into nets, taking fish in the net mouth, and as bycatch in mid-water trawls (Thompson *et al.* 2013, Elgin Associates unpublished (a)). Other overseas fisheries interactions include large direct catches in the Black Sea until 1983 that resulted in significant declines, and large numbers killed in bycatch in the eastern tropical Pacific tuna fishery (Jefferson *et al.* 2008, Perrin 2009a). A precipitous decline in abundance of short-beaked common dolphins was recorded in coastal waters of the eastern Ionian Sea from 1996 to 2007 (Bearzi *et al.* 2008). A 12-month assessment of fishing effort and catch, together with circumstantial evidence, suggested that the decline in dolphin abundance was caused largely by prey depletion resulting from overfishing, which was mainly due to purse seining (Bearzi *et al.* 2008). Other threats include high levels of contaminants in tissues in some samples from Australian waters and in many regions overseas (Vetter *et al.* 2001, Lavery *et al.* 2008), detrimental impacts from seismic activities and other acoustic disturbance, and pathogens implicated in mortality events and strandings (Bannister *et al.* 1996, Kemper *et al.* 2005, Woinarski *et al.* 2014). Some vessel strike mortality has been reported for this species in Australian waters (Cusick *et al.* 2012).

Conservation and listing status

The short-beaked common dolphin is listed as a cetacean species under the EPBC Act, Data Deficient in the Northern Territory, but is not listed in other states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014), and was assessed as 'No category assigned but possibly secure' in previous Australian status assessments (Bannister *et al.* 1996, Ross 2006). Globally, the short-beaked common dolphin was assessed as Least Concern for the IUCN Red List in 2008 (Hammond *et al.* 2008g), and is listed in Appendix II of CITES.

Pantropical spotted dolphin *Stenella attenuata* (Level 2 PSA Residual Risk – Medium)

Distribution and range

The pantropical spotted dolphin is widely distributed from tropical to temperate regions in all oceans and some seas, ranging from about latitude 40°N to 40°S, but is much more abundant in warmer tropical waters within this range (Rice 1998, Jefferson *et al.* 2008, Perrin 2009b). This species has been recorded from NSW, Queensland, Northern Territory, and WA waters extending south to Augusta (Ross 2006, Porter 2008). A record from Victorian waters is considered to be erroneous (Ross 2006). Its southern Australian range overlaps partly with the SPF area.

Population size and trends

Pantropical spotted dolphins may be the most abundant or second most abundant cetacean species globally, with about 2.5–3 million individuals (Porter 2008a, Perrin 2009b). They are considered to be potentially abundant within their Australian range, but there are no robust estimates of the Australian population size or trends (Woinarski *et al.* 2014). An estimated 640,000 northeastern offshore pantropical spotted dolphins were present in the eastern tropical Pacific (ETP) region in 1979–2000, representing a decline of about 80 per cent from their original abundance due to unsustainable bycatch in the purse seine tuna fishery in this region since the early 1960s (Gerrodette and Forcada 2005, Perrin 2009b). Despite significant fishery management to reduce bycatch to relatively low levels of a few 100 of these dolphins annually in more recent decades, this population has not exhibited signs of recovery (Gerrodette and Forcada 2005, Perrin 2009b). Other overseas regional population estimates include about 228,000 coastal pantropical spotted dolphins in the ETP, about 438,000 in Japanese waters in the 1990s, and about 15,000 in the eastern Sulu Sea (Jefferson *et al.* 2008, Perrin 2009b).

Biology and feeding ecology

Pantropical spotted dolphins adults grow to 1.6–2.6 m long and can weigh up to 119 kg but exhibit wide geographic variation, with different coastal and offshore forms recognised (Jefferson *et al.* 2008, Perrin 2009b). These dolphins occur mainly in deeper water outer shelf and continental slope habitats and open oceanic habitats, but are recorded closer to shore where deep water occurs nearer the coast, and in some shallower shelf habitats (Ross 2006, Jefferson *et al.* 2008, Porter 2008a). These gregarious dolphins occur in mostly small-to-medium-sized groups of less than 100 dolphins for the coastal form, but offshore groups are usually larger and may contain thousands of dolphins (Ross 2006, Jefferson *et al.* 2008). Larger groups are composed of three types of subgroups each with about 20 individuals: females and their young, juvenile dolphins, and mature males (Porter 2008a). Individual dolphins exhibit daily movements of 20–30 km and have home ranges up to 200–300 nm, with migration of some populations onshore during winter and offshore during summer (Ross 2006, Porter 2008a).

These dolphins feed primarily on small epipelagic and mesopelagic fishes, squids and crustaceans, but also consume nemertean marine worms (Ross 2006, Porter 2008a, Perrin 2009b). Their diet varies regionally and with reproductive state, and lactating females consume a higher proportion of fish, possibly due to their higher nutritional value (Bannister *et al.* 1996, Ross 2006). Diving behaviour off Hawai'i indicates that these dolphins feed mainly at night possibly associated with diurnal vertical migrations of their prey in the deep scattering layer (Perrin 2009b). The diet of pantropical spotted dolphins in the eastern Pacific overlaps strongly with the diet of yellowfin tuna *Thunnus albacares*, leading to strong dolphin and tuna associations, used by fishers to locate and catch tuna (Perrin 2009b).

These dolphins may have a promiscuous breeding system, females mature at about 9–11 years and males at 12–15 years, gestation is about 11.5 months, and the interbirth interval is about two to three years (Perrin 2009b). The maximum age reported is 50 years (Bannister *et al.* 1996), and generation length is estimated to be 23.1 years (Taylor *et al.* 2007).

Risks and threatening processes

Where pantropical spotted dolphins occur in the SPF area there is a risk of incidental capture in fisheries gear (Elgin Associates unpublished (a)). An estimated 560 pantropical spotted dolphins were caught as bycatch in the Taiwanese gillnet fishery off northern Australia between 1974 and 1986 (Harwood and Hembree 1987). Porter (2008a) noted that the Australian population is likely to be subject to significant bycatch in the shark gillnet fishery operating adjacent to, and sometimes illegally within, the northern EEZ. Some pantropical spotted dolphins have been captured in inshore shark nets in NSW and Queensland waters (Ross 2006).

An estimated 3 million offshore pantropical spotted dolphins were killed as incidental bycatch in the purse seine tuna fishery in the ETP from 1959 to 1972, leading to increased fishery management and regulations that significantly reduced bycatch rates (Gerrodette and Forcada 2005, Jefferson *et al.* 2008, Perrin 2009b). However, the population is not showing signs of recovery, possibly as a result of stress from ongoing chase and capture in nets affecting fecundity or survival, or changes to the carrying capacity of the ecosystem that may be preventing recovery (Jefferson *et al.* 2008, Perrin 2009b). Pantropical spotted dolphins have been recorded as bycatch mortality in trawl nets in Malaysia (Elgin Associates unpublished (a)). These dolphins are also recorded as bycatch in purse seine, trawl and gillnet fisheries throughout their range (Jefferson *et al.* 2008). Other overseas fisheries interactions include direct takes in drive and harpoon fisheries in Japan, Philippines, Indonesia and Solomon Islands, and pantropical spotted dolphins have been implicated in depredation and interference with line fisheries in some regions, resulting in deliberate culling of hundreds of dolphins (Reeves *et al.* 2003, Perrin 2009b). Pollution resulting in accumulation of heavy metals in tissues is a threat to these dolphins (Bannister *et al.* 1996, Woinarski *et al.* 2014). In regions where these dolphins are not harpooned or pursued by purse seine fishers, they readily ride bow waves of vessels (Perrin 2009b), which may increase the risk of vessel strike.

Conservation and listing status

The pantropical spotted dolphin is listed as a cetacean species and as a migratory species as '*Stenella attenuata* E Tropical Pacific, SE Asian populations' under the EPBC Act, Data Deficient in the Northern Territory, but is not listed in states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014) and similarly in previous Australian status assessments (Bannister *et al.* 1996, Ross 2006). Globally, the pantropical spotted dolphin was assessed as Least Concern for the IUCN Red List in 2008 (Hammond *et al.* 2008h), and is listed in Appendix II of CITES.

Spinner dolphin *Stenella longirostris* (Level 2 PSA Residual Risk – Medium)

Distribution and range

The spinner dolphin has a pantropical distribution similar to that of the pantropical spotted dolphin encompassing tropical to most subtropical regions in all oceans and some seas, ranging from about latitude 30–40°N to 20–40°S (Rice 1998, Jefferson *et al.* 2008, Perrin 2009c). This species has been recorded from NSW, Queensland, the Northern Territory and Western Australian waters extending south to Bunbury, and from Christmas Island, the Cocos (Keeling) Islands and from Scott Reef off WA (Bannister *et al.* 1996, Ross 2006, Porter 2008b, Woinarski *et al.* 2014). Its southern Australian range overlaps partly with the SPF area.

The taxonomy of spinner dolphins is unsettled. Globally, four subspecies are recognised (Committee on Taxonomy 2014). Perrin *et al.* (1999) considered that two of these subspecies are present in Australian waters: Gray's spinner dolphin *S. l. longirostris* and the dwarf spinner dolphin *S. l. roseiventris*. They considered that the Australian distribution of the smaller inshore dwarf spinner dolphin encompassed northern tropical waters from about Broome in northwestern WA to the Gulf of Carpentaria across to Cape York Peninsula in Queensland and extending through the Timor and Arafura Seas, whereas the larger Gray's spinner dolphin subspecies occurred along the east and west coasts and further offshore in the Pacific and Indian Ocean regions (Perrin *et al.* 1999, Allen *et al.* 2012). In contrast, Porter (2008b) considered that only the dwarf spinner dolphin subspecies was present in Australian waters.

Population size and trends

The global abundance of spinner dolphins is estimated to be about 1.4–1.5 million individuals making this one of the most abundant dolphins in the world (Porter 2008b, Jefferson *et al.* 2008, Perrin 2009c). There is no robust estimate of the population size or trends in Australian waters (Woinarski *et al.* 2014), but surveys in the early 2000s indicated relatively low abundance off northern Australia (Porter 2008b). Estimates of abundance of eastern spinner dolphins *S. l. orientalis* in the ETP region were about 450,000 to 600,000 in 2000 and 2003 (Gerrodette and Forcada 2005, Bearzi *et al.* 2012). This subspecies was heavily impacted by bycatch in the yellowfin tuna purse seine fishery in the ETP that reduced their abundance to less than half of its original size, and although management actions have greatly reduced dolphin bycatch by two orders of magnitude in recent decades, the eastern spinner dolphin population has exhibited very slow rates of increase and limited signs of recovery (Gerrodette and Forcada 2005, Perrin 2009c). An estimated 801,000 whitebelly spinner dolphins were present in the ETP in 2000, with some subspecies population estimates available in other regions, but there are no abundance estimates for the dwarf spinner dolphin subspecies (Perrin 2009c, Bearzi *et al.* 2012).

Biology and feeding ecology

Spinner dolphins exhibit wide geographic variation among the recognised subspecies and coastal and offshore forms, and adults grow to about 1.3–2.3 m long and weigh 23–80 kg with males slightly larger than females in all subspecies (Jefferson *et al.* 2008, Perrin 2009c). These dolphins have an oceanic range but in many tropical regions they use shallow coastal waters including sandy-bottomed bays of oceanic islands and coral atolls by day and move offshore to deeper water at night to feed (Jefferson *et al.* 2008, Perrin 2009c). Spinner dolphins in the ETP are oceanic and prefer tropical surface water habitats where they are often closely associated with pantropical dolphins, yellowfin tuna and seabirds (Porter 2008b, Perrin 2009c). The dwarf spinner dolphin from northern Australia and Southeast Asia occurs almost exclusively in shallow water habitats and feeds over shallow reefs (Porter 2008b, Jefferson *et al.* 2008). The distribution of spinner dolphins along parts of the southern coast of WA may be associated with the warm Leeuwin Current (Bannister *et al.* 1996, Ross 2006).

Spinner dolphins from Hawai'i and some other tropical island groups have a fission-fusion society with dynamic and fluid association patterns of different family groups, whereas in other regions their social structure is characterised by more stable groups and some long-term association patterns (Porter 2008b, Jefferson *et al.* 2008, Perrin 2009c). Group sizes range from a few dolphins up to several thousand, and the maximum recorded movement of individuals was 275 nm over 16 days (Perrin 2009c).

In the western and eastern Pacific regions, oceanic spinner dolphins feed at night mainly on small mesopelagic fishes including myctophids, squids and crustaceans, and can dive 300–600 m or deeper, but most feeding is done at shallower depths (Jefferson *et al.* 2008, Perrin 2009c). In contrast, dwarf spinner dolphins feed over shallow reefs on benthic and reef fishes and some invertebrates (Perrin *et al.* 1999).

Females are sexually mature at about four to seven years and males at 7–10 years, gestation is about 10 months, nursing occurs for one to two years and the interbirth interval is about three years (Perrin 2009c). The maximum age reported is 26 years, and generation length is estimated to be 13.7 years (Taylor *et al.* 2007).

Risks and threatening processes

Spinner dolphins are recorded as bycatch in different fisheries throughout their range, including in purse seines, trawls, gillnets and driftnets (Jefferson *et al.* 2008, Bearzi *et al.* 2012). An estimated 4900 spinner dolphins were caught as bycatch in the Taiwanese gillnet fishery off northern Australia between 1981 and 1985 (Harwood and Hembree 1987). Porter (2008b) noted that the northern Australian population is likely to be subject to some level of bycatch in the shark gillnet fishery operating adjacent to, and sometimes illegally within, the northern Australian EEZ. Some spinner dolphins have been recorded as bycatch in inshore shark nets in Queensland waters (Ross 2006).

The close association of spinner dolphins with yellowfin tuna in the ETP resulted in very large bycatch mortality in the tuna purse seine fishery, and spinner dolphins are considered to be the second-most important dolphin species interacting with this fishery after the pantropical spotted dolphin (Jefferson *et al.* 2008, Bearzi *et al.* 2012). Increased fishery management including per-vessel mortality limits have significantly reduced bycatch rates for spinner dolphins, however, the eastern spinner dolphin population appears to be increasing much more slowly than the expected rate of increase and is not showing clear signs of recovery (Gerrodette and Forcada 2005, Bearzi *et al.* 2012). The slow population increase may be associated with underreporting of bycatch of these dolphins, stress associated with the chase and capture in purse seines affecting fecundity or survival, or changes to the carrying capacity of the ecosystem that may be limiting recovery (Gerrodette and Forcada 2005). Hundreds or thousands of spinner dolphins are estimated to be killed each year in fisheries in the Indian Ocean, and human use of spinner dolphin bycatch has led to increased catches in direct fisheries for these dolphins in the Philippines, Indonesia, Taiwan, Sri Lanka and the Caribbean (Bearzi *et al.* 2012). Chronic disturbance and harassment by dolphin-watching tourist operations is considered to be a threat to spinner dolphin populations in some regions including Brasil, Hawai'i and Indonesia (Perrin 2009c, Bearzi *et al.* 2012). Pollution resulting in accumulation of persistent toxic pollutants in tissues is considered a potential threat to these dolphins (Bannister *et al.* 1996, Woinarski *et al.* 2014).

Conservation and listing status

The spinner dolphin is listed as a cetacean species and as a migratory species as '*Stenella longirostris* E Tropical Pacific, SE Asian populations' under the EPBC Act. This species is listed as Data Deficient in the Northern Territory, in WA the full species is not listed but the subspecies *S. l. longirostris* is Priority 4, and in NSW and Queensland this species is not listed (Woinarski *et al.* 2014). This species and the two subspecies in Australian waters were recently all assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014) and the species was assessed similarly in previous Australian status assessments (Bannister *et al.* 1996, Ross 2006). Globally, the spinner dolphin was assessed as Data Deficient for the IUCN Red List in 2012 (Bearzi *et al.* 2012), and is listed in Appendix II of CITES.

Dusky dolphin *Lagenorhynchus obscurus* (Level 2 PSA Residual Risk – Low)

Distribution and range

Dusky dolphins have a wide but discontinuous distribution in cool temperate waters of the Southern Hemisphere (Rice 1998, Van Waerebeek and Wursig 2009). They occur in apparently disjunct populations off Tasmania and southern Australia, New Zealand, central and southern South America, southwestern Africa and around some oceanic islands (Gill *et al.* 2000, Jefferson *et al.* 2008). Their distribution in Australian waters is not well known, with only 12 records occurring intermittently over the 175–year period until 2000 (Gill *et al.* 2000). Australian records include offshore areas south-east of Tasmania, around eastern Tasmania and Bass Strait, from Victoria, SA and WA, and about 800 km south, south-east of southern WA (Gill *et al.* 2000). Based on the infrequent sightings in Australian waters, Gill *et al.* (2000) considered that these dolphins may not be resident, and that Australian records may be of dolphins temporarily visiting from the east coast of New Zealand where this species is more abundant. Similarly, Constantine (2008) noted that Australian waters might be considered as the extreme limits of the distribution of this species. The Australian distribution range of this species overlaps extensively with the SPF area.

Population size and trends

Dusky dolphins are rarely seen in Australian waters (Constantine 2008). There are no estimates of the Australian or global population size or trends for this species (Hammond *et al.* 2008i, Woinarski *et al.* 2014). Abundance was estimated to be about 7250 individuals off Argentina, and some populations have been depleted by directed catches and fisheries bycatch (Van Waerebeek and Wursig 2009).

Biology and feeding ecology

Dusky dolphins grow to a maximum size of 2.1 m and maximum weight of 100 kg, but most adults are less than 2 m long and weigh up to 70–85 kg (Jefferson *et al.* 2008, Van Waerebeek and Wursig 2009). These relatively small, mainly coastal, dolphins occur predominantly in neritic waters over continental shelf and upper slope habitats, but also occur in deep-water habitats where oceanic water approaches the coast such as along parts of the east coast of New Zealand (Van

Waerebeek and Wursig 2009). Dusky dolphins are highly social and gregarious, forming groups of up to 50–500 dolphins with some larger groups containing more than 1000 individuals (Jefferson *et al.* 2008). Groups of 3–70 individuals have been sighted in Australian waters (Gill *et al.* 2000). Dusky dolphins can move considerable distances, up to 780 km, and in some areas exhibit diurnal and seasonal inshore-offshore movements (Van Waerebeek and Wursig 2009). Most records of dusky dolphins from the Australian region occurred over warmer seasons from October through to April, and at least some may be associated with changes in oceanographic features such as the position of the Subtropical Convergence (Gill *et al.* 2000).

Dusky dolphins feed mainly on small schooling fishes including anchovies, lantern fishes and pilchards, but also feed on a wide variety of other fish species and squid (Van Waerebeek and Wursig 2009). During the day they can exhibit cooperative foraging on small schooling fish, but are adaptable and are also recorded feeding individually and nocturnally on lantern fish and squid off the east coast of New Zealand (Constantine 2008, Van Waerebeek and Wursig 2009). A group of dusky dolphins observed off eastern Tasmania was recorded associated with hundreds of short-tailed shearwaters *Puffinus tenuirostris* and large schools of fish that were probably jack mackerel (Gill *et al.* 2000).

Dusky dolphins may have a promiscuous mating system involving sperm competition, sexual maturity is reached at about four to six years, gestation lasts for 12.9 months, and the interbirth interval is about 2.4 years (Taylor *et al.* 2007, Van Waerebeek and Wursig 2009). Longevity is about 35 years, and generation length is estimated to be 16.4 years (Taylor *et al.* 2007).

Risks and threatening processes

Dusky dolphins have been recorded as bycatch in trawl nets internationally (Fertl and Leatherwood 1997). High rates of bycatch mortality of dusky dolphins occurred in mid-water trawls off the Patagonian coast from 1982–1994, with about 400–600 dolphins taken each year in the mid-1980s then declining by the mid-1990s, resulting in annual mortality of up to 8 per cent of the regional population (Hammond *et al.* 2008i). Fisheries-related mortality in Peruvian coastal waters in the early 1990s was considered unsustainable with up to 7000 dusky dolphins taken annually from harpooning and from directed and incidental catch in drift nets (Van Waerebeek and Wursig 2009). About 200 dusky dolphins were killed in gillnets off Kaikoura, New Zealand in 1984, with a lower but unknown level of bycatch in more recent decades (Van Waerebeek and Wursig 2009). Potential threats include incidental bycatch in discarded netting, fisheries impacts on prey species, pollution, impacts of dolphin-based ecotourism, and climate and oceanographic change which is predicted to have an unfavourable effect on the species' range (Bannister *et al.* 1996, MacLeod 2009, Van Waerebeek and Wursig 2009, Woinarski *et al.* 2014). Dusky dolphins often approach vessels and ride bow waves, which increases the risk of vessel strike (Jefferson *et al.* 2008).

Conservation and listing status

The Dusky dolphin is listed as a cetacean species and a migratory species under the EPBC Act, but is not listed in states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014) and similarly in previous Australian status assessments (Bannister *et al.* 1996, Ross 2006). Globally, the dusky dolphin was assessed as Data Deficient for the IUCN Red List in 2008 (Hammond *et al.* 2008i), and is listed in Appendix II of CITES.

Short-finned pilot whale *Globicephala macrorhynchus* (Level 2 PSA Residual Risk – Medium)

Distribution and range

Short-finned pilot whales have an extensive circumglobal distribution in tropical, subtropical and some warm temperate regions of all oceans and the Red Sea, with most records within the range from latitude 50°N to 40°S (Rice 1998, Jefferson *et al.* 2008, Olson 2009). Around Australia, this species has been recorded from strandings in all states and the Northern Territory and additional sightings from other Australian waters (Kemper *et al.* 2005, Ross 2006, Hindell and Gales 2008). Most records in Australian waters are located north of 30°S in oceanic and some coastal areas, and the southern distribution records may be related to southward flowing warm water currents (Bannister *et al.* 1996, Ross 2006). The southern range of this species overlaps with the northern range of the long-finned pilot whale *G. melas* (Hindell and Gales 2008). The southern Australian distribution range of short-finned pilot whales overlaps extensively with the SPF area.

Population size and trends

Short-finned pilot whales are considered to be relatively common within their Australian range and globally, but there are no robust estimates of the Australian or global population size or population trends (Taylor *et al.* 2008b, Woinarski *et al.* 2014). Abundance estimates are available for some populations in Northern Hemisphere regions including: around 589,000 in the eastern tropical Pacific, about 60,000 off Japan, and about 7700 in the Sulu Sea, Philippines (Jefferson *et al.* 2008, Taylor *et al.* 2008b).

Biology and feeding ecology

Short-finned pilot whales are highly sexually dimorphic with adult females growing up to 5.1 m whereas males grow up to 7.2 m in length and up to 3600 kg, with morphologically and genetically distinct geographic forms occurring in Japanese waters (Jefferson *et al.* 2008, Oremus *et al.* 2009). These pilot whales occur in both coastal and offshore oceanic habitats, with higher densities over outer shelf and continental slope habitats or associated with high relief underwater topography (Olson 2009). Short-finned pilot whales are largely nomadic, with some seasonal movements over the continental shelf in response to movements of their squid prey (Hindell and Gales 2008).

They are highly social odontocetes and typically form stable groups of 20–40 individuals that reflect close matrilineal associations, with larger groups containing hundreds of pilot whales reported in some regions (Jefferson *et al.* 2008, Hindell and Gales 2008, Olson 2009). Matrilineal groups contain individuals of both sexes and all age classes and these pilot whales typically remain within their natal group throughout their lifetime (Olson 2009). Although males stay with female kin, they are thought to breed with females from other family groups during temporary larger aggregations, which is an unusual social structure among mammals (Olson 2009). Their strong social bonds are thought to be a factor in the propensity for these pilot whales to mass strand (Olson 2009).

Short-finned pilot whales mostly eat squid, with cuttlefish, octopus and a range of fish species also consumed (Ross 2006, Jefferson *et al.* 2008, Hindell and Gales 2008). Dives to deeper than 600 m have been recorded, with dive patterns varying diurnally in response to movements of vertically migrating prey in the deep scattering layer (Olson 2009).

Short-finned pilot whales have a polygynous mating system, and genetic analyses show strong differentiation between ocean basins and within the Pacific Ocean, indicating that there is limited dispersal of female lineages between regional subpopulations (Oremus *et al.* 2009). Life history characteristics include long life span, delayed maturation, and long interbirth interval resulting in only four to five calves being produced by a female during her lifetime (Hindell and Gales 2008, Olson 2009). Females are sexually mature at eight to nine years and males at 13–17 years (Jefferson *et al.* 2008), there is an extended gestation period of 14.9 months, and females suckle young for an extended period leading to a remarkably long interbirth interval of 6.9 years (Ross 2006, Taylor *et al.* 2007, Hindell and Gales 2008). Females become post-reproductive at about 40 years, and female longevity is estimated to be at least 63 years (Jefferson *et al.* 2008). Generation length is estimated to be 23.5 years (Taylor *et al.* 2007). These life history characteristics and the slow rate of reproduction result in a slow capacity for recovery from depletion.

Risks and threatening processes

Short-finned pilot whales have been recorded as bycatch in a range of mid-water and bottom trawl fisheries operations in the mid-Atlantic, US east coast, off Mauritania and off north-east Africa in pelagic freezer/factory trawlers (Fertl and Leatherwood 1997, Zollett 2009, Elgin Associates unpublished (a)). Pilot whales are considered to be particularly susceptible to entanglement in driftnets and are taken as bycatch in driftnet fisheries in the North Pacific Ocean, and previously in the squid purse seine fishery off California (Jefferson *et al.* 2008, Taylor *et al.* 2008b). They are also killed in drive fisheries in Japan, and in harpoon fisheries in parts of the Philippines, Indonesia and in the Caribbean (Jefferson *et al.* 2008, Taylor *et al.* 2008b). A few short-finned pilot whales are taken in the longline fishery off Hawai'i, and some individuals have been killed by gunshot wounds and spear wounds in the Caribbean (Taylor *et al.* 2008b, Hamer *et al.* 2012). These pilot whales are considered likely to be susceptible to loud sounds such as those from navy sonar and seismic exploration, and some mass stranding events of short-finned pilot whales have been associated with high levels of anthropogenic sound and large-scale military exercises (Taylor *et al.* 2008b, Zirbel *et al.* 2011). Other potential threats include pollution resulting in bioaccumulation of toxic pollutants in tissues, and prey depletion from expanding commercial fisheries (Bannister *et al.* 1996, Woinarski *et al.* 2014).

Conservation and listing status

The short-finned pilot whale is listed as a cetacean species under the EPBC Act, Rare in SA, Data Deficient in the Northern Territory, but is not listed in other states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014) and as 'No category assigned but possibly secure' in previous Australian status assessments (Bannister *et al.* 1996, Ross 2006). Globally, the short-finned pilot whale was assessed as Data Deficient for the IUCN Red List in 2008 (Taylor *et al.* 2008b), and is listed in Appendix II of CITES.

Long-finned pilot whale *Globicephala melas* (Level 2 PSA Residual Risk – Medium)

Distribution and range

Long-finned pilot whales have a disjunct distribution in temperate to subpolar waters of the Northern and Southern Hemispheres (Rice 1998, Jefferson *et al.* 2008). The Southern Hemisphere subspecies *G. melas edwardii* is isolated from the Northern Hemisphere subspecies, and most records in the Southern Hemisphere occur within the range from 20°S to 65°S, but extend south to 68°S (Rice 1998, Jefferson *et al.* 2008, Olson 2009). Their distribution in Australian waters is poorly known. They have been recorded from sightings and strandings in many locations around and south of Australia, from all states and the Northern Territory, with additional records from subtropical Lord Howe Island and subantarctic Macquarie Island (Bannister *et al.* 1996, Hutton and Harrison 2004, Ross 2006, Gales and Hindell 2008). The northern range of this species overlaps with the southern range of the short-finned pilot whale (Gales and Hindell 2008). The Australian distribution range of long-finned pilot whales overlaps extensively with the SPF area.

Population size and trends

Long-finned pilot whales are considered to be relatively abundant within their Australian range and globally, but there are no robust estimates of the Australian or global population size or population trends (Taylor *et al.* 2008c, Woinarski *et al.* 2014). Abundance estimates are available for some populations, including about 200,000 in summer south of the Antarctic Convergence, and about 780,000 in the central and northeastern North Atlantic Ocean (Jefferson *et al.* 2008, Taylor *et al.* 2008c).

Biology and feeding ecology

Long-finned pilot whales are sexually dimorphic with adult females growing up to 5.7 m and weighing up to 1300 kg, while males grow up to 6.7 m and weigh up to 2300 kg (Jefferson *et al.* 2008). These pilot whales occur in deep oceanic waters and in areas of high productivity along the continental slope, and move into shallower continental shelf waters in pursuit of prey (Ross 2006). Long-finned pilot whales are considered to be migratory, with seasonal movements apparently occurring in response to movements of their main squid prey (Ross 2006).

Long-finned pilot whales are highly social and form stable groups containing 10–50 individuals, but can aggregate to form larger groups containing thousands of pilot whales (Jefferson *et al.* 2008, Gales and Hindell 2008). Matrilineal groups contain whales of both sexes and all age classes and individuals typically remain within their natal group throughout their lifetime (Olson 2009). Long-finned pilot whales are one of the most commonly recorded species involved in mass stranding events and their strong social bonds may influence this behaviour (Olson 2009). However, multiple maternal lineages were found among stranded long-finned pilot whales, which challenges the assumption that strong kinship cohesion leads to mass stranding of these whales (Oremus *et al.* 2013). Their strong social bonds also make these pilot whales susceptible to herding by whalers in drive fisheries (Jefferson *et al.* 2008, Olson 2009). More than 100 stranding events have been recorded around Australia, with about half from Tasmania (Bannister *et al.* 1996, Ross 2006). The majority of mass strandings from Australia have occurred in warmer months from December to March (Ross 2006), and recent satellite tracking of pilot whales that stranded in Tasmania and were successfully released has shown that these individuals survived, at least in the short term (Gales *et al.* 2012).

Long-finned pilot whales mostly eat squid, but will also take small to medium-sized fish such as mackerel, herring and cod when these are available (Ross 2006, Jefferson *et al.* 2008, Gales and Hindell 2008). Stomach contents of long-finned pilot whales that stranded in Tasmania included mainly cephalopod beaks and remains of fish (Gales *et al.* 1992). These pilot whales can dive deeper than 1000 m but tend to forage during shallower dives at night on their vertically migrating prey (Gales and Hindell 2008).

Long-finned pilot whales have a polygynous mating system (Jefferson *et al.* 2008). Life history characteristics include long life span, delayed maturation, and long interbirth intervals of three to six years (Gales and Hindell 2008, Olson 2009). Age at sexual maturity for females varies from 5–15 years and averages 17 years for males (Ross 2006), gestation lasts for about 12 months and lactation occurs over an extended period for up to three years (Gales and Hindell 2008). Longevity varies from 35–45 years for males and more than 60 years for females (Jefferson *et al.* 2008). Generation length is estimated to be 24 years (Taylor *et al.* 2007). These life history characteristics and slow rate of reproduction result in a slow capacity for recovery from depletion.

Risks and threatening processes

Long-finned pilot whales have been recorded as bycatch in offshore, mid-water and bottom trawls, and some individuals have been observed feeding in association with trawl nets (Fertl and Leatherwood 1997, Zollett 2009, Elgin Associates unpublished (a)). Pilot whales are susceptible to entanglement in driftnets and have been recorded as bycatch in driftnet, gillnet and purse seine fisheries (Jefferson *et al.* 2008, Taylor *et al.* 2008c). They have also been taken in large-scale drive fisheries in the North Atlantic Ocean, including in the Faroe Islands and Greenland (Jefferson *et al.* 2008, Taylor *et al.* 2008c). These pilot whales are considered likely to be susceptible to acoustic trauma from loud anthropogenic sounds, with possible links between naval activities and strandings (Taylor *et al.* 2008c, Zirbel *et al.* 2011). Other threats include pollution resulting in bioaccumulation of toxic pollutants in tissues, and potential for prey depletion from expanding commercial fisheries (Bannister *et al.* 1996, Woinarski *et al.* 2014).

Conservation and listing status

The long-finned pilot whale is listed as a cetacean species under the EPBC Act, but is not listed in any states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Least Concern in Australian waters (Woinarski *et al.* 2014) and as 'No category assigned but possibly secure' in previous Australian status assessments (Bannister *et al.* 1996, Ross 2006). Globally, the long-finned pilot whale was assessed as Data Deficient for the IUCN Red List in 2008 (Taylor *et al.* 2008c), and is listed in Appendix II of CITES.

Killer whale *Orcinus orca* (Level 2 PSA Residual Risk – Medium)

Distribution and range

Killer whales are the most widely distributed marine mammal, with an extensive circumglobal distribution throughout all oceans and in most seas (Rice 1998, Ford 2009). Their latitudinal range encompasses equatorial to high latitude polar regions to the ice-edge and within pack ice (Baird 2000, Ford 2009). They are more commonly recorded in temperate regions of high productivity where prey are abundant, while less information is available from tropical and offshore oceanic regions where fewer sightings occur (Forney and Wade 2006, Ford 2009).

The taxonomy of killer whales is uncertain and needs revision. At present, one cosmopolitan species of killer whale with two unnamed subspecies are recognised (Committee on Taxonomy 2014), however it has long been known that morphologically different forms occur in some regions that may represent different species or subspecies (Rice 1998). In recent decades, a number of distinct ecotypes (A, B, C and D) have been identified that differ in their morphology and phenotypic characteristics, prey preferences and behaviour, and molecular phylogenetic analyses have indicated that at least some of these ecotypes should be considered to be separate species and others may be subspecies (e.g. Pitman and Ensor 2003, Pitman *et al.* 2007, Jefferson *et al.* 2008, Pitman *et al.* 2011, reviewed in Woinarski *et al.* 2014). These different ecotypes have different geographic ranges: Type A killer whales have the broadest distribution and occur in all oceans and seas from the equator to the edge of polar seas in the Northern and Southern Hemispheres; Type B whales are mainly recorded in the Antarctic and Southern Ocean with some groups exhibiting large-scale periodic movements to lower latitudes; Type C whales occur mainly in pack ice habitats in east Antarctica; Type D whales are primarily pelagic with a circumpolar subantarctic distribution range (Baird 2000, Pitman and Ensor 2003, Pitman *et al.* 2007, Ainley *et al.* 2009, Pitman *et al.* 2011, Durban and Pitman 2012, reviewed in Woinarski *et al.* 2014).

In the Australian region, killer whales have been recorded from all state and Northern Territory waters, around Christmas Island, subantarctic Macquarie and Heard Islands, and south of Australia in the Southern Ocean to high latitude polar waters close to the Antarctic coast (e.g. Bannister *et al.* 1996, Kemper *et al.* 2005, Ross 2006, Morrice and Gill 2008, Van Waerebeek *et al.* 2010, reviewed in Woinarski *et al.* 2014). They are commonly sighted in southeastern Australian coastal waters and along the edge of the continental shelf from southeastern Tasmania, Victoria and southern NSW, around Macquarie Island, and in some Australian Antarctic Territory waters (Bannister *et al.* 1996, Ross 2006, Morrice and Gill 2008, Van Waerebeek *et al.* 2010). Killer whale Types A, B and C occur in Australian Antarctic Territory waters, with some types occurring around Macquarie Island and in Australian coastal waters (Pitman and Ensor 2003, Morrice 2007, Morin *et al.* 2010, R. Pitman and D. Donnelly pers. comm. in Woinarski *et al.* 2014). The Australian distribution range of killer whales overlaps completely with the SPF area.

Population size and trends

Killer whales are commonly sighted in some coastal waters in southeastern Australia and around Macquarie Island but there is no reliable estimate of the Australian population size or trends (Bannister *et al.* 1996, Morrice 2007, Woinarski *et al.* 2014). Forney and Wade (2006) provided a minimum global abundance estimate of about 50,000 killer whales, but considered that this was likely to be an underestimate because abundance estimates are lacking for large areas of the South Pacific, Indian and South Atlantic Oceans, and some high latitude areas in the Northern Hemisphere. Killer whales are thought to be relatively abundant in the Southern Ocean where about 1600 were taken by Soviet whalers. The estimate of about 25,000 killer whales in the region south of 60°S is considered to be uncertain (Forney and Wade 2006, Taylor *et al.* 2008d). The global population trend for killer whales is unknown (Taylor *et al.* 2008d).

Biology and feeding ecology

Killer whales are the largest delphinids and are sexually dimorphic with adult females growing up to 7.7–8.5 m and weighing up to 7500 kg, while males grow up to 9.0–9.8 m and weigh nearly 10,000 kg (Jefferson *et al.* 2008, Ford 2009). Extensive geographic variation occurs among different ecotypes. Killer whales use a wide range of coastal to open ocean marine habitats and are occasionally reported in estuaries and rivers (Forney and Wade 2006, Ford 2009). Their density increases with latitude and in areas of high productivity, and large aggregations of tens to hundreds of Type B and C Killer whales are recorded close to ice-edge habitats in the Southern Ocean (Pitman and Ensor 2003, Forney and Wade 2006, Ainley *et al.* 2009). In Australian waters they occur in coastal areas, along the continental shelf, in deeper slope and oceanic regions, and in subantarctic and Antarctic areas (Bannister *et al.* 1996, Ross 2006, Morrice and Gill 2008, Van Waerebeek *et al.* 2010).

Movement patterns vary among ecotypes, with resident populations exhibiting seasonal movement and offshore forms showing larger-scale movement in response to prey (Baird 2000, Jefferson *et al.* 2008). Type A and some Type B ecotypes periodically migrate from Antarctic to lower latitude waters (Pitman and Ensor 2003, Ford 2009). Seasonal trends in sighting records suggest that some killer whales in Australian waters may undertake seasonal migrations in response to prey aggregations (Morrice 2007). Photo-identified individuals have been recorded moving from Jervis Bay in NSW to the Derwent River near Hobart, Tasmania, and one killer whale identified off Victoria was resighted off southern NSW (D. Donnelly pers. comm. in Woinarski *et al.* 2014).

Killer whales are highly social delphinids that form complex multi-level social groups. Some resident killer whale populations have matrilineal groups containing up to four generations of related whales that remain in their natal group throughout their life, and these groups can aggregate to form larger pods with up to three matrilines and 49 individuals (Baird 2000, Ford 2009). Pods form clans with similar vocal dialects to maintain social interactions and group cohesion, and some pods regularly associate with others to form higher-level communities (Baird 2000, Ford 2009). Some killer whale groups in Australian waters have high fidelity with long-term associations lasting at least 15 years (D. Donnelly pers. comm. in Woinarski *et al.* 2014). Killer whale groups containing up to 52 whales have been recorded south of Australia (Bannister *et al.* 1996), and aggregations containing more than 100 individuals have been observed off southern WA (D. Donnelly pers. comm. in Woinarski *et al.* 2014).

Killer whales are apex marine predators and are known to prey on more than 140 different species including at least 50 marine mammal species, many species of bony fish, penguins, turtles, sharks and other elasmobranchs, and cephalopod, with different ecotypes specialising in different types of prey such as marine mammals or fish (Baird 2000, Ford 2009). In Australian waters, these whales have been recorded attacking or preying on various fish species including fish caught on longlines, sharks, and a wide range of marine mammals including dolphins and whales, dugongs, fur seals and Australian sea lions (Bannister *et al.* 1996, Ross 2006, Morrice and Gill 2008, D. Donnelly pers. comm. in Woinarski *et al.* 2014). Foraging varies among different ecotypes but usually involves cooperative hunting and highly coordinated group behaviour such as herding of fish and attacks on marine mammals, with dive depths varying from shallow 20–30 m dives to more than 200 m (Baird 2000, Ford 2009).

Life history characteristics have been well studied in resident killer whales off British Columbia and Washington and include long life span, delayed maturation, and long interbirth interval resulting in a slow rate of reproduction (Ford 2009). Age at sexual maturity is 10–12 years for females and about 15 years for males, gestation extends over 15–18 months, weaning occurs at one to two years or older and the average interbirth interval is estimated to be about five years but ranges from 2–14 years (Baird 2000, Taylor *et al.* 2007, Ford 2009). Maximum longevity is about 50–60 years for males and 80–90 years for females, with females producing an average of five calves over their 25-year reproductive period that finishes at about 40 years of age (Jefferson *et al.* 2008, Ford 2009). Generation length is estimated to be 25.7 years (Taylor *et al.* 2007).

Risks and threatening processes

Killer whales commonly interact with trawl fisheries internationally and are frequently reported as scavenging around trawlers (Fertl and Leatherwood 1997, Elgin Associates unpublished (a)). Some bycatch has been recorded internationally in trawl net and driftnet fisheries but is considered to be rare, and no incidental bycatch mortality has been reported from Australian waters (Fertl and Leatherwood 1997, Taylor *et al.* 2008d, Elgin Associates unpublished (a)). Small numbers are taken in coastal fisheries in Japan, Indonesia, the Caribbean region and Iceland (Taylor *et al.* 2008d). Killer whales are considered to be one of the main species involved in depredation of fish catch from longline fisheries at higher latitudes including eastern and southern Australia, leading to reports of fishers illegally killing these whales off Tasmania and elsewhere (Bannister *et al.* 1996, Shaughnessy *et al.* 2003, Reeves *et al.* 2003, 2013, Taylor *et al.* 2008d, Hamer *et al.* 2012). Prey depletion is considered to be a threat to some populations of killer whales that specialise in feeding on fish species targeted by commercial fisheries such as Antarctic toothfish *Dissostichus mawsoni*, southern bluefin tuna *Thunnuss maccoyii* and salmon (Bannister *et al.* 1996, Taylor *et al.* 2008d, Ainley *et al.* 2009). As apex predators, killer whales are potentially at risk from bioaccumulation of toxic pollutants in tissues, and high levels of polychlorinated biphenyls (PCBs). Other pollutants have been recorded in killer whales from some regions (Ross *et al.* 2000, Rayn *et al.* 2004, Taylor *et al.* 2008d). Increased whale-watching activities and anthropogenic noise can result in disturbance and degradation of important habitats for killer whales (Williams *et al.* 2006), and scars and damage from vessel strikes are evident on some killer whales in Australian waters (D. Donnelly pers. comm. in Woinarski *et al.* 2014).

Conservation and listing status

The killer whale is listed as a cetacean species and as a migratory species under the EPBC Act, Data Deficient in the Northern Territory, but is not listed in other states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014). In previous Australian status assessments this species was assessed as 'No category assigned but probably secure' by Bannister *et al.* (1996), and 'No category assigned but possibly secure' by Ross (2006). Globally, the killer whale was assessed as Data Deficient for the IUCN Red List in 2008 (Taylor *et al.* 2008d), and is listed in Appendix II of CITES.

Sperm whale *Physeter macrocephalus* (Level 2 PSA Residual Risk – Medium)

Distribution and range

Sperm whales are one of the most widely distributed marine mammal species, with a cosmopolitan distribution in most deeper-water marine habitats from equatorial to polar regions in both northern and southern hemispheres, and occurring in the Mediterranean Sea and some other seas (Rice 1998, Whitehead 2009). Females and young males mostly occur in lower latitudes extending to about 40–50°, whereas males range more widely and move to higher latitude habitats including polar waters as they mature, with periodic return movements to lower latitude warmer waters to breed (Whitehead 2003).

In the Australian region, sperm whales have an extensive distribution in Commonwealth waters and have been recorded from all state and Northern Territory waters, from Australian Antarctic Territory waters and other oceanic offshore areas around Australia (e.g. Townsend 1935, Bannister *et al.* 1996, Smith *et al.* 2012a, reviewed in Woinarski *et al.* 2014). Sperm whales are relatively concentrated in a narrow area near the steep continental shelf edge from Esperance to Cape Leeuwin off southern WA, and are more widely dispersed offshore from Perth to Carnarvon off the west coast of WA (Bannister 2008). Sperm whales occur off the north-west and west coasts of Tasmania, and seasonally off NSW including near Wollongong and Sydney, Lord Howe Island in the Tasman Sea, and off Stradbroke Island in Queensland (Bannister *et al.* 1996; Evans *et al.* 2002; Hutton and Harrison 2004). Historical sightings and catch records from American whalers in the 19th and early 20th centuries include many records of sperm whales off WA and parts of southern Australia, and along much of the east coast (Townsend 1935, Smith *et al.* 2012a). The Australian distribution range of sperm whales overlaps completely with the SPF area.

Population size and trends

There is no reliable estimate of the total sperm whale population size or overall trends in Australian waters (Bannister *et al.* 1996, Bannister 2008, Woinarski *et al.* 2014). 'Open boat' whaling records during the 1800s showed most catches occurred off WA and in the Tasman Sea (Townsend 1935, Bannister 2008). 'Modern' sperm whaling in the 1900s occurred primarily off Albany in southern WA mainly from 1955 to 1978 where annual catches exceeded 400 whales, with larger catches south of Australia prior to 1975 considered likely to have affected the demography of sperm whales in Australian waters (Bannister 1968, 2008). Whaling significantly reduced the abundance of large breeding males and caused a significant decline in pregnancy rate that contributed to the closure of the whaling station in 1978 (Kirkwood *et al.* 1980, Bannister 2008). Aerial surveys, catch records and modelled estimates indicated that abundance of sperm whales in this region substantially declined from 1947 to 1979, with females aged 13 years and older reduced to 91 per cent of their 1947 abundance, whereas males aged 20 years and older were heavily depleted to only 26 per cent of their 1947 abundance (Kirkwood *et al.* 1980). Aerial surveys off Albany in 2009 showed no evidence of recovery and an apparent further decline in the numbers of sperm whales in this region compared with the earlier aerial surveys during whaling from 1968 to 1978 (Carroll *et al.* 2013). It is not clear whether this change reflects a decline in the sperm whale population or movement of whales to other areas (G. Carroll pers. comm. in Woinarski *et al.* 2014), and the recent survey off Albany does not provide inference on population trends across other regions around Australia.

The estimated global pre-whaling population size was about 1,110,000 sperm whales (Whitehead 2002), but two overlapping phases of commercial whaling from 1712 through to 1988 caused substantial depletion of these whales and ongoing effects on population recovery (Whitehead 2009). The global population was estimated to have been reduced to about 71 per cent of its pre-whaling size by 1880 (Whitehead 2002), and an estimated 405,898 Sperm whales were killed in the Southern Hemisphere in the 1900s during the 'modern' whaling period (Clapham and Baker 2009). By 1999, the global population was estimated to be about 360,000, representing depletion to about 32 per cent of the pre-whaling abundance (Whitehead 2002). Although whaling caused a significant population reduction, sperm whales are among the most abundant large whale species (Jefferson *et al.* 2008), but there are insufficient data to accurately determine abundance or population structure in ocean basins (NMFS 2010a). There is no direct evidence of an increase in any part of the global population since whaling ceased, nor evidence in most regions that they have not increased, but there is ongoing concern that some regional populations of sperm whales are declining (Taylor *et al.* 2008e).

Biology and feeding ecology

Sperm whales are the largest odontocetes and the most sexually dimorphic cetaceans with adult females growing up to 11–12 m and weighing up to 13.5 tonnes (t), while mature males grow to about 16–18 m and weigh up to 57 t (Jefferson *et al.* 2008, Whitehead 2009). They occur primarily in deep water offshore pelagic or continental slope habitats and are generally more abundant in areas of higher primary productivity, including upwelling areas (Whitehead 2009). They may approach closer to coasts in deep water habitats near oceanic islands or where the continental shelf is narrow, such as off Albany (Bannister 2008). Sperm whales exhibit variable movement and migration patterns with mid-latitude groups tending to migrate pole-ward in summer then to lower latitudes in winter, whereas in some equatorial and temperate regions no clear seasonal migration patterns are evident (Whitehead 2003). They have been recorded off eastern Antarctica in Australian Antarctic Territory waters in summer in deep water habitats averaging about 4000 m depth (Gedamke and Robinson 2010). Sperm whales are known to travel westwards along the coast off Albany and have been reported to move across southern Australian waters between the western South Pacific and south-east Indian Ocean regions (Bannister 2008).

Female sperm whales form stable social 'nursery' groups of about 10–25 females and calves in oceanic habitats in water deeper than 1000 m where sea surface temperatures are warmer than about 15–18°C (Whitehead 2003, Bannister 2008). Female groups form multilevel societies in the Pacific Ocean, with temporary larger aggregations of female units from the same cultural clan, and these vocal clans contain thousands of females with distinct vocalisations that may be culturally transmitted (Whitehead 2003, Whitehead *et al.* 2012). Home ranges are usually smaller for females compared to males, although female groups sometimes undertake intra-ocean dispersal movements, while males tend to roam widely with more frequent inter-oceanic movements (Jefferson *et al.* 2008, Whitehead 2009). Young males remain with females in tropical and subtropical regions until they are between 4–21 years old then depart from their natal group to form loosely aggregated 'bachelor' herds (Whitehead 2003). Males subsequently move to higher latitude colder regions as they age and mature to become mostly solitary, then periodically return to warmer breeding grounds where they search for female nursery groups for mating (Whitehead 2003, Jefferson *et al.* 2008). Population structure is uncertain, with some evidence for genetic differentiation within and between some ocean basins but low or negligible nuclear DNA differentiation evident between populations in different ocean basins (Whitehead 2009). Recent molecular research in Australian waters indicates that sperm whales have a matrilineal population structure, with females more likely to exhibit natal philopatry whereas males are more likely to disperse (L. Moller pers. comm. in Woinarski *et al.* 2014).

Sperm whales strand frequently compared with many cetacean species, and strandings are relatively common in Tasmania (Bannister 2008). An 11–13 year periodicity in sperm whale and other cetacean stranding events in Tasmania and Victoria is correlated with climatic and oceanographic changes that may influence the northward movement of prey species and result in increased cetacean abundance and stranding events in these regions (Evans *et al.* 2005).

Sperm whales are key predators of oceanic cephalopods and also consume an extraordinary range of other species including some other invertebrates, large sharks, skates and demersal fishes in deeper ocean habitats (Jefferson *et al.* 2008, Whitehead 2009). They typically dive to about 400–600 m to forage, but are thought to be capable of reaching extraordinary depths of 3200 m or deeper and can dive for more than one hour (Bannister *et al.* 1996, Whitehead 2003, Jefferson *et al.* 2008). Sperm whales mainly prey on mesopelagic squid but also eat giant squid and demersal and mesopelagic fish and some crustaceans, with males tending to eat larger individuals than females (Evans and Hindell 2004, Jefferson *et al.* 2008; Whitehead 2009). Stomach contents from stranded sperm whales in Tasmania contained remains of more than 50 species from 17 cephalopod families and some myctophid fish and other species, with high variability among individuals indicating that these whales are opportunistic predators that target locally abundant prey species (Evans and Hindell 2004).

Sperm whales have a long life span, slow growth and delayed maturation, a very low birth rate, polygynous breeding and a complex social structure, which makes them highly susceptible to over-exploitation (Whitehead 2009). Females reach sexually maturity around 9–12 years and give birth about every five years with birth rates declining among older age classes; gestation is about 14–16 months and females suckle their young for several years (Taylor *et al.* 2007, Bannister 2008, Whitehead 2009). Males usually don't breed until they are about 25 years or older, and continue to grow and reach physical maturity at about 50 years (Bannister 2008, Whitehead 2009). Maximum longevity is thought to be at least 70 years and possibly older (Whitehead 2003, Jefferson *et al.* 2008). Generation length is estimated to be 27.3–27.5 years (Taylor *et al.* 2007). These life history characteristics result in low reproductive rates and slow rates of population increase that limit the capacity for sperm whale populations to recover after depletion, and populations are particularly sensitive to reduced survivorship of mature breeding whales (Whitehead 2002, Taylor *et al.* 2008e, Woinarski *et al.* 2014).

Risks and threatening processes

Sperm whales have been recorded as bycatch in trawl nets internationally (Fertl and Leatherwood 1997), but are thought not to commonly interact with trawl fisheries (Elgin Associates unpublished (a)). Entanglement and bycatch has been recorded in a variety of other fisheries gear including gillnets and driftnets in the Mediterranean Sea and in other nets and lines in many other regions (Reeves *et al.* 2003, 2013, Taylor *et al.* 2008e, Zollett 2009). Small numbers have been taken in coastal fisheries in Indonesia, and under International Whaling Commission (IWC) Special Permit by Japan (Taylor *et al.* 2008e). Sperm whales are one of the main species involved in depredation of fish catch from longline fisheries at higher latitudes, and this interaction has resulted in some entanglements and deaths, and reports of fishers shooting these whales (Taylor *et al.* 2008e, Hamer *et al.* 2012). As high trophic level predators, sperm whales are potentially at risk from bioaccumulation of toxic pollutants in tissues, and high levels of organochlorines and metals have been recorded in tissues of sperm whales from Australia and in some regions overseas (Evans 2003, Evans *et al.* 2004). Ingestion of marine debris including plastics has been recorded in some sperm whales from Australia and overseas, in some cases resulting in gut obstruction and death (Evans and Hindell 2004, Woinarski *et al.* 2014). Vessel strikes are known to cause injury and in some cases death of sperm whales (Laist *et al.* 2001), and climate and oceanographic changes may alter trophic interactions and prey distribution in southern Australian waters (Evans *et al.* 2005).

Conservation and listing status

The sperm whale is listed as a cetacean species and as a migratory species under the EPBC Act, Vulnerable in NSW, Rare in SA, Priority 4 in WA, Data Deficient in the Northern Territory, but is not listed in other states within its Australian range (Woinarski *et al.* 2014). This species was recently assessed as Vulnerable in Australian waters (Woinarski *et al.* 2014), and as insufficiently known in the previous Australian status assessment (Bannister *et al.* 1996). Globally, the sperm whale was assessed as Vulnerable for the IUCN Red List in 2008 (Taylor *et al.* 2008e), and is listed in Appendix I of CITES.

Summary: odontocete species at risk from direct interactions with mid-water trawls in the SPF

- *The 15 odontocete species described above have different distribution ranges that vary in their extent of overlap with the SPF area. The species at highest risk of interactions with mid-water trawls in the SPF are bottlenose dolphins and short-beaked common dolphins whose diet includes small pelagic fish and these dolphins are known to interact extensively with trawl fisheries in Australia and internationally; some common bottlenose dolphins and possibly short-beaked common dolphins were previously recorded as bycatch in mid-water trawls in the SPF.*
- *The other odontocete species exhibit a wide range of biological and ecological characteristics including abundance, diet and life history traits, and the nature and extent of their interactions with trawl fisheries and other fisheries varies; hence the risks of interactions with the DCFA need to be assessed separately for each species.*
- *Although the hourglass dolphin remained at high risk for the mid-water trawl sector of the SPF after the residual risk assessment, its oceanic distribution range may only overlap marginally with the SPF area, the species has not been recorded interacting with trawl fisheries, is not obviously threatened, and is assessed as Least Concern in Australian waters and globally. Therefore, the panel did not consider the hourglass dolphin to be a particularly high risk species for direct interactions associated with the SPF mid-water trawl sector.*

Southern right whale *Eubalaena australis* (Level 2 PSA Residual Risk – Medium)

Distribution and range

Southern right whales have a circumpolar distribution in the Southern Hemisphere from about 20°S to about 55°S but are also recorded further south in Antarctic waters to about 65°S (Bannister *et al.* 1999, Jefferson *et al.* 2008, Bannister 2008). During the austral summer they occur on feeding grounds mainly between latitudes 40–55°S and part of the population migrates to warmer temperate waters for calving during winter, including coastal habitats along the southern Australian coast (Bannister 2008).

In the Australian region, southern right whales have an extensive distribution within Commonwealth and state waters, in some Australian Antarctic Territory waters, and in other oceanic areas south of Australia (Bannister *et al.* 1996, 1999, Pirzl 2008). Genetic analyses indicate that there are two subpopulations of these whales in Australian coastal waters, a larger south-west subpopulation and a smaller south-east subpopulation (Carroll *et al.* 2011). During winter, the southwestern subpopulation is distributed from about Ceduna in SA to Cape Leeuwin in WA with some whales recorded north to Exmouth (Pirzl 2008, Bannister 2011). The southeastern subpopulation is mainly distributed in waters south of Sydney in NSW, with a few individuals recorded as far north as Hervey Bay in Queensland (Franklin and Burns 2005, Pirzl 2008). The migration patterns and routes for these subpopulations are not well understood. Prior to whaling over-exploitation that caused severe declines, these whales had extensive calving grounds in southern Australia (Bannister 2008). The southwestern subpopulation is increasing, but the southeastern subpopulation remains relatively depleted with a restricted occupancy of coastal habitats following whaling (Pirzl 2008, Bannister 2011). The Australian distribution range of southern right whales overlaps extensively with the SPF area.

Population size and trends

The abundance of the total Australian population of southern right whales was estimated to about 3500 in 2009, with an estimated abundance of about 2900 whales including about 1220 adults in the larger southwestern subpopulation, and about 600 whales in the depleted southeastern subpopulation (Bannister 2011). Long-term monitoring of the southwestern subpopulation since 1976 has shown significant increases in abundance with an annual rate of increase of about 6.8 per cent during 1993–2010 (Bannister 2011). The southeastern subpopulation has not exhibited similar signs of recovery and appears to be relatively depleted (Pirzl 2008).

Whaling during the 1800s caused severe declines in abundance of these whales in Australian waters and throughout the Southern Hemisphere. The global population declined from an estimated size of about 55,000–70,000 prior to whaling down to about 300 whales by the 1920s, although more recent modelling indicates that the initial and minimum global population sizes may have been higher (Jackson *et al.* 2008). An estimated 26,000–40,000 southern right whales were killed in southeastern Australian and New Zealand waters from 1827 to 1930 and this unregulated whaling caused the commercial extinction of these whales (Bannister 1986). Protection from whaling in 1935 enabled some increase in abundance, but subsequent illegal Soviet whaling killed about 395 southern right whales in southern Australian and southern New Zealand waters between 1951 and 1971 that impaired this initial recovery phase (Tormosov *et al.* 1998, Clapham and Ivashchenko 2009). The global population in 2011 was estimated to have recovered to about 20–25 per cent of the original pre-whaling abundance (IWC 2011).

Biology and feeding ecology

Southern right whales are relatively large and rotund whales, growing up to 17 m long and weighing up to at least 80 t, with females growing larger than males (Jefferson *et al.* 2008). They aggregate along southern Australian coastal waters mainly between July and October, and usually occur within a few kilometres of the shore in shallow waters and sometimes within the surf zone (Bannister 2008). Females have high site fidelity to calving grounds, and their three-year calving cycle results in variable habitat occupancy along the southern Australian coast (Burnell 2001, Pirzl 2008). Individuals can travel westward over hundreds of kilometres along the southern Australian coast within a winter season (Burnell 2001, Bannister 2008), and larger scale movements of three right whales over 3700 km between southern Australian and subantarctic New Zealand regions have been recorded (Pirzl *et al.* 2009). Two of these whales were females with calves, which indicates that each female calved in both Australian and New Zealand winter calving grounds (Pirzl *et al.* 2009).

Major calving and coastal aggregation sites for the southwestern subpopulation include Head of Bight in SA, and Israelite Bay and Doubtful Island Bay regions in WA, with some smaller aggregation sites in these regions (Bannister *et al.* 1996, Burnell 2001, DSEWPaC 2012a). For the southeastern subpopulation, small and variable numbers of calving females aggregate off Warrnambool in Victoria, and small numbers of right whales are recorded from coastal Tasmania, Victoria, southern NSW and eastern SA (Pirzl 2008, DSEWPaC 2012a). These aggregation and breeding sites all overlap with the SPF area.

During spring and summer these whales migrate offshore to higher southern latitudes for pelagic feeding (Bannister 2008, Torres *et al.* 2013). They use surface skimming or shallow dives to trap plankton on their fine baleen, and this feeding behaviour makes them susceptible to vessel strike. They feed primarily on planktonic copepods and other crustaceans at latitudes below 40°S associated with the Polar Front, and south of 50–60°S their diet consists mainly of euphausiids (Bannister *et al.* 1996, Tormosov *et al.* 1998, Torres *et al.* 2013). Southern right whales from different breeding subpopulations apparently intermingle on southern pelagic feeding grounds and may potentially mate (Carroll *et al.* 2011, IWC 2011). Genetic analyses indicate that the southeastern and southwestern Australian subpopulations represent two distinct breeding stocks, and are consistent with maternal philopatry and male dispersal life history patterns with some recent historical or ongoing reproductive interchange (Carroll *et al.* 2011). Reduced breeding success in southern right whales has been correlated with increased sea surface temperatures associated with El Nino-Southern Oscillation events and climate change in the Australian population, and in the South Atlantic region (Leaper *et al.* 2006, Pirzl *et al.* 2008).

Southern right whales appear to have a promiscuous polygamous mating system whereby multiple males compete for breeding with females probably through sperm competition associated with sequential mating rather than by direct aggressive behaviour (Bannister *et al.* 1996, Jefferson *et al.* 2008). Sexual maturity occurs between five and nine years of age, and the mean calving interval is about 3.6 years but ranges from two to six years (Bannister *et al.* 1996, Burnell 2001). Maximum longevity is estimated to be more than 50 years (Bannister 2008) and may exceed 65 years (Burnell 2008). Generation length is estimated to be 28.8 years (Taylor *et al.* 2007). These life history characteristics result in relatively slow reproductive rates and slow capacity for populations to recover from depletion (Woinarski *et al.* 2014).

Risks and threatening processes

Southern right whales have been injured or killed from entanglements in fishing gear in Australian waters and internationally (Kemper *et al.* 2008, Reeves *et al.* 2013). Southern right whales are also known to be injured or killed from vessel collisions in Australian waters and internationally (Laist *et al.* 2001, Kemper *et al.* 2008). In the Australian region these whales face increased risk of vessel strike from heavy shipping traffic when they leave their southern Australian wintering grounds (Torres *et al.* 2013). Shipping movements are highest in the region used by the smaller and relatively depleted southeastern Australian subpopulation (DSEWPaC 2012a). Climate and oceanographic variability are known to affect foraging and subsequent reproductive success in southern right whales (Leaper *et al.* 2006, Pirzl *et al.* 2008), and habitat modelling indicates southward shifts and potential reduction in suitable foraging habitats in future, resulting from increased sea surface temperatures and altered oceanographic fronts (Torres *et al.* 2013). Other threats include increased port expansion and coastal development that may degrade coastal breeding and aggregation sites, which is particularly important for the smaller southeastern subpopulation, pollution, mortality from shooting, and increasing anthropogenic noise and acoustic disturbance from seismic surveys and other activities in southern Australian waters (Kemper *et al.* 2008, DSEWPaC 2012a, Woinarski *et al.* 2014).

Conservation and listing status

The southern right whale is listed as Endangered and as a cetacean species and as a migratory species under the EPBC Act, Threatened (Critically Endangered) in Victoria, Endangered in NSW and in Tasmania, Vulnerable in SA and in WA, and is not listed in Queensland (Woinarski *et al.* 2014). This species was recently assessed as Near Threatened in Australian waters (Woinarski *et al.* 2014), and was assessed as Vulnerable in the previous Australian status assessment (Bannister *et al.* 1996). Globally, the southern right whale was assessed as Least Concern for the IUCN Red List in 2008 (Reilly *et al.* 2008a).

Humpback whale *Megaptera novaeangliae* (Level 2 PSA Residual Risk – Medium)

Distribution and range

Humpback whales have a cosmopolitan distribution encompassing all the major ocean basins of the world, but are absent from some equatorial regions, a few enclosed seas and some areas of the high Arctic region (Clapham and Mead 1999, Jefferson *et al.* 2008). They migrate from tropical winter breeding and calving grounds to colder productive high latitude summer feeding grounds, except for the Arabian Sea population that is resident year-round (Bannister 2008, Clapham 2009).

In the Australian region, humpback whales have an extensive distribution within Commonwealth, state and territory waters including some Australian Antarctic Territory waters and in the Southern Ocean south of Australia during summer (Bannister *et al.* 1996, Thiele *et al.* 2000, Kemper *et al.* 2005, Bannister 2008). Two migratory subpopulations occur in Australian waters; the eastern Australian subpopulation designated 'E1', and the western Australian subpopulation designated 'D' by the Scientific Committee of the IWC. E1 humpback whales breed and calves are subsequently born in tropical coastal shelf areas along the northern coast of eastern Australia with putative breeding grounds within the Great Barrier Reef lagoon and possibly in the Coral Sea (Bannister 2008, Gales *et al.* 2010, Smith *et al.* 2012b). The western Australian D subpopulation breeds along the north-west coast of WA in the Kimberley region (Jenner *et al.* 2001, Bannister 2008). Australian humpback whales migrate south along the east or west coasts to feed in summer in the productive waters of the Southern Ocean south of 55°S (Dawbin 1966, Thiele *et al.* 2000, Gales *et al.* 2010, Franklin *et al.* 2012). A low level of interchange occurs between the western and eastern Australian subpopulations (Noad *et al.* 2000, Anderson *et al.* 2010), and a few individuals have been recorded moving between eastern Australia E1 and the Oceania E2 subpopulation in the western South Pacific (Olavarría *et al.* 2007, Garrigue *et al.* 2011). The Australian distribution range of humpback whales overlaps completely with the SPF area.

Population size and trends

The abundance of humpback whales in Australian waters and globally has varied substantially over the past hundred years reflecting the severe depletion from whaling during the 1900s, and subsequent ongoing recovery of some populations (reviewed in Bannister 2008, Reilly *et al.* 2008b, Woinarski *et al.* 2014). Commercial pelagic and coastal whaling, exacerbated by illegal Soviet whaling during the 1900s, caused a 95 per cent reduction in humpback whale abundance, and resulted in extirpation of some subpopulations (Clapham *et al.* 2008). In the Southern Hemisphere, a total of 215,840 humpback whales were killed between 1904 to 1983 (Clapham and Baker 2009), including 48,721 whales taken by Soviet whalers, of which only 2710 were reported to the IWC (Ivashchenko *et al.* 2011).

Whaling over-exploitation caused the near extinction of the eastern Australian E1 subpopulation, which was possibly reduced to a few hundred whales from a pre-exploitation abundance estimated to be about 22,000–25,700 whales (Chittleborough 1965, Jackson *et al.* 2009). After whaling ceased, monitoring of E1 whales migrating along the coast of northern NSW and southern Queensland has shown increasing abundance since 1978 (e.g. Bryden *et al.* 1990, Paterson 1991, Paterson *et al.* 2001, Noad *et al.* 2008, Paton *et al.* 2011). Abundance in 2010 was estimated to be about 14,522 whales based on surveys at North Stradbroke Island, Queensland (Noad *et al.* 2011). The E1 subpopulation has continued to increase rapidly with annual rates of increase of 10.5–10.9 per cent from 1984 to 2010 (Paterson *et al.* 2004, Noad *et al.* 2011), which approach the maximum plausible rate of 11.8 per cent annual growth for humpback whale populations (Zerbini *et al.* 2010).

Similar patterns of whaling-induced decline and post-whaling increases in abundance are evident in the western Australian D subpopulation that has been monitored since 1963 from Shark Bay, WA. The pre-whaling abundance was estimated to be about 20,000 whales or higher, declining to fewer than 1000 whales by around 1963, then increasing since the mid-1970s (Bannister and Hedley 2001, Bannister 2008). Surveys off Shark Bay in 2008 provided a best-abundance estimate of 28,830 (Hedley *et al.* 2011). Surveys from North West Cape about 350 km north of Shark Bay provided estimates of this subpopulation D increasing from about 7276 whales in 2000 to about 26,100 whales in 2008 (Salgado Kent *et al.* 2012). This subpopulation is thought to be one of the largest subpopulations of humpback whales (Salgado Kent *et al.* 2012), and estimated rates of annual increase have ranged from 10.15 per cent for the period 1982–1994 (Bannister and Hedley 2001) to 9.7 per cent from 1999 to 2008 (Hedley *et al.* 2011).

Biology and feeding ecology

Humpback whales are distinguished from most other baleen whales by their relatively robust body shape and extremely long pectoral flippers that are about one-third of their body length. Adult females are usually 1–1.5 m longer than males, with reliable records of maximum adult lengths around 16–17 m, although sizes of 14–15 m are more typical (Clapham and Mead 1999, Clapham 2009). Maximum adult weight is about 40–45 t (Bannister 2008, Jefferson *et al.* 2008). Humpback whales are mostly solitary or occur in small groups, with larger groups temporarily forming in breeding and feeding areas (Clapham 2009).

Humpback whales are highly migratory and typically undertake long annual return migrations of up to 16,000 to 18,800 km (Rasmussen *et al.* 2007, Robbins *et al.* 2011) from summer feeding grounds in high-latitude cold productive waters to their winter breeding and calving grounds in warm subtropical and tropical waters (e.g. Chittleborough 1965, Clapham 2009, Burns *et al.* 2014, Constantine *et al.* 2014). Australian humpback whales migrate through the SPF area. Temporal segregation of different sex, maturational and reproductive classes of whales is evident during these annual migrations (Dawbin 1966, Clapham 2000, Franklin *et al.* 2011). The eastern Australian E1 subpopulation is thought to breed within the Great Barrier Reef lagoon region (Simmons and Marsh 1986, Gales *et al.* 2010, Smith *et al.* 2012b), but some whales may breed near Chesterfield Reef in the Coral Sea (Bannister 2008). Therefore, further research is needed to identify the key aggregation and breeding grounds for eastern Australian humpback whales. On their southern migration to Antarctic waters, large numbers of E1 whales aggregate in subtropical Hervey Bay, Queensland (Paterson 1991, Corkeron *et al.* 1994, Chaloupka *et al.* 1999, Franklin *et al.* 2011). Major aggregation and calving sites for western Australian D subpopulation whales include the southern Kimberley region between Broome and Camden Sound, with migratory aggregation and resting areas at Exmouth Gulf and Shark Bay and some locations further south (Bannister and Hedley 2001, Jenner *et al.* 2001, Bannister 2008).

Although some intermingling of whales from the western and eastern Australian subpopulations occurs on summer feeding grounds south of Australia, there is limited gene flow between these subpopulations and low levels of genetic differentiation (Anderson *et al.* 2010, Schmitt *et al.* 2014). Similarly, the low levels of interchange between E1 and Oceania E2 subpopulation in the western South Pacific results in limited gene flow (Olavarría *et al.* 2007, Garrigue *et al.* 2011).

After leaving Australian coastal waters these whales migrate to the highly productive Southern Ocean waters south of 55°S where they gorge feed on massive swarms of Antarctic krill that aggregate near the ice edge during summer (Chittleborough 1965, Bannister 2008, Gales *et al.* 2010, Constantine *et al.* 2014). Analyses of humpback whale catches off eastern and western Australia during the 1950s showed little evidence of local feeding by migrating whales, although a few whales had recently fed (Chittleborough 1965, Bannister 2008). Some opportunistic feeding on schools of small fish including sardines and coastal krill *Nyctiphanes australis* has been observed along the Australian coast, with feeding on small schooling fish regularly recorded off Eden in southern NSW (Bannister *et al.* 1996, Stockin and Burgess 2005, Stamation *et al.* 2007, Gales *et al.* 2009), which would supplement energy reserves during migration. Humpback whales in the Northern Hemisphere also feed on small schooling fish including herring, mackerel, sardines, anchovies and capelin (Clapham and Mead 1999). These whales are gulp feeders that engulf large volumes of seawater and prey via expanding ventral pleats to greatly increase their mouth capacity, and the prey are subsequently trapped on baleen plates as seawater is expelled from the mouth (Clapham 2009). Some individuals or groups of humpback whales use bubbles to form bubble nets or curtains to concentrate their fish prey (Clapham 2009).

Humpback whales have a broadly promiscuous and polygamous mating system (Clapham 2000). Sexual maturity occurs between 4–11 years of age, and breeding is highly seasonal with calves born between June and September, with peak births during August (Chittleborough 1965). Females usually give birth every two to three years, but can breed in successive years (Clapham 2000). Gestation is 11 to 12 months and lactation occurs for 10 to 12 months (Clapham and Mead 1999). Maximum longevity was initially estimated to be about 48 years but recent reanalysis of the data indicates longevity is about 96 years (Fleming and Jackson 2011). Generation length is estimated to be 21.5 years (Taylor *et al.* 2007).

Risks and threatening processes

Two humpback whales (one alive, one dead) were reported as incidental bycatch in trawl nets in the Atlantic region off the northeastern US (Fertl and Leatherwood 1997). Humpback whales are susceptible to entanglement in a range of fisheries gear including gillnets, shark nets, trap nets, ropes and lines, and these entanglements are known to occur in Australian waters and internationally and can lead to serious injury or mortality (reviewed in Paterson 1990, Shaughnessy *et al.* 2003, Johnson *et al.* 2005, Cassoff *et al.* 2011, Reeves *et al.* 2013). These whales are also known to be injured or killed from vessel collisions (Laist *et al.* 2001, Redfern *et al.* 2013), and their surface behaviour and relatively shallow dives while travelling increase the risk of vessel strike. The rapidly increasing abundance of humpback whales in Australian waters will result in increased numbers of these whales becoming entangled or injured from interaction with vessels in future. Other threats include increasing anthropogenic noise and acoustic disturbance from seismic surveys and other activities (McCauley and Cato 2003, Zirbel *et al.* 2011), increasing port expansion and coastal development and associated increased vessel traffic that may affect migration pathways, coastal aggregation and breeding sites (Bannister *et al.* 1996, Woinarski *et al.* 2014), pollution (Evans 2003), and increased disturbance from whale-watching activities (Department of the Environment and Heritage 2005). Climate and oceanographic variability and change could alter the distribution and abundance of krill and other prey resources and affect lower latitude migratory and breeding habitats, and resumption of large-scale whaling is a potential threat to the recovery of humpback whale populations (Woinarski *et al.* 2014).

Conservation and listing status

The humpback whale is listed as Vulnerable and as a cetacean species and as a migratory species under the EPBC Act, Endangered in Tasmania, Threatened (Vulnerable) in Victoria, Vulnerable in Queensland, NSW, SA and in WA, and Least Concern in the Northern Territory (Woinarski *et al.* 2014). This species was recently assessed as Least Concern in Australian waters (Woinarski *et al.* 2014), and was assessed as Vulnerable in the previous Australian status assessment (Bannister *et al.* 1996). Globally, the humpback whale was assessed as Least Concern for the IUCN Red List in 2008 (Reilly *et al.* 2008b).

Bryde's whale *Balaenoptera edeni* (Level 2 PSA Residual Risk – Medium)

Distribution and range

Bryde's whales have a circumglobal distribution in tropical to temperate waters of the Pacific, Indian and Atlantic Oceans between latitudes 40°N and 40°S (Jefferson *et al.* 2008, Kato and Perrin 2009). They are unusual among balaenopterid whales in that they remain in tropical to warm-temperate waters where sea temperature is 16.3°C or warmer (Bannister 2008, Kato and Perrin 2009). In the Southern Hemisphere they may have a continuous distribution from eastern Australia to the central Pacific, and in the Indian Ocean their distribution extends west from WA (Kato and Perrin 2009). Distinct inshore and offshore forms of Bryde's whales are recorded in some regions, but the taxonomy and nomenclature of the 'Bryde's whale complex' is confused and the number of species or subspecies within this 'complex' is unclear (Best 2001, Reilly *et al.* 2008c, Kato and Perrin 2009).

In the Australian region, Bryde's whales have been recorded from all Australian state waters but there are no confirmed records from the Northern Territory (Bannister *et al.* 1996, Kemper *et al.* 2005, Arnold 2008). These whales are more likely to occur in warmer regions off the east and west coasts of Australia, particularly off Queensland and near the subtropical Abrolhos Islands and north of Shark Bay in WA, and are likely to be less abundant along the cooler southern Australian coast (Bannister *et al.* 1996, Bannister 2008). They have been observed in NSW near Byron Bay and in the Manning River, and from Scott Reef off northwestern Australia (Woinarski *et al.* 2014). There are 12 stranding records from southeastern Australia (Priddel and Wheeler 1997). In the Australian region, most individuals from the Indian Ocean conform to the larger 'ordinary' form of Bryde's whale, and three individuals from Victoria and one from WA are typical of *B. edeni* (Bannister *et al.* 1996, Bannister 2008). However, the identity of three individuals taken during whaling off WA and two off eastern Australia is uncertain, as they appear to be intermediate with other forms and may have been Omura's whales (Bannister 2008). The southern Australian distribution range of Bryde's whales overlaps partly with the SPF area.

Population size and trends

Bryde's whales are thought to be relatively uncommon off Australia but most records are from strandings, which may underestimate their abundance (Bannister 2008). There are no reliable estimates of population size or trends from Australian waters (Woinarski *et al.* 2014). Global abundance and the population trend of Bryde's whales are unknown, and estimates are complicated by the uncertain taxonomy, and whaling catch records that were combined with sei whales *Balaenoptera borealis* prior to 1972 (Reilly *et al.* 2008c). Whaling has reduced some populations, particularly in the western North Pacific (Reilly *et al.* 2008c). A total of 7881 Bryde's whales were taken in the Southern Hemisphere during the 1900s (Clapham and Baker 2009), including 1468 whales taken illegally by Soviet whalers, of which only 19 were reported to the IWC (Ivashchenko *et al.* 2011). Population estimates are available for some regions including about 10,000 in the Eastern Tropical Pacific and about 20,000 to 30,000 in the North Pacific but Southern Hemisphere populations have not been reassessed in recent decades (Reilly *et al.* 2008c, Jefferson *et al.* 2008).

Biology and feeding ecology

Female Bryde's whales are larger than males and may grow to about 16.5 m long while males grow to about 15.0 m, and maximum weight is about 40 t (Jefferson *et al.* 2008). Bryde's whales from the Southern Hemisphere are larger than those from the Northern Hemisphere, and the offshore pelagic form is larger than the smaller coastal form (Best 2001, Kato and Perrin 2009).

Migratory movements from higher latitudes in spring-summer toward equatorial regions in autumn-winter occur in some populations of the larger offshore pelagic form, but movement patterns of other Bryde's whales are poorly known (Best 2001, Reilly *et al.* 2008c, Kato and Perrin 2009). Genetic structure is evident within and between different ocean basins and hemispheres which indicates that these populations should be considered as separate management units (Kanda *et al.* 2007).

Bryde's whales feed mainly on pelagic schooling fishes, including anchovy, sardine, mackerel, pilchard and herring (Kato and Perrin 2009). They also feed opportunistically on some crustaceans including euphausiids, copepods, pelagic red crabs and on cephalopods (Best 2001, Kato and Perrin 2009). The inshore form appears to be more reliant on schooling fish whereas the offshore form may feed more on euphausiids, but they are also recorded to alter pelagic feeding from fish to euphausiids in different years (Best 2001, Bannister 2008, Kato and Perrin 2009). Bryde's whales lunge-feed on extensive schools of anchovies at Cape Cuvier in WA, and stomach contents of whales taken near the Western Australian coast contained large quantities of anchovies (Bannister 2008). These whales have also been observed feeding on large schools of small fish near Byron Bay, NSW and in other coastal locations (Arnold 2008, P. Beeman pers. comm. in Woinarski *et al.* 2014). Bryde's whales have also been observed using 'bubble net' feeding to concentrate prey (Kato and Perrin 2009). These whales are mostly solitary or occur in small groups, with larger groups of 10–20 whales observed on feeding grounds (Jefferson *et al.* 2008).

Breeding occurs over an extended season for the inshore form of Bryde's whales off South Africa, whereas offshore pelagic stocks have a winter peak in calving but their breeding grounds are largely unknown (Best 2001, Bannister 2008, Kato and Perrin 2009). Sexual maturity occurs at about seven years, and age at first reproduction is about eight to nine years (Kato and Perrin 2009). Gestation lasts for about 11–12 months, calves are weaned at about six months and the calving interval is about two years (Bannister 2008, Kato and Perrin 2009). Generation length is estimated to be about 18.4 years (Taylor *et al.* 2007).

Risks and threatening processes

Bryde's whales are occasionally recorded as bycatch in fisheries gear (Reilly *et al.* 2008c, Reeves *et al.* 2013), and one whale entangled in fishing gear around the mouth was considered to have probably died from impaired foraging and starvation over a long period (Cassoff *et al.* 2011). Anderson (2014) reviewed available information on baleen whales that are known to associate with oceanic tuna schools in the tropical Indian Ocean, and this association is used by purse seiners in the region to locate tuna schools. Although the whale species involved in these fishery interactions are uncertain in some areas, Anderson (2014) concluded that Bryde's whale are the main whale species involved with purse seine operations targeting tuna schools in the major fishing area east of the Seychelles. This species is also occasionally recorded to be injured or killed by vessel strike (Laist *et al.* 2001, Reilly *et al.* 2008c). Expansion of pelagic fisheries targeting schooling pelagic fishes such as anchovy, which is an important prey species for Bryde's whales, may increase direct and indirect interactions with these whales (Bannister *et al.* 1996, Elgin Associates unpublished (a)). Up to 50 Bryde's whales have been taken by Japanese whalers in the North Pacific Ocean under IWC Special Permit, and small numbers are taken by artisanal whalers in Indonesia (Jefferson *et al.* 2008). Other threats include increasing anthropogenic noise and acoustic disturbance, increasing port expansion and coastal development and associated increased vessel traffic (Bannister *et al.* 1996, Woinarski *et al.* 2014), and pollution (Evans 2003). Plastic and packaging film were found tightly packed in the stomach of a Bryde's whale that stranded and died near Cairns in Queensland (Arnold 2008).

Conservation and listing status

The Bryde's whale is listed as a cetacean species and as a migratory species under the EPBC Act, Data Deficient in Victoria and in the Northern Territory, Rare in SA, and is not listed in Queensland, NSW, Tasmania and WA (Woinarski *et al.* 2014). This species was recently assessed as Data Deficient in Australian waters (Woinarski *et al.* 2014), and was assessed as 'No category assigned but possibly secure' in the previous Australian status assessment (Bannister *et al.* 1996). Globally, the Bryde's whale was assessed as Data Deficient for the IUCN Red List in 2008 (Reilly *et al.* 2008c), and is listed in Appendix I of CITES.

Sei whale *Balaenoptera borealis* (Level 2 PSA Residual Risk – Medium)

Distribution and range

The sei whale has a cosmopolitan distribution and occurs in oceanic areas in all major ocean basins, but tends to be less common in shallower continental shelf seas (Jefferson *et al.* 2008, Horwood 2009). Two genetically different subspecies have been proposed, with *B. borealis schlegellii* occurring in the Southern Hemisphere including Australian waters (Rice 1998, Horwood 2009). Sei whales are thought to complete long annual seasonal migrations from subpolar summer feeding grounds to lower latitude winter breeding grounds but details of their migrations and locations of breeding grounds are largely unknown (Horwood 2009). In the Southern Hemisphere, sei whales mainly occur from 45–60°S during summer but some whales are recorded further south (Parker 1978, Thiele *et al.* 2004, Bannister 2008, Woinarski *et al.* 2014).

In the Australian region, sei whales are recorded from Australian Antarctic Territory waters and Commonwealth waters, with infrequent records off Tasmania, NSW, Queensland, the GAB and Western Australia (Parker 1978, Bannister *et al.* 1996, Thiele *et al.* 2000). A sei whale carcass was trawled from 113 m depth about 160 km offshore the Northern Territory (Chatto and Warneke 2000). Parker (1978) noted that sei whales were the most commonly observed whales during Australian National Antarctic Research Expedition voyages in the 1960s and 1970s. These whales are not commonly recorded near Australian mainland waters, but they are occasionally observed feeding in the Bonney Upwelling region in southern Australia during summer and autumn (Gill 2002, Miller *et al.* 2012). Sei whales including females with calves have been reported near the coast and 40 km south of Tasmania, and at 37°S, south of SA (Bannister 2008). Four sei whales were taken from mainland whaling stations between 1958–1963, and sei whales or Bryde's whales were commonly sighted by sperm whalers off Albany in WA during the 1900s (Bannister 2008). The Australian distribution range of sei whales overlaps completely with the SPF area.

Population size and trends

Sei whale population size and trends in Australian waters are unknown (Woinarski *et al.* 2014). Whaling significantly depleted sei whale populations in all regions and their global abundance is poorly known (Reilly *et al.* 2008d, Horwood 2009). A total of 203,843 sei whales were killed in the Southern Hemisphere last century (Clapham and Baker 2009), including 59,327 whales taken by Soviet whalers, of which only 33,001 were reported to the IWC (Ivashchenko *et al.* 2011). Global abundance was estimated to be about 130,000 in the 1930s and rapidly decreased during whaling in the 1960s to less than 20,000 in the 1970s (Reilly *et al.* 2008d). Whaling impacts were particularly severe in the Southern Hemisphere where sei whale abundance was estimated to have decreased by about 89 per cent from about 98,000 in 1930 down to about 11,000 in 2007 (Reilly *et al.* 2008d). This estimated severe decline corresponds to declining sightings and catches during the 1960s and 1970s (Parker 1978, Reilly *et al.* 2008d). The global population trend is unknown (Reilly *et al.* 2008d).

Biology and feeding ecology

Sei whales are sleek, streamlined whales that grow to almost 20 m but are more typically 15–17 m long, they weigh up to 20–45 t, and females are slightly larger than males (Jefferson *et al.* 2008, Horwood 2009). These whales mainly occur in offshore oceanic regions although some occur in coastal waters, and seasonal feeding and breeding cycles strongly influence their distribution and latitudinal movements (Horwood 2009).

Sei whales have greater flexibility in feeding techniques than other baleen whales and skim feed on copepods and amphipods in mid-latitudes using their relatively fine baleen fringes, whereas in higher latitude waters they lunge feed on Antarctic krill (Bannister 2008, Jefferson *et al.* 2008). These whales also lunge-feed on small schooling fish including sardines and anchovies, and feed on cephalopods when encountered (Jefferson *et al.* 2008, Horwood 2009). Sei whales are mostly solitary during migrations, but form small groups of two to five whales in warmer waters, with larger aggregations of 20–100 whales on feeding grounds (Horwood 2009).

Breeding occurs mainly in winter, and sei whales may occasionally hybridise with fin whales *Balaenoptera physalus* (Jefferson *et al.* 2008). Age at sexual maturity and age at first reproduction are estimated to be about 9–10 years (Taylor *et al.* 2007, Horwood 2009). Gestation lasts about 10–12 months, calves are weaned by six to nine months, and the mean calving interval is estimated to be about 2.5 years (Taylor *et al.* 2007, Jefferson *et al.* 2008). Longevity is estimated to be about 60 years (Bannister *et al.* 1996). Generation length is estimated to be 23.3 years (Taylor *et al.* 2007).

Risks and threatening processes

Two sei whales have been reported killed by vessel strike in the Northern Hemisphere (Laist *et al.* 2001, Reilly *et al.* 2008d). Sei whales have been reported entangled and drowned in fishing gear in coastal waters, and Japanese whalers in the North Pacific Ocean have an annual take of 100 whales under IWC Special Permit (Reilly *et al.* 2008d, Jefferson *et al.* 2008). Other threats include increasing anthropogenic noise and acoustic disturbance, habitat degradation, pollution, and climate and oceanographic variability and change (Bannister *et al.* 1996, Woinarski *et al.* 2014).

Conservation and listing status

The sei whale is listed as Vulnerable and as a cetacean species and as a migratory species under the EPBC Act, Vulnerable in SA and in WA, not Listed (Data Deficient) in Victoria, Data Deficient in the Northern Territory, and is not listed in Queensland, NSW and Tasmania (Woinarski *et al.* 2014). This species was recently assessed as Endangered in Australian waters (Woinarski *et al.* 2014), and was assessed as Vulnerable in the previous Australian status assessment (Bannister *et al.* 1996). Globally, the sei whale was assessed as Endangered for the IUCN Red List in 2008 (Reilly *et al.* 2008d), and is listed in Appendix I of CITES.

Fin whale *Balaenoptera physalus* (Level 2 PSA Residual Risk – Medium)

Distribution and range

The fin whale has a cosmopolitan distribution and occurs in all major ocean basins and some seas, but is uncommon or absent from equatorial and high latitude ice habitats (Jefferson *et al.* 2008, Aguilar 2009). Three subspecies are now recognised, with the subspecies *B. physalus quoyi* occurring in the Southern Hemisphere including Australian waters (Aguilar 2009, Committee on Taxonomy 2014). Fin whales in the Southern Hemisphere complete long annual seasonal migrations from higher latitude summer feeding grounds to lower latitude winter breeding grounds (Aguilar 2009), but more variable and complex movement patterns are evident in some regions of the Northern Hemisphere (Mizroch *et al.* 2009). In the Southern Hemisphere, fin whales mainly occur from 40–65°S during summer (Reilly *et al.* 2008e).

In the Australian region, fin whales occur within Commonwealth waters and most state waters, and from Australian Antarctic Territory waters (Bannister *et al.* 1996, Thiele *et al.* 2000, Bannister 2008). They are infrequently recorded in coastal areas around Australia (Bannister *et al.* 1996, Bannister 2008), but their calls have been recorded off WA during autumn and winter, and off southern Australia and sporadically off NSW (Gedamke *et al.* 2007, McCauley *et al.* 2000, R. McCauley pers. comm. in Woinarski *et al.* 2014). Fin whale calls have been recorded in Australian Antarctic Territory waters from January to February and April to June (Gedamke *et al.* 2007, Širović *et al.* 2009, Gedamke and Robinson 2010). Fin whales are occasionally sighted in the Bonney Upwelling region off Victoria in summer and autumn, including a female and calf in April 2000 (Gill 2002, Miller *et al.* 2012). The Australian distribution range of fin whales overlaps completely with the SPF area.

Population size and trends

Fin whale abundance and population trend in Australian waters are unknown (Woinarski *et al.* 2014). Globally, fin whales were very abundant prior to commercial whaling. Their estimated global abundance was about 400,000 whales in 1920, of which about 325,000 occurred in the Southern Hemisphere (Reilly *et al.* 2008e). A total of 725,331 fin whales were killed during commercial whaling last century in the Southern Hemisphere, which severely depleted the population by more than 70 per cent (Reilly *et al.* 2008e, Clapham and Baker 2009). More recently, 748 fin whales were recorded during the 2005–06 summer from the Antarctic region south of WA, with more than 100 whales sighted off the Ross Sea (Bannister 2008). Fin whales are relatively abundant in the North Pacific and North Atlantic, and global abundance may be about 140,000 (Jefferson *et al.* 2008). The global population trend is unknown, but some populations may be increasing following the cessation of whaling (Reilly *et al.* 2008e).

Biology and feeding ecology

Fin whales are the second-largest whale species and have a sleek and streamlined body (Jefferson *et al.* 2008). In the Southern Hemisphere females grow up to about 26–27 m and males to about 25 m, while fin whales in the Northern Hemisphere are less than 24 m (Jefferson *et al.* 2008, Aguilar 2009). The largest fin whales weigh up to 120 t, but most weigh less than 90 t (Jefferson *et al.* 2008).

Fin whales mostly occur in oceanic pelagic habitats and tend to aggregate in areas of high productivity, but commonly occur in coastal waters in some regions (Aguilar 2009). Around Australia, they are thought to occur mostly in deeper water habitats, with coastal records including a small number taken from mainland whaling stations and occasional stranding records (Bannister *et al.* 1996, Bannister 2008). Fin whales are mostly solitary or occur in small groups of up to seven whales, while larger groups of more than 100 whales may form during feeding (Bannister 2008). Fin whales sometimes form mixed feeding schools with blue whales *Balaenoptera musculus*, and the two species are known to occasionally interbreed (Aguilar 2009).

Fin whales lunge feed on dense swarms of crustaceans or small schooling fish by gulping large volumes of water and prey that become trapped on their baleen plates. Southern Hemisphere fin whales feed mainly on Antarctic krill *Euphausia superba and E. vallentini* and occasionally on other planktonic crustaceans (Bannister 2008, Aguilar 2009). In the Northern Hemisphere, they feed on euphausiids and other crustaceans, schooling fishes including herring, capelin *Mallotus villosus* and mackerel, and sometimes squid (Aguilar 2009). Dives range from 100–200 m depths with maximum depths of 500 m (Bannister 2008).

Breeding begins in late autumn and calving occurs mainly in winter (Mizroch *et al.* 2009). Females reach sexual maturity at about eight years with age at first reproduction estimated to be 9–10 years (Taylor *et al.* 2007, Reilly *et al.* 2008e). Gestation is about 10–11 months, weaning occurs after six to eight months, and the mean calving interval is about 2.2 years (Taylor *et al.* 2007, Bannister 2008). Longevity is estimated to be 90–100 years (Bannister *et al.* 1996). Generation length is estimated to be 25.9 years (Taylor *et al.* 2007).

Risks and threatening processes

Fin whales have been reported feeding behind a trawl codend (Fertl and Leatherwood 1997), and are occasionally recorded as bycatch in fishing gear (Reilly *et al.* 2008e, Zollett 2009, Reeves *et al.* 2013). Fin whales are one of the most commonly recorded large whale species involved with vessel collisions and vessel strike is known to cause injury and deaths, particularly in the Mediterranean fin whale population (Laist *et al.* 2001, Reilly *et al.* 2008e, Redfern *et al.* 2013). A small number of fin whales were taken by Japanese whalers under IWC Special Permit in the Antarctic region (Clapham and Baker 2009). Other threats include increasing anthropogenic noise and acoustic disturbance, habitat degradation, pollution, and climate and oceanographic variability and change (Bannister *et al.* 1996, Woinarski *et al.* 2014).

Conservation and listing status

The fin whale is listed as Vulnerable and as a cetacean species and as a migratory species under the EPBC Act, Vulnerable in Tasmania, SA and in WA, not listed (Data Deficient) in Victoria, and is not listed in NSW and Queensland (Woinarski *et al.* 2014). This species was recently assessed as Endangered in Australian waters (Woinarski *et al.* 2014), and was assessed as Vulnerable in the previous Australian status assessment (Bannister *et al.* 1996). Globally, the fin whale was assessed as Endangered for the IUCN Red List in 2008 (Reilly *et al.* 2008e), and is listed in Appendix I of CITES.

Blue whale *Balaenoptera musculus* (Level 2 PSA Residual Risk – Medium)

Distribution and range

Blue whales have a cosmopolitan distribution and are recorded from all oceans except the Arctic, with separate populations in the Southern Hemisphere, North Pacific and North Atlantic oceans (Reilly *et al.* 2008f, Bannister 2008, Jefferson *et al.* 2008). Four blue whale subspecies are recognised, with two subspecies occurring in Australian waters and elsewhere in the Southern Hemisphere: the Antarctic blue whale *B. musculus intermedia*, and the pygmy blue whale *B. musculus brevicauda* (Rice 1998, Bannister 2008, Committee on Taxonomy 2014). These two subspecies have different morphology, distribution, genetics, reproductive characteristics and vocal behaviours (e.g. Branch *et al.* 2007, 2009, Attard *et al.* 2010, McCauley and Jenner 2010).

Antarctic blue whales have a circumpolar distribution in the Southern Hemisphere and during summer they occur in Antarctic feeding grounds from the pack ice zone northward to the Antarctic Convergence around 52–56°S (Branch *et al.* 2007; Samaran *et al.* 2010). They are thought to migrate to lower latitude areas in winter but some remain in Antarctic waters over winter (Bannister *et al.* 1996, Stafford *et al.* 2004, Branch *et al.* 2007, Širović *et al.* 2009). Pygmy blue whales occur in the Indian Ocean and Southern Ocean from the Madagascar Plateau to WA and across southern Australia to Tasmania, with northward migrations to lower latitude regions including Indonesia (Branch *et al.* 2007, Gales *et al.* 2010, Double *et al.* 2014). Pygmy blue whales occur mainly at latitudes north of 54°S during the summer feeding season although some occur further south off Antarctica at latitudes 65–69°S (Branch *et al.* 2007, Attard *et al.* 2012).

In the Australian region, blue whales have been recorded from all state and Northern Territory waters, Australian Antarctic Territory waters and in the Southern Ocean south of Australia (e.g. Bannister *et al.* 1996, Thiele *et al.* 2000, Gill 2002, Bannister 2008, Širović *et al.* 2009). Antarctic blue whales occur in Australian Antarctic Territory waters during the summer feeding season and can undertake extensive movements (Thiele *et al.* 2000, Gedamke and Robinson 2010, Double *et al.* 2013). Some of these whales subsequently migrate to lower latitude winter breeding grounds in the Indian and Pacific oceans, but these breeding grounds are not yet well defined (Stafford *et al.* 2004, Branch *et al.* 2007). Antarctic blue whales have been recorded off Tasmania and at Cape Leeuwin, Geographe Bay and Perth Canyon off WA mainly from May to November, hence these areas may represent important migratory or breeding habitats (Stafford *et al.* 2004, Gedamke *et al.* 2007).

Feeding aggregations of pygmy blue whales occur mainly from November to May in the Bonney Upwelling off Victoria and SA, and in the Perth Canyon off WA (Gill 2002, Rennie *et al.* 2009, Gill *et al.* 2011, Miller *et al.* 2012, Double *et al.* 2014). Genetic analysis indicates that these whales are part of the same breeding subpopulation (Attard *et al.* 2010). Pygmy blue whales also migrate along the coast of Western Australia and some satellite tagged whales migrated north across the Timor Sea and into the Banda Sea and Molucca Sea regions in Indonesian waters (McCauley and Jenner 2010, Gales *et al.* 2010, Double *et al.* 2012, 2014). Pygmy blue whales also occur along the east coast of Australia but their migration routes are not known (McCauley and Jenner 2010). 'Tasman-Pacific' type pygmy blue whale calls have been regularly detected along the east coast and this subpopulation may use the Tasman Sea area over an extended period or year-round (McCauley *et al.* 2013). The Australian distribution range of blue whales overlaps completely with the SPF area.

Population size and trends

The total abundance of blue whales and the abundance of the subspecies in Australian waters are unknown (Woinarski *et al.* 2014). Prior to commercial whaling in 1904, blue whales were very abundant in the Southern Hemisphere. Total global whaling catches of blue whales last century are estimated to have been 382,595 (Branch *et al.* 2008), with 362,770 killed in the Southern Hemisphere from 1904–1973 (Clapham and Baker 2009). Soviet whalers killed 13,035 Antarctic and pygmy blue whales in the Antarctic from the 1950s to the early 1970s but only 3651 of these were reported to the IWC (Ivashchenko *et al.* 2011). The global abundance of blue whales is uncertain but is thought to be in the range 10,000–25,000, and the population trend is increasing (Reilly *et al.* 2008f).

The pre–whaling abundance of Antarctic blue whales in the Southern Hemisphere was estimated to be about 239,000, but these whales were severely overexploited, down to 0.15 per cent of this abundance resulting in only about 360 whales estimated to remain in 1973 (Branch *et al.* 2004). Antarctic blue whales were estimated to have increased to about 2280 by 1996, increasing at about 7.3 per cent per year (Branch *et al.* 2004, 2007). The abundance of pygmy blue whales is uncertain. Their pre-whaling abundance was likely to be an order of magnitude lower than for the Antarctic blue whale, and they may have been less depleted by whaling (Branch *et al.* 2007). The estimated abundance of pygmy blue whales from the Perth Canyon during surveys in 2000 to 2005 was 532–1754 whales (Jenner *et al.* 2008).

Biology and feeding ecology

Blue whales are the largest animals ever known to occur, and despite their huge size they have a relatively slender and streamlined body (Jefferson *et al.* 2008). Southern Hemisphere whales are a larger average size than those in the Northern Hemisphere, and females are larger than males (Sears and Perrin 2009). The largest blue whales recorded were 31.7–32.6 m, and the maximum recorded weight was about 190 t female, but adults mostly range from 50–150 t (Sears and Perrin 2009).

Blue whales are highly mobile and migratory, with one satellite-tagged Antarctic blue whale tracked moving over 5300 km in the Southern Ocean and Australian Antarctic Territory waters over 74 days (Andrews-Goff *et al.* 2013). They mostly occur in deeper water pelagic habitats with high productivity and zooplankton densities in the Antarctic and subantarctic regions during the austral summer, and along oceanographic fronts and in upwelling areas (Branch *et al.* 2007, Rennie *et al.* 2009). Many blue whales subsequently migrate to lower latitude putative feeding, breeding and calving grounds during the austral winter, but these are currently poorly defined and some blue whales may not migrate each year (Branch *et al.* 2007, Širović *et al.* 2009). At lower latitudes these whales aggregate in deeper waters along continental margins, and in some shallower habitats in the Bonney Upwelling off southern Australia and in Geographe Bay, WA (Branch *et al.* 2007, Gill *et al.* 2011). The pygmy blue whale subpopulation that occurs in southern Australian waters migrates north along WA to Indonesia, and the Banda and Molucca Seas region may be a calving and breeding area for this subpopulation (Branch *et al.* 2007, McCauley and Jenner 2010, Gales *et al.* 2010, Attard *et al.* 2010, Double *et al.* 2012, 2014). Recent genetic analyses of Antarctic blue whale biopsy samples have shown significant population structure among the six Antarctic management areas designated by the IWC that may result from some degree of female fidelity to Antarctic feeding grounds, and reflect the distribution and abundance of krill (Sremba *et al.* 2012).

Blue whales feed almost exclusively on krill, and use lunge feeding to engulf large swarms near the surface or by diving to 100 m or deeper (Sears and Perrin 2009). Antarctic blue whales in Antarctic waters feed primarily on Antarctic krill, and feed on other *Euphausia* species at lower latitudes (Branch *et al.* 2007, Bannister 2008a, Samaran *et al.* 2010). Pygmy blue whales feed on smaller *Nyctiphanes australis* euphausiids in southern Australian waters (Gill 2002), and deep water *Euphausia recurva* in the Perth Canyon at depths of 200–300 m (Rennie *et al.* 2009).

Female Antarctic blue and pygmy blue whales reach sexual maturity at about 10 years of age (Branch 2008), and age at first reproduction is about 11 years (Taylor *et al.* 2007). Antarctic blue whales breed in June to July, gestation extends for 10–11 months, and pregnant females calve in April to May the following year (Branch 2008). Mean calving interval is about 2.5–2.6 years (Taylor *et al.* 2007, Branch 2008). Lifetime ovulation rate for Pygmy Blue whales average 7.6, which is significantly lower than for Antarctic blue whales which average 13.6; these rates indicate that pygmy blue whales may recover more slowly from whaling impacts (Branch *et al.* 2009). Maximum longevity is estimated to be at least 80–90 years but is likely to be longer (Sears and Perrin 2009). Generation length has been estimated to be 30.8 years (Taylor *et al.* 2007). These life history characteristics result in a relatively low reproductive rate resulting in a slow capacity for recovery from the massive over-exploitation from whaling last century (Woinarski *et al.* 2014).

Risks and threatening processes

There are few reports of lethal entanglements of blue whales, but 12 per cent of blue whales in eastern Canadian waters have scars indicating that they had made contact with fishing gear (Sears and Perrin 2009). Their large size and power may enable most blue whales to tear through fishing gear if contact occurs (Sears and Perrin 2009). Vessel strike is known to cause injury and in some cases death of blue whales (Laist *et al.* 2001, Redfern *et al.* 2013), and shipping movements are increasing in Australian waters used by these whales. At least 25 per cent of identified blue whales in the St. Lawrence area have scars from vessel collisions including whale-watching vessels, and scars from vessel strikes are known from other regions particularly in areas of heavy shipping traffic (Sears and Perrin 2009). Blue whales are also recorded to react strongly to approaching vessels, hence increased noise and disturbance from vessel traffic is a threat to recovering populations (Sears and Perrin 2009, Double *et al.* 2014). Other forms of anthropogenic noise including military active sonar

(Goldbogen *et al*. 2013) and seismic surveys (Di Iorio and Clark 2010) that can cause disturbance and avoidance behavior in blue whales, and disturbance from seismic surveys may be important in pygmy blue whale habitats in southern Australia (Gill *et al*. 2011, Double *et al*. 2014). Persistent pollutants may affect the health status of blue whales, and PCBs are commonly found in whales from eastern Canadian waters (Sears and Perrin 2009). Climate and oceanographic variability and change are likely to alter sea-ice habitats (Nicol *et al*. 2008) and other environmental conditions in the Southern Ocean and other important habitats for blue whales, and may alter the distribution and availability of essential krill prey resources (Atkinson *et al*. 2004, Flores *et al*. 2012, Woinarski *et al*. 2014).

Conservation and listing status

The blue whale is listed as Endangered and as a cetacean species and as a migratory species under the EPBC Act, Threatened (Critically Endangered) in Victoria, Endangered in NSW, Tasmania, SA and in WA, Data Deficient in the Northern Territory, and is not listed in Queensland (Woinarski *et al*. 2014). This species was recently assessed as Endangered in Australian waters (Woinarski *et al*. 2014). In the previous Australian status assessment by Bannister *et al*. (1996) the conservation status of the full blue whale species was not assessed; the 'true' blue whale *B. m. musculus* (Antarctic blue whale) was assessed as Endangered and the 'pygmy' blue whale was assessed as 'No category assigned because of insufficient information'. Globally, the blue whale was assessed as Endangered for the IUCN Red List in 2008 (Reilly *et al*. 2008f), and is listed in Appendix I of CITES.

Summary: mysticete whale species at risk from direct interactions with mid-water trawls in the SPF

- *The six baleen whale species described above have different distribution ranges and these overlap extensively or completely with the SPF area. Five of these species are listed as threatened species and are therefore matters of national environmental significance requiring a high level of protection under the EPBC Act. Southern right whales and blue whales are listed as Endangered, while fin, sei and humpback whales are listed as Vulnerable.*
- *The six whale species exhibit different biological and ecological characteristics, and their abundance and the extent to which populations are recovering following significant depletion from whaling, varies. Their diet and life history traits, and the nature and extent of potential interactions with fisheries operations also differ between species. Bryde's whales feed mainly on small pelagic schooling fishes, while fin, sei and humpback whales feed mainly on crustaceans but also feed on small pelagic fish species to varying degrees.*
- *Humpback whale abundance is increasing rapidly and the southwest subpopulation of southern right whales is also increasing, hence there is increased risk of vessel strike and other interactions such as entanglement in fishing gear for these species within the SPF area. Vessel strike has also been recorded for the other whale species, particularly for fin whales. Entanglement or bycatch in various types of fishing gear has been reported for all six whale species. Occasional incidental bycatch in trawl nets and other fishing gear has been reported for humpback, fin and Bryde's whales, and fin whales have been reported feeding behind a trawl codend. Therefore these whale species have a wide range of known and potential interactions with mid-water trawl and other fisheries.*

5.3.2 Nature and extent of interactions

The nature of interactions and likelihood of cetaceans directly interacting with the mid-water trawl sector of the SPF vary significantly among the 21 species reviewed in Section 5.3.1. Many of the smaller odontocete cetaceans are known to interact with trawl nets and other fishing gear leading to entanglement and bycatch, and are at some risk from vessel collision and other anthropogenic threats including acoustic disturbance. The seven great whale species (southern right, humpback, Bryde's, sei, fin and blue whale mysticete species, and the odontocete sperm whale species) are at risk from trawlers and other vessels from collisions and from other anthropogenic threats including acoustic disturbance, and some of these whale species interact with trawl nets and are at risk from entanglement and bycatch in fishing gear. Some cetacean species in the SPF area feed on small pelagic fish species, which increases the potential for interactions. Short-beaked common dolphins and *Tursiops* spp. bottlenose dolphins are at higher risk as they have been recorded to interact extensively with trawl fisheries in Australian waters and internationally. Furthermore, common bottlenose dolphins and possibly short-beaked common dolphins have been recorded as bycatch in mid-water trawls in the SPF (Lyle and Willcox 2008). Three common dolphins were recorded as bycatch in the GHAT sector of the SESSF during 2009 and 2010 (Tuck *et* *al*. 2013). Eighty dolphins were reported as bycatch mortality interactions (with another four dolphins reported alive) in the GHAT sector of the SESSF during the three seasons from May 2011 to April 2013 (AFMA 2013f, 2014d). Analysis of 40 of the dolphin mortalities indicated that 38 were common dolphins and two were bottlenose dolphins (AFMA 2013f). Hundreds of short-beaked common dolphins are estimated to have died in purse seine fisheries targeting small pelagic fish species in the SASF (Hamer *et al*. 2008).

Tuck *et al*. (2013) reviewed the available information on fisheries bycatch in key Commonwealth fisheries including the mid-water trawl sector of the SPF for the period 2001–2010. Reported marine mammal interactions with mid-water trawls in the SPF during this period comprised of 184 reported pinniped interactions (refer to Section 5.2.2), and 25 reported dolphin mortalities in mid-water trawls during 2001–2009. At the commencement of the mid-water trawl operations in late 2002, a 'soft' rope-mesh SED was used and there was a high level of observer coverage (Lyle and Willcox 2008). In October 2004, 14 short-beaked common dolphins or common bottlenose dolphins (species not confirmed) died in two separate mid-water trawl tows to the east of Flinders Island, and in November 2004 three common bottlenose dolphins died in a tow about 150 nm further south (Lyle and Willcox 2008, Tuck *et al*. 2013). In April 2005, one unidentified dolphin was killed in a tow off eastern Tasmania, and in May 2005 seven unidentified dolphins were killed in tows in this region (Lyle and Willcox 2008, Tuck *et al*. 2013).

Tuck *et al*. (2013) noted that there had been no reported incidental interactions with dolphins and mid-water trawls in the SPF since June 2005, after the introduction of bycatch management measures. Furthermore, no interactions with cetaceans have been reported with mid-water trawl gear in the SPF since that time (AFMA 2014c). The absence of reported interactions with cetaceans and other TEPS coincided with a reduction in fishing effort in the SPF fishery, a decline in observer coverage to less than 13 per cent of observed shots since 2007, no observer coverage and little or no fishing in 2010 and 2011, and no mid-water trawl fishery catches in 2011 (Moore and Skirtun 2012, Tuck *et al*. 2013). Tuck *et al*. (2013) concluded "overall bycatch levels are difficult to estimate, given a decline in on-board observer coverage on mid-water trawls since 2007 which coincides with a reduction in effort in the fishery".

However, these limited bycatch mortality records from mid-water trawls in the SPF understate the nature and extent of interactions with cetaceans, as they do not include the full range of direct interactions that may have occurred (or could occur in future) between cetaceans and the mid-water trawl sector of the SPF. Tuck *et al*. (2013) reported on bycatch interactions for some cetacean species but other cetacean species are known to occur in this SPF area and may have been observed during fishing operations. As noted in section 2.2.2 the definition of 'direct interactions' with protected species used by the panel for this assessment, and which are directly relevant to assessing the nature and extent of interactions with cetaceans, includes any interactions with fishing operations or gear (including net feeding); any physical contact (including collisions); bycatch which can result in injury or mortality; acoustic disturbance from fishing operations; and any behavioural changes in these species brought about by habituation to fishing operations. Therefore, for the purposes of this assessment all of these direct interactions are discussed below.

Interactions with fishing operations or gear including net feeding

The frequency and risks of interactions between cetaceans and fishing operations in the SPF are uncertain, but cetaceans that prey on small pelagic fish such as common and bottlenose dolphins are more likely to interact with the mid-water trawl sector of the SPF. Interactions with fishing operations and gear are likely to mostly occur underwater and will be largely undetected unless some form of underwater monitoring and reviewable recordings are made. Feeding within and near trawl nets has been recorded for a range of cetacean species including short-beaked common dolphins and bottlenose dolphins in trawl fisheries in Australian waters and internationally (e.g. Corkeron *et al*. 1990, Broadhurst 1998, Jaiteh *et al*. 2013, Elgin Associates unpublished (a)), which increases the risk of bycatch. Common bottlenose dolphins are the cetaceans most often documented feeding in association with trawlers worldwide (Fertl and Leatherwood 1997, Broadhurst 1998). The mortality of dolphins recorded in mid-water trawls in the SPF probably resulted from these dolphins feeding in association with the trawl operations and may have occurred from feeding within the trawl nets, but the exact nature of the interactions leading up to the death of these dolphins is unknown. Underwater recording of numerous mid-water trawls in the SPF in 2005 did not coincide with any recorded dolphin interactions (Lyle and Willcox 2008) therefore dolphin behaviour and feeding positions during interactions with mid-water trawls in the SPF are unknown. Lyle and Willcox (2008) concluded that cetacean interactions with fishing activities in the SPF are relatively uncommon and unpredictable.

Four types of cetacean feeding patterns in association with trawlers have been reported. The majority of cetaceans that interact with trawlers are reported to forage behind trawl nets, cetaceans may enter trawl nets to feed, some cetaceans feed on discards or on fish that escape or fall from the net, and some cetaceans feed on prey that are attracted to fishing vessels (Corkeron *et al*. 1990, Fertl and Leatherwood 1997, Broadhurst 1998, Northridge *et al*. 2005, Jaiteh *et al*. 2013).

Physical contact including collisions

Direct interactions include physical contacts and collisions with cetaceans, but the risk of trawlers in the SPF colliding with and injuring cetaceans is uncertain. Another area of uncertainty arises from the extent of injuries resulting from collisions, as collisions can result in deep trauma to tissues and organs that may not be obvious externally (Laist *et al*. 2001, Moore *et al*. 2013). Many collisions of vessels with cetaceans go undetected, and this is more likely for larger vessels and ships such as the type proposed for use in the DCFA. Most severe or fatal injuries to whales are caused by ships that are 80 m or longer and involve vessels travelling 14–15 kilometres per hour or faster, where whales are usually not seen or are sometimes seen too late to be avoided (Laist *et al*. 2001, Vanderlaan and Taggart 2007). Smaller cetacean species are considered to be more at risk of vessel strike from small, fast vessels (Silber *et al*. 2009, Redfern *et al*. 2013).

Eleven whale species are known to be hit by ships, and of these, fin whales are struck most frequently, with right whales, humpback whales and sperm whales also commonly hit (Laist *et al*. 2001, Vanderlaan and Taggart 2007, Redfern *et al*. 2013). Ship strikes are a significant threat to small, depleted whale populations such as Northern Hemisphere right whales, and to fin and sperm whales in the Mediterranean Sea and Black Sea (Vanderlaan and Taggart 2007, Notarbartolo di Sciari and Birkun 2010). Populations of humpback whales, southern right whales and some other great whale species are increasing in Australian waters (reviewed in Bannister 2008, Woinarski *et al*. 2014) and many of these whales migrate into or through southern Australian waters in the SPF area, hence the incidence of vessel strike in this region is likely to increase in future. Vessel strikes are thought to be relatively common in Australian waters but are not well documented (Bannister *et al*. 1996, Ross 2006). Southern right whale mortalities from vessel collisions have been recorded in southern Australian waters (Kemper *et al*. 2008), and collisions with vessels and entanglement in fishing gear are regarded as the current main direct threats to humpback whales (Fleming and Jackson 2011). Blue whales are also reported to be injured and in some cases killed by collisions from ships in Australian waters, and surface feeding on krill swarms in upwelling areas such as the Bonney Upwelling increases the risk of vessel strike for these and other whale species that occur in this region (Gill 2002, Miller *et al*. 2012).

Bycatch injury or mortality

Entanglement, injury and fisheries bycatch mortality is the major threat to many smaller cetacean species in Australian waters and internationally, particularly from purse seine, gillnet and trawl fishing, and from discarded fisheries gear (reviewed in Shaughnessy *et al*. 2003, Zollett and Rosenberg 2005, Read *et al*. 2006, Zollett 2009, Reeves *et al*. 2013, Anderson 2014). Cetacean bycatch rates have been substantially reduced in some fisheries in recent decades, but there is potential for increased frequency and intensity of interactions and bycatch mortality as human populations and fisheries operations increase (e.g. Hall *et al*. 2000, Read *et al*. 2006, Stephenson *et al*. 2008, Allen *et al*. 2014). Globally, gillnets are a major threat to cetaceans with 75 per cent of odontocete species and 64 per cent of mysticete species plus many other groups of marine mammals recorded as bycatch in gillnets in the past two decades (reviewed in Reeves *et al*. 2013, Geijer and Read 2013). Longline fisheries are also a major threat to many odontocetes with 20 species recorded as bycatch from 1964 to 2010 (Hamer *et al*. 2012). Purse seine fishing has been the major cause of dolphin bycatch internationally (e.g. Gerrodette and Forcada 2005), and significant bycatch of short-beaked common dolphins has been recorded in the SASF (Hamer *et al*. 2008).

Globally, 25 cetacean species (23 odontocete and two mysticete) species have been recorded as bycatch mortality in working trawls or in discarded trawling gear (reviewed in Fertl and Leatherwood 1997, Zollett and Rosenberg 2005). Bycatch has been recorded in nearly all areas where trawling occurs including in waters around Australia and off New Zealand (Fertl and Leatherwood 1997). The risk of bycatch varies among cetacean species depending upon a range of factors including whether the cetaceans target prey species in feeding grounds that are also used by fisheries, the types of prey species and fisheries activities involved, and intersection of fishing zones with migratory pathways or habitats regularly used by cetaceans (Couperus 1997). Environmental and operational activities are important factors influencing bycatch including seasonal changes in prey availability, habitat and proximity to the continental shelf edge, vessel and net size, trawl tow speed and duration, trawl depth, diurnal trawling patterns, and whether single vessel or pair trawling is used (Couperus 1997, Zollett and Rosenberg 2005, Zeeberg *et al*. 2006, Fernández-Contreras *et al*. 2010, Elgin Associates unpublished (a)). The species of cetaceans present and their abundance and behaviour are also important, with different age classes and sexes likely to interact in different ways with trawls, with higher mortality of juveniles reported in some trawls indicating that inexperience may increase the risk of bycatch (e.g. Fertl and Leatherwood 1997, Chilvers and Corkeron 2001).

In New Zealand waters the primary threat to the endemic Hector's dolphin *Cephalorhynchus hectori* is from bycatch mortality in gillnet and trawl fisheries (Slooten *et al*. 2006, Slooten 2007, Slooten and Dawson 2010, Slooten 2013). Mid-water and bottom trawling for jack mackerel also results in bycatch of common dolphins in New Zealand waters and both trawl effort and dolphin captures have increased in recent decades (Thompson and Abraham 2009, Thompson *et al*. 2013). The jack mackerel trawl fishery off the west coast of the North Island was responsible for 91 per cent of observed dolphin mortalities in trawl fisheries from 1995 to 2007, and headline depth was the variable that explained most of the dolphin bycatch (Thompson and Abraham 2009). Other explanatory variables were trawl duration, light conditions, diurnal patterns and geographic location (Thompson and Abraham 2009).

In the SPF the 25 records of bycatch mortality of dolphins in mid-water trawls during 2004–05 are indicative of the bycatch problem (Lyle and Willcox 2008, Tuck *et al*. 2013), but do not provide sufficient information to determine the likely full extent of injury and bycatch mortality. This uncertainty arises from various related issues. Cetaceans that are injured from interactions with mid-water trawls but escape or are not caught in nets are largely undetected, and in some cases injuries may impair health status leading to subsequent unrecorded mortality. Where lethal interactions occur, some dead cetaceans may drop out of nets (particularly in bottom opening excluder devices e.g. Allen *et al*. 2014), which represents another form of undetected cryptic mortality. Where dead cetaceans are observed in nets, identity of the species may not be recorded or may be uncertain (as in most of the dolphin bycatch records from mid-water trawls in the SPF, see Lyle and Willcox 2008, Tuck *et al*. 2013). In some cases, cetacean mortality may be under-reported by fishers resulting in lower estimates of bycatch mortality from logbooks compared with independent observer records (e.g. Stephenson *et al*. 2008, Allen *et al*. 2014).

Of relevance to the DCFA, cetaceans are more often caught in mid-water trawls than in bottom trawls (Crespo *et al*. 1997, Fertl and Leatherwood 1997, Hall *et al*. 2000), and this may occur because:

- small pelagic fish species are important prey items for some groups of cetaceans and mid-water trawls are used to target these fish
- mid-water trawl gear is generally towed at relatively high speeds
- mid-water trawl nets are generally much larger than most demersal trawls
- mid-water trawl nets often operate for extended periods within the normal diving depth of cetaceans, hence where the trawl time exceeds the breath-holding capacity, individuals caught in the net drown (Zollett and Rosenberg 2005, Elgin Associates unpublished (b)).

Pair trawling is a relatively high-risk trawling technique and accounts for about half of the cetacean bycatch in waters off New Zealand (Fertl and Leatherwood 1997, Thompson *et al*. 2013). Pair trawlers tend to tow nets faster than single trawlers, and the nets have higher headlines and greater overall dimensions (Fertl and Leatherwood 1997). The introduction of large freezer/factory industrial fishing vessels and other improvements in fishing technology have enabled the expansion of trawl fisheries (Crespo *et al*. 1997, Zeeberg *et al*. 2006). These vessels fish with larger gear, for longer and often farther offshore, which increases the likelihood of interactions with cetaceans (Crespo *et al*. 1997, Reeves *et al*. 2003, Zollett and Rosenberg 2005). Nets with a larger circumference have a larger net opening and the greater extension of their bridles and doors may cause a significant herding effect for cetaceans and other large marine predators (Zeeberg *et al*. 2006).

Cetacean mortality in trawl nets can occur when nets are shot, during trawling, or when the vessel stops hauling and the trawl entrance collapses (haulback) trapping animals (Fertl and Leatherwood 1997). Long haul times also increase the risk of cetacean bycatch mortality in trawl nets (Du Fresne *et al*. 2007). Dolphins can get their rostrum caught in the net while attempting to extract fish, they can drown when their tail stock is caught in the hanging line of the trawl and have also been caught in turtle exclusion and cetacean excluder devices (Fertl and Leatherwood 1997, Stephenson *et al*. 2008). Where fish pumps are used to empty the catch from the net, cetacean bycatch is often not observed because the ability of the observer to record marine mammal catches is compromised (Morizur *et al*. 1999, Zollett and Rosenberg 2005), particularly where the final emptying of the codend occurs at night (Ross and Isaac 2004).

Discarded trawl nets contribute significantly to marine debris and cetaceans and other marine animals are caught in discarded nets resulting in 'ghost netting' (Fertl and Leatherwood 1997, Reeves *et al*. 2003). Trawl netting may also be ingested by some cetaceans (Fertl and Leatherwood 1997). Fishers often use a technique termed 'cutting out' of living entangled animals from fishing nets, resulting in these animals being released while still entangled. Mortalities occur from drowning after release or from a prolonged demise resulting from impaired foraging, increased drag, emaciation, infection, haemorrhage and severe tissue damage leading to death (Cassoff *et al*. 2011, Moore *et al*. 2013). Larger whales entangled in fixed trap and net gear can undergo a very slow demise, averaging six months in entangled North Atlantic right whales *Eubalaena glacialis*, but sometimes extending over several years (Moore and van der Hoop 2012). Similarly, entanglement in fishing gear is known to cause chronic injury, debilitation and death of southern right whales in Australian waters (Kemper *et al*. 2008). In the North Atlantic region between 1980 and 2004, aerial surveys detected that at least 73 per cent of 493 large whales sighted were currently entangled or had been entangled in fishing gear at least once previously (Moore and van der Hoop 2012). In humpback whale populations in the Northern Hemisphere, at least 50 per cent or more of the identified animals have scarring indicating previous entanglement in fisheries gear (Robbins and Mattila 2004, Johnson *et al*. 2005, Fleming and Jackson 2011). In Australian waters, cetacean species recorded entangled in marine debris include bottlenose dolphins, common dolphins, humpback whales and southern right whales (Shaughnessy *et al*. 2003, Kemper *et al*. 2008).

Acoustic disturbance

Acoustic disturbance from anthropogenic activities can be particularly important for cetaceans because their acoustic sense is very highly developed and therefore sounds are vitally important to their ecology and survival (McCauley and Cato 2003, Jefferson *et al*. 2008, Jensen *et al*. 2009). Sound-induced effects vary from no discernible effect; adverse effects on prey; masking of signals; various behavioural responses; temporary threshold shifts in hearing ability; permanent threshold shifts or, in extreme cases, direct damage to hearing or other organs (McCauley and Cato 2003, Nowacek *et al*. 2007, Zirbel *et al*. 2009). Heavy vessel traffic, seismic testing, drilling and pile driving, dredging and naval sonar can lead to increased underwater noise disturbance and can reduce habitat quality for cetaceans, particularly in areas important for feeding, breeding, calving or resting (Nowacek *et al*. 2007, Zirbel *et al*. 2009). The effect of most anthropogenic noise on cetaceans is uncertain or unknown (McCauley and Cato 2003), hence the effects of trawler and other fishing activities on cetaceans in the vicinity is difficult to determine (Reeves *et al*. 2003). Trawler operations are inherently noisy and the acoustic disturbance may affect cetacean behaviour. Acoustic 'pingers' are used as deterrents to reduce marine mammal bycatch in fishing nets in gillnet and some other fisheries (e.g. Carretta and Barlow 2011, Dawson *et al*. 2013). The use of pingers introduces another form of noise, and may induce altered behaviour and reduce bycatch of some cetacean species in some types of nets such as gillnets, but results are inconsistent for other cetacean species and when used in relatively noisy trawling operations (Carretta and Barlow 2011, Dawson *et al*. 2013, Allen *et al*. 2014).

Behavioural changes

Other forms of behavioural change from fisheries interactions can be important. For example, some individuals or groups of bottlenose dolphins frequently interact with trawlers in Australian waters and this alters their feeding ecology and increases the risk of bycatch and potential risk of predation by sharks or killer whales (e.g. Corkeron *et al*. 1990, Broadhurst 1998, Jaiteh *et al*. 2013, Allen *et al*. 2014). Trawl fisheries may provide a more reliable source of food from bycatch disposal and catch depredation in an otherwise patchy environment for food resources, and this altered food availability and predictability can affect social interactions and population demographics leading to habitual interactions with trawlers by some groups of dolphins (Corkeron *et al*. 1990, Chilvers and Corkeron 2001). Interactions with trawlers may increase at night (Fertl and Leatherwood 1997, Crespo *et al*. 1997), and bottlenose dolphins have been seen to exploit fish attracted to illumination of surface waters from deck lights on trawler vessels (Zollett and Rosenberg 2005). Up to 30–40 killer whales have been reported interacting with Dutch mid-water trawl freezer vessels off the Shetland Islands, and scavenged off discards or fed on fish that slipped through the net or slipped overboard during hauling or shooting of the net (Couperus 1994). Subtle changes in behaviour and potential for habituation can be difficult to detect and require detailed long-term monitoring of behaviour and ecology, which is beyond the capability of most fisheries observer programs.

Summary: nature and extent of interactions of mid-water trawl gear with cetaceans

- *The SPF area encompasses the known distribution range of most cetacean species occurring in Australian waters; this area is known to be important to many cetacean species and interactions with mid-water trawl and other fisheries have occurred for many species.*
- *A total of 25 dolphin mortalities were reported in mid-water trawls in the SPF during 2004 and 2005, comprising of some common bottlenose dolphins and possibly short-beaked common dolphins. The absence of reported interactions with cetaceans in this fishery in more recent years coincides with low levels of fishing and observer effort. Therefore, it is difficult to estimate the overall extent of direct interactions with cetaceans by mid-water trawl gear in the SPF.*
- *The nature and likelihood of interactions between cetaceans and mid-water trawl fisheries varies substantially among species. Bottlenose dolphins and short-beaked common dolphins are likely to be at higher risk based on reported interactions with trawls and bycatch in Australia and internationally.*
- *Direct interactions with fishing operations include net feeding, foraging behind trawlers, and feeding on discards and fish escaping from nets. Vessel collisions resulting in injury or death of whales and some other cetaceans are thought to be relatively common in Australian waters but are not well documented. Most severe or fatal injuries to whales are caused by collisions from vessels greater than 80 m.*
- *Fisheries bycatch mortality is the major threat to many smaller cetacean species in Australian waters and internationally. Cetacean bycatch occurs in most areas where trawling occurs and they are more often caught in mid-water trawls than in bottom trawls. The risk of bycatch increases where prey species are also targeted by fisheries and where fishing grounds overlap with important habitats used by cetaceans for aggregating, feeding, breeding and as migratory routes.*
- *Analyses of common dolphin bycatch in New Zealand mid-water trawl fisheries showed that bycatch occurred in vessels longer than 90 m, and bycatch was highest in trawls where the headline depth was between 10–40 m, and during longer tows of two to six hours in duration. Light conditions and fishing location also significantly influenced common dolphin bycatch rates. Sharp vessel turns and changes in speed may increase the risk of bycatch.*
- *Cetaceans that frequently interact with trawlers and other fisheries can become habituated, leading to altered social interactions and increased risk of bycatch.*
- *Acoustic disturbance can be important for cetaceans because they have a very highly developed acoustic sense and sounds are vitally important for their ecology and survival.*

5.3.3 Management

Management of interactions between cetaceans and fishing operations

Management actions and mitigation measures to reduce cetacean bycatch can include modification to fishing gear including the use of excluder devices, modification to fishing practices including offal management, temporal and spatial closures, and fisheries bycatch triggers and move-on rules. These management measures are reviewed below, and then discussed in relation to the DCFA.

Cetacean excluder devices (CEDs)

Various types of excluder devices have been developed and used in trawl fisheries to reduce bycatch of cetaceans and other marine megafauna (e.g. Northridge *et al.* 2005, Stephenson *et al.* 2008, Elgin Associates (unpublished (b)). The design and function of excluder devices to facilitate the escape of large marine animals that enter trawl nets are reviewed in Elgin Associates (unpublished (b)), and are outlined in Section 5.2.3.

Excluder devices need to be carefully designed for each fishery, to take into account a range of variables including the size and behaviour of the species to be excluded from the fishing gear, characteristics of each gear type including size and operation, fishing operations, towing speed, the hydrodynamics of trawl set up in relation to trawl size/grid and escape hole ratios, storage of trawl nets on the vessels, and the size of target and non-target species (Zollett and Rosenberg 2005, Elgin Associates (unpublished (b)). Hence, an excluder device that proves to be effective in reducing bycatch mortality in one fishery while maintaining catch per unit effort of target species may not be effective in another fishery that encounters different marine mammals and is targeting different fish species.

Appropriately designed excluder devices have been shown to be effective in reducing bycatch of some pinnipeds (refer to Section 5.2.3), but there are no studies that indicate excluder designs tested to date are consistently effective in reducing cetacean bycatch in trawls (reviewed in Elgin Associates unpublished (b)). This may be because dolphins, for example, are less manoeuvrable within the trawl net than are fur seals and sea lions. Underwater cameras monitoring the effectiveness of excluder devices have shown that some dolphins appear distressed when they are near excluder grids and seem reluctant or unable to enter a narrow and confined release route and instead tend to swim upstream out of the mouth of the net (e.g. Zeeberg *et al*. 2006). Elgin Associates (unpublished (b)) concluded that further information is required on the escape behaviour of dolphin species known to interact with trawl nets, and at present there is no solution to filter or deter cetaceans from the net opening. Some studies indicate that modified CEDs with top-opening escape hatches may be the most effective way of further reducing cetacean bycatch because some dolphins have been observed to seek an exit in the upper part of the trawl (e.g. Northridge *et al*. 2005, Allen *et al*. 2014). However, Zollett and Rosenberg (2005) reported that three female bottlenose dolphins preferred to exit at the bottom of a trawl net tested during experiments in a captive facility.

As noted in Section 5.2.3, from the commencement of mid-water trawls in the SPF, the nets were fitted with a 'soft' rope-mesh SED (Browne *et al*. 2005). Following dolphin bycatch in mid-water trawls, Lyle and Willcox (2008) examined three SED designs and were able to evaluate interactions with seals. However, no cetacean interactions were recorded in the 98 tows used to assess SED performance from underwater video footage; hence their effectiveness for mitigating dolphin bycatch is unknown. That study identified that an upward-opening SED should be trialled for the mid-water trawl fishery in the SPF to examine the effect in mitigating dolphin and seal mortalities, but this has not been done due to lack of funding and the recent minimal trawl effort in the fishery (AFMA 2011, Tuck *et al*. 2013). Seafish Tasmania Pty Ltd commissioned the design of a soft-grid SED (see Section 5.2.5) for use on its proposed large mid-water trawl freezer vessel in the SPF, but this has not been tested during trials at sea. Until the behaviour of the cetacean species most likely to interact with mid-water trawls in the SPF is better understood, it will be difficult to design an excluder device that effectively mitigates both pinniped and cetacean bycatch in this fishery. It is also possible that common dolphins and bottlenose dolphins and other cetacean species in the SPF area may react differently to the stress of being constrained within trawl nets and may require different excluder designs, which would further complicate bycatch mitigation planning.

Three fisheries case studies where excluder devices were assessed in relation to cetacean behaviour and bycatch in trawl fisheries were reviewed by Elgin Associates (unpublished (b)), and are summarised below.

<u>European (Dutch and Irish) pelagic fleet fishing off Mauritania, northwest Africa</u>

Zeeberg *et al*. (2006) noted that between 40 and 70 foreign trawlers (Russian, Lithuanian, and Icelandic) including 5–10 European (Dutch and Irish) pelagic freezer/factory trawlers (with net openings of around 90 by 50 m) operated in this fishery and are among the largest fishing vessels in the world. Significant bycatch of marine megafauna occurs in this fishery, and cetaceans comprised 8 per cent of the megafauna bycatch recorded by observers with 70–720 dolphins captured between 2001 and 2005, with the main bycatch species being common dolphins (Zeeberg *et al*. 2006). Heessen *et al*. (2007) noted that observations by trawler crew are likely to underestimate the extent of megafauna bycatch. Cetacean bycatch occurred almost exclusively at night, and there was a strong seasonal relationship with cetacean bycatch associated with the return of migrating sardines (Zeeberg *et al*. 2006). Pods of 10–20 short-finned pilot whales or groups of 5–30 dolphins were captured by trawl operations in spring.

The large animal excluder device used in this fishery was not designed specifically to reduce dolphin bycatch, but to mitigate bycatch of all megafauna including sharks, manta rays, sea turtles, and dolphins (Zeeberg *et al*. 2006, Heessen *et al*. 2007). Captured megafauna are retained by a part of the net consisting of a large mesh filter 'shark-grid' that allows smaller fish to pass, but prevents larger animals from entering the codend. The grid is designed to guide pelagic megafauna to an escape route along the bottom of the trawl (Figure 5.16). As some dolphins had been observed to seek an exit in the upper part of the net, a cetacean exit was built in ahead of the grid (Figure 5.16) to enable cetaceans to accelerate upwards to reach the water surface (Zeeberg *et al.* 2006). Usually the captured megafauna are discarded into the sea while the codend is still in the water but before the fish pumping starts to prevent megafauna blocking the fish pump (Heessen *et al.* 2007). Zeeberg *et al.* (2006) noted that several types of cetacean 'barriers' consisting of vertical ropes in the front part of the trawl and acoustic deterrents were under development to prevent dolphins from entering the net opening or guide them out during hauling, but no details are available on their efficacy in reducing dolphin bycatch (Elgin Associates unpublished (b)).

Zeeberg *et al.* (2006) tested the tunnel exclusion on Dutch mid-water trawl freezer vessels. The vessels alternately fished with and without the excluder, and use of the excluder did not significantly influence the catches of the target species (Heessen *et al.* 2007). A 40–100 per cent reduction in bycatch of the megafauna species most vulnerable to bycatch was recorded. However, although cetaceans made up only 8 per cent of the retained bycatch, none were released alive. These bycatch results show that further research is needed to reduce cetacean bycatch mortality using excluder devices in that fishery (Elgin Associates unpublished (b)).

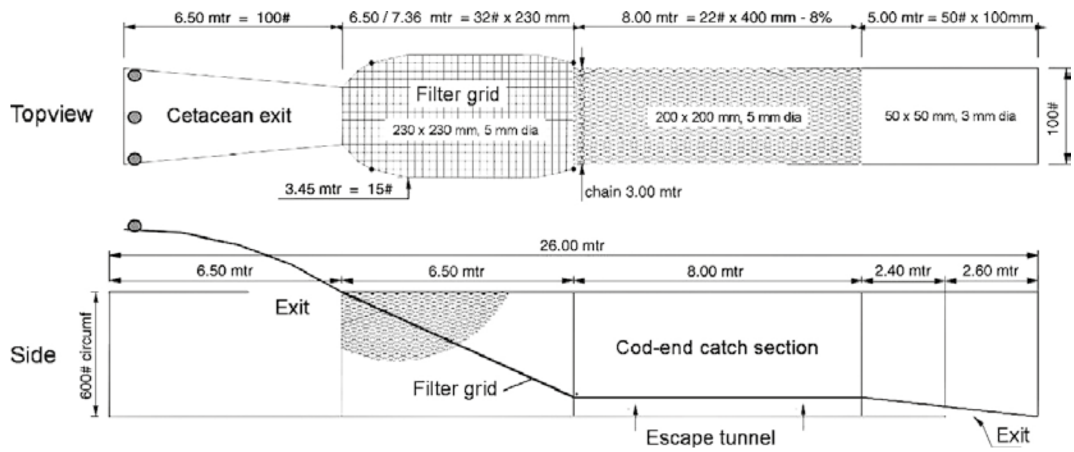

Figure 5.16 Diagram of the aft section of a mid-water trawl (about 50–70 m in front of the codend), showing the position of the cetacean exit ahead of the filter grid and connection to the escape tunnel. The filter grid slopes top-downwards with about a 20° inclination that forces larger non-target species downward to the tunnel entrance. **Source:** Reprinted from *Fisheries Research* 78, 2-3, J. Zeeberg, A.Corten and E. de Graaf. Bycatch and release of pelagic megafauna in industrial trawler fisheries off Northwest Africa. pp. 186–195, Copyright (2006) with permission from Elsevier B.V.

Western Australian Pilbara Fish Trawl Interim Managed Fishery

The PFTIMF is an otter trawl fishery targeting demersal scalefish species. In 2002, bycatch data were obtained from 427 trawl shots representing 1581 hours of trawling and an observer coverage rate of 7.7 per cent. Common bottlenose dolphins were observed around and in almost every trawl shot, with four incidental dolphin deaths reported (Stephenson and Chidlow 2003). Allen *et al*. (2014) subsequently compared data from skippers' logbooks and independent observers to assess trends in common bottlenose dolphin bycatch patterns between 2003 and 2009. Dolphins were caught in all fishery areas, across all depths and throughout the year. Bycatch rates reported by independent observers (n = 52 dolphins in 4124 trawls, or 12.6 dolphins per 1000 trawls) were approximately double those reported by skippers (n = 180 dolphins in 27,904 trawls, or 6.5 dolphins per 1000 trawls).

The effectiveness of exclusion grids and escape hatches fitted to trawl nets in the PFTIMF to reduce dolphin interactions was assessed by Stephenson *et al*. (2008) in conjunction with an assessment of acoustic pingers (Stephenson and Wells 2006). During this research, dolphins were recorded entering trawl nets to forage in more than 98 per cent of trawls and purposely made contact with the fishing gear including clinging to the headrope and bouncing along the net (Stephenson *et al*. 2008). Similar high rates of interactions and behaviour were observed during subsequent research on modified net designs (Allen and Loneragan 2010, Jaiteh *et al*. 2013). In more recent observer programs, common bottlenose dolphins were the only species observed to deliberately enter trawl nets and were recorded feeding on captured fish in more than 75 per cent of trawls (Wakefield *et al*. 2014). Seven dolphins were observed within close proximity to exclusion gear inside trawl nets (Wakefield *et al*. 2014).

BRDs have been mandatory in the PFTIMF since 2006 and initially consisted of a semi-flexible metal grid and a bottom-opening escape hatch with a loose skirt of netting covering the hatch to prevent the loss of target species (Stephenson *et al*. 2008, Allen *et al*. 2014). In 2008, the BRDs were moved forward in the net from a position just before the codend to the beginning of the net extension to prevent dolphins from backing down into the extension, thereby providing a shorter escape route between the BRDs and the opening of the net (Allen *et al*. 2014). More recently, different excluder devices have been trialled with top or bottom opening net configurations. These devices were not designed specifically to mitigate dolphin bycatch, but to reduce the bycatch of all marine megafauna including turtles, seasnakes, sawfish, rays and sharks.

Use of a semi-flexible exclusion grid constructed from braided stainless wire and pipe (Figures 5.17, 5.18) appeared to reduce the bycatch of dolphins by almost half (Stephenson *et al*. 2008). Allen *et al*. (2014) categorised dolphin bycatch data on three broad net configurations as follows.

1. Prior to the introduction of the BRDs (August 2003 until February 2006; excluding BRD trials) bycatch was 8.9 dolphins per 1000 trawls (skipper's logbook) and 18.8 dolphins per 1000 trawls (independent observer data).

2. In BRD trials from the previous period, after the compulsory introduction of the BRDs and before BRDs were moved forward (primarily March 2006 to May 2008), bycatch was 5.2 dolphins per 1000 trawls (skipper's logbook) and 10.3 dolphins per 1000 trawls (independent observer data).

3. After the BRDs were moved forward in the net (June 2008 until September 2009) bycatch was 3.9 dolphins per 1000 trawls (skipper's logbook) and 11.3 dolphins per 1000 trawls (independent observer data).

Stephenson *et al*. (2008) tested the semi-flexible cetacean exclusion device, and underwater video footage was obtained for 446 shots. Most dolphins backed down into the net to about 3 m from the grid and then swam upstream out of the net. Seven dolphins were recorded interacting with the grid or escape opening. Three dolphins were assumed to have escaped alive and four were distressed and were assumed to have died (Stephenson *et al*. 2008). Two dolphins had their tail fluke caught in the grid, so it was suggested that the bar spacing should be reduced to less than 155 mm, to reduce the likelihood of this occurring (Stephenson *et al*. 2008). Dolphins were generally caught during daylight. Net depth (50–80 m) did not affect the capture rate of dolphins (Stephenson *et al*. 2008).

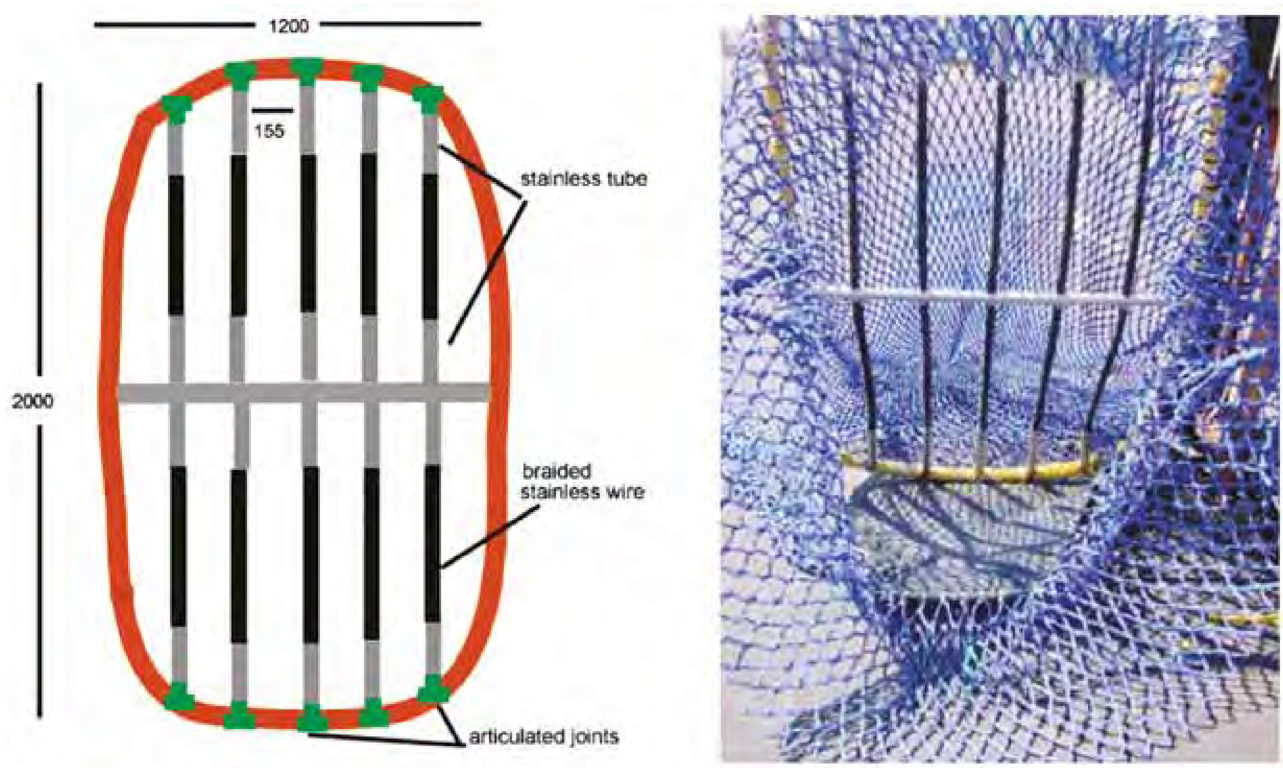

Figure 5.17 Semi-flexible grid constructed from stainless tube and braided stainless wire.

Source: Stephenson *et al.* (2008), reproduced with permission from the Department of Fisheries, Government of Western Australia.

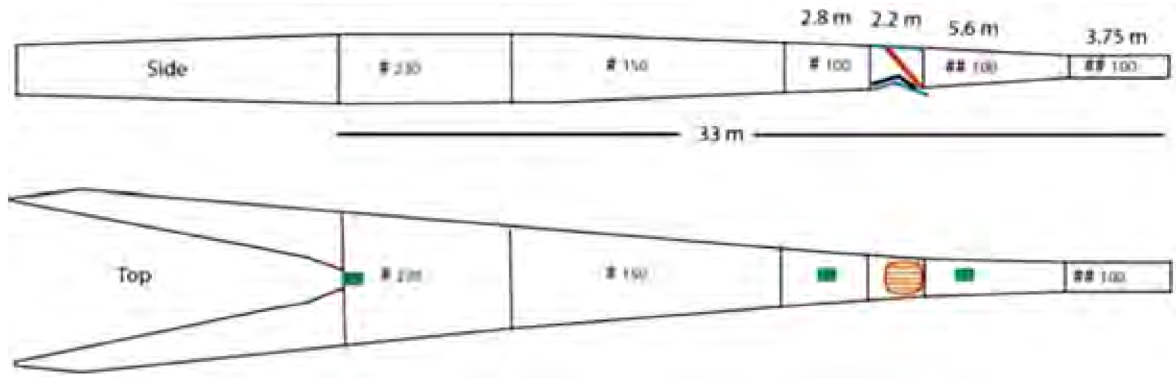

Figure 5.18 Net design used during the selection grid trials showing the grid (red), cover net at the bottom opening escape (blue), the Kevlar flap (black), and the location of the cameras (green).

Source: Stephenson *et al.* (2008), reproduced with permission from the Department of Fisheries, Government of Western Australia.

In 2012, three exclusion gear configurations in trawl nets were evaluated in trials conducted on the three vessels fishing in the PFTIMF (Wakefield *et al*. 2014). The 'downward excluding net' configuration included a semi-rigid downward-angled exclusion grid (six stainless steel tubes spaced at 150 mm apart with a side tube length of 795 mm) with an escape hatch cut into the bottom of the trawl net forward of the grid and a mesh cover opening backward to facilitate the expulsion of megafauna and benthos during trawling (Figure 5.19a).

The 'upward excluding net' had an upwardly inclined grid and the escape hatch and mesh cover were moved to the top of the net immediately forward of the grid (Figure 5.19b). The grid was rigid and the spacing of the stainless steel tubes was increased to 200 mm with the length of the side bars increased to 1030 mm (Wakefield *et al*. 2014).

The second modified 'experimental net' used the same rigid grid as the upward excluding net, but it was orientated downward (Figure 5.19c). The escape hatch was cut into the bottom of the net forward of the grid with a similar mesh cover opening backwards, but the grid and escape hatch were stitched into 50 mm square mesh to keep this section of the net cylindrical, to improve water flow through the net (Figure 5.19c). An additional 3 m longitudinal escape slit was cut into the top of the square mesh net forward of the exclusion grid, to facilitate the escape of predominantly air-breathing animals, based on the assumption that they would tend to push upwards to escape (Allen and Loneragan 2010). The escape slit was held together with magnets along its edges to keep it closed during trawling and after an animal had passed through it (Wakefield *et al*. 2014).

The effectiveness and efficiency of these three different exclusion gear configurations in mitigating dolphin and other megafauna species interactions were assessed. All trawl vessels in the PFTIMF were fitted with above-water and subsurface within-net camera systems, and observer coverage during the trials was high (Wakefield *et al*. 2014). Despite more than 75 per cent of trawls having high levels of interaction around and within trawl nets by common bottlenose dolphins, captures of megafauna were rare (Wakefield *et al*. 2014).

Ten dolphin mortalities were recorded during the trials and another seven common bottlenose dolphins were observed underwater in close proximity to exclusion gear inside the trawl nets during five trawls (Wakefield *et al*. 2014). All seven of these dolphins appeared to be distressed and exhibited short and infrequent bursts of swimming towards the mouth of the net, but did not always move upwards toward the top of the net. Four of the seven dolphins asphyxiated and died and were retained within the net ahead of the exclusion grid. Two of the other three dolphins exited from the upward excluding net through the top opening escape hatch within 0.3–5.0 minutes, and were considered to have a high chance of survival. Interestingly, the dolphin that exited the net in the shortest time approached the exclusion grid head first and exited through the escape hatch head first, whereas the other six interactions involved dolphins approaching the grid tail first and this usually led to the tail passing through the grid and becoming lodged. The other dolphin appeared to asphyxiate and was retained within the net forward of the grid until the trawl was near the surface during hauling and under excessive turbulence, causing the tail to become dislodged from the exclusion grid, and the dolphin fell out of the net through the top opening escape hatch that was now orientated downward. This was the only observation of an asphyxiated dolphin passing through an escape hatch. No megafauna or scalefish were observed to exit the trawl nets through the top opening escape slit in the 'experimental net', but one dolphin was observed attempting to enter the trawl net through this escape slit (Wakefield *et al*. 2014).

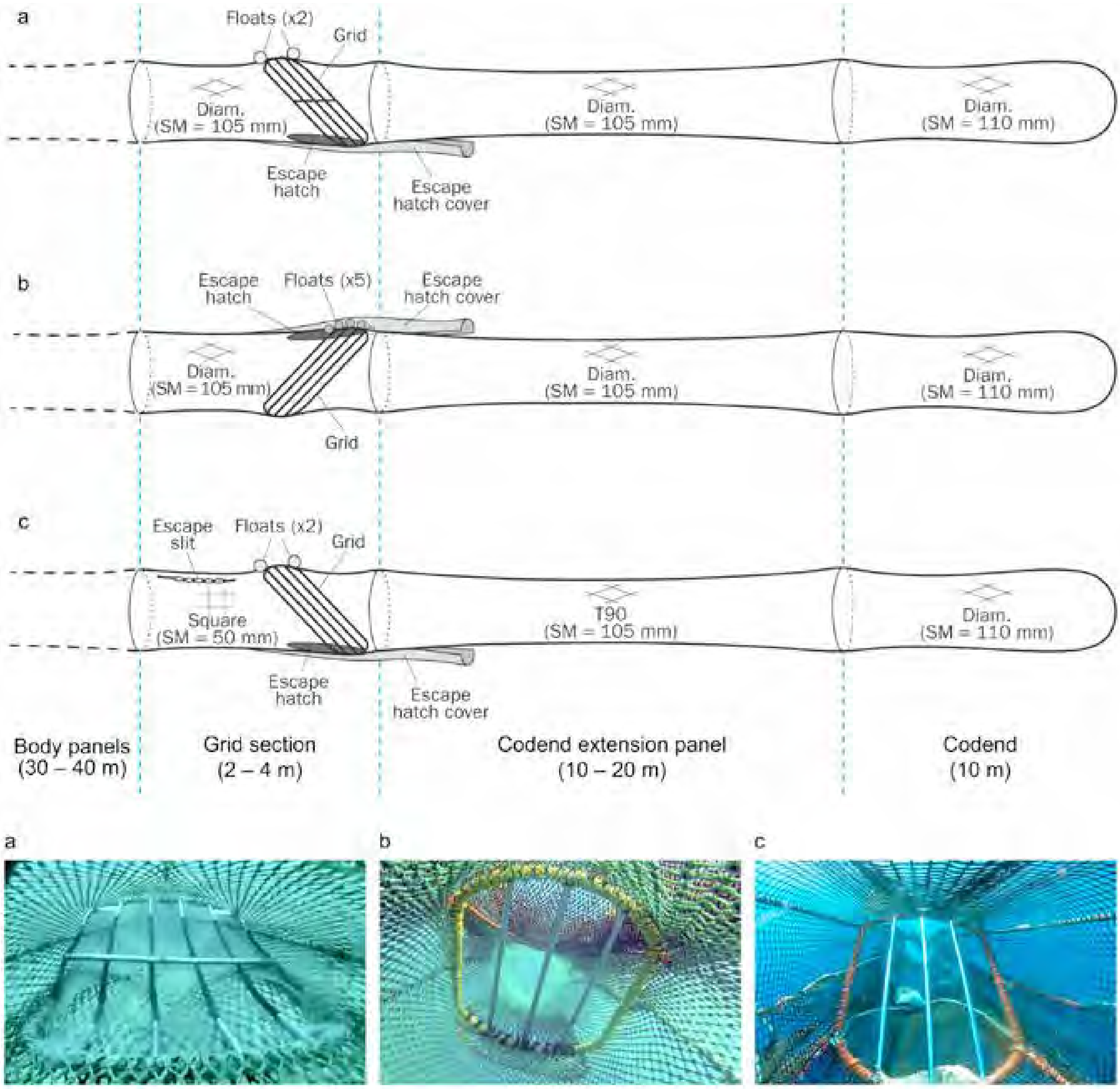

Figure 5.19 Diagrams and *in situ* images taken from the net camera systems with the camera positioned behind the grid facing forward for the three different net configurations (a) downward excluding net, (b) upward excluding net and (c) experimental stretched mesh net. **Source:** Wakefield *et al.* (2014), reproduced with permission of the Department of Fisheries, Government of Western Australia.

UK Bass Pair Trawl Fishery and adjacent European fishery

The European sea bass *Dicentrarchus labrax* commercial fishery is located in the western English Channel and Bay of Biscay, where mid-water pair trawlers target these shoaling fish offshore prior to spawning (Northridge *et al.* 2011). The UK pelagic pair trawl fishery is usually operated by two pairs of Scottish 30–40 m trawlers with trawl nets towed near the surface, with up to 50 pairs of French boats operating the same gear mostly in the Bay of Biscay area (Northridge 2007). Cetacean bycatch rates are very high, with mean bycatch rates of about one short-beaked common dolphin per tow (Table 5.3, Northridge *et al.* 2011).

Table 5.3 Common dolphin bycatch in the UK bass pair trawl fishery

WINTER SEASON	POINT ESTIMATE OR CENSUS	LOWER CONFIDENCE LEVEL	UPPER CONFIDENCE LEVEL
2000–01	190	172	265
2001–02	38	23	84
2002–03	115	88	202
2003–04	439	379	512
2004–05	139	139	146
2005–06	84	84	85
2006–07	70	55	117
2007–08	0	0	0
2008–09	2	2	2
2009–10	28	28	28

Source: Northridge *et al.* (2011).

Necropsies of stranded animals showed that bycatch, most probably from pelagic fishing operations, was the cause of death in 65 per cent of stranded common dolphins that were assessed and where cause of death was established (Department for Environment, Food and Rural Affairs 2003). Data from 2004–09 showed that the overlap between pelagic fisheries and a short-beaked common dolphin 'hotspot' led to direct mortality through bycatch and, together with recent range-shifts, may have contributed to a localised decline of these dolphins in this winter hotspot since 2007 (de Boer *et al.* 2012).

After extensive consultation, an exclusion grid to reduce common dolphin bycatch in the bass pair trawl fleet was developed and tested at sea with the Scottish Pelagic Fishermen's Association, but no cetaceans were encountered during the initial trial. Among 37 tows observed during March 2002, only two tows had dolphin bycatch (eight animals in total), compared with dolphin bycatch in 11 out of 52 tows in March 2001 (Northridge *et al.* 2011), demonstrating the unpredictable nature of dolphin bycatch.

During the 2004–05 season, some common dolphins were observed to use a 2 by 3 m escape opening fitted into the net midway along its length (Northridge 2006). A barrier immediately behind the escape opening allowed fish to pass but was not passable by dolphins. Nine dolphins were observed escaping with 32 dolphins recovered from the nets having drowned, representing a minimum 22 per cent escape rate (Northridge 2006). Most of the animals that drowned had done so some distance in front of the escape hatch and barrier, indicating that they may have detected the barrier and stopped further forward in the net where they tried to escape. A few other dolphins reached the barrier but did not use the escape hatch, indicating that escape routes need to be more numerous and more obvious (Northridge 2006). Although these trials with exclusion devices showed some promise, the trials ceased in 2006 after intervention from an animal welfare organisation, resulting in subsequent research focusing on the use of acoustic deterrents to reduce bycatch (Northridge *et al.* 2011).

A preliminary model of a barrier to prevent dolphins from entering a trawl net in tuna and sea bass fisheries was tested in a flume tank as part of the 'Necessity' project (Meillat *et al.* 2006), and various bycatch excluder devices were being considered for adaptation to this model (reviewed in Elgin Associates unpublished (b)).

Pingers

Active sound emitting 'pingers' are small self-contained battery-operated devices that are designed to emit regular or randomised acoustic signals at a range of frequencies that are loud enough to alert or deter marine animals from the immediate vicinity of fishing gear, and were originally developed for use in gillnet fishing operations and to deter pinnipeds from mariculture operations (reviewed in Dawson *et al.* 2013, Elgin Associates unpublished (b)). Pingers differ in the level of sound emitted, ranging from relatively low-intensity sounds (less than 150 decibels (dB) re 1 μPa at 1 m) known as acoustic deterrent pingers, through to high output emitters (more than 185 dB) referred to as acoustic harassment devices that were designed to cause discomfort or pain when an animal approached closely (Dawson *et al.* 2013). Other mid-range acoustic emitting devices have been designed to deter depredation by common bottlenose dolphins from static fishing gear or designed to deter bycatch of pelagic dolphin species in mid-water trawls; these are marketed under various trade names such as Dolphin Dissuasive Devices (DDD), Aquamark, Aquatec and Cetasaver, and some of these pingers have higher 165–190 dB source levels (Table 1 in Dawson *et al.* 2013). The term DDD is also more broadly used to refer to loud pingers (Elgin Associates unpublished (b)).

Pingers have been shown to be effective in several gillnet fisheries to reduce the bycatch of small cetaceans and in some cases reduce depredation rates (reviewed in Caretta and Barlow 2011, Geijer and Read 2013, Dawson *et al.* 2013). Significant reductions in bycatch of short-beaked common dolphins, striped dolphins, harbour porpoise *Phocoena phocoena*, franciscana *Pontoporia blainvillei* and some beaked whales, have been demonstrated in replicated experiments and long-term monitoring in some gillnet fisheries (Caretta and Barlow 2011, Dawson *et al.* 2013). For example, in the California-Oregon drift gillnet fishery for swordfish and shark, and the New England groundfish fishery, the mandatory use of pingers has been monitored for more than a decade and bycatch rates of dolphins and porpoises have been reduced by 50–60 per cent (Palka *et al.* 2008, Caretta and Barlow 2011, Dawson *et al.* 2013). However, compliance rates have been highly variable resulting in lower reductions in bycatch than were found in the initial controlled experiments that tested pingers for these fisheries (Geijer and Read 2013, Dawson *et al.* 2013). Continued poor compliance rates with requirements to use pingers in the New England gillnet fishery has resulted in harbour porpoise bycatch exceeding the calculated sustainable PBR levels in recent years (Geijer and Read 2013, Dawson *et al.* 2013). Harbour porpoises appear to avoid areas with pingers, with 14 replicated controlled experiments in North America and Europe showing significant reductions in bycatch associated with the use of pingers (Dawson *et al.* 2013). However, some studies have indicated that some degree of habituation can occur and can result in increased harbour porpoise bycatch over time, particularly in inshore areas where harbour porpoises are at least seasonally resident (reviewed in Dawson *et al.* 2013). Beaked whales have not been recorded as bycatch in the California-Oregon drift gillnet fishery since pingers were used after 1995 (Caretta and Barlow 2011, Geijer and Read 2013).

Cetacean bycatch rates were significantly higher in nets in which one or more pingers had failed or were sparsely distributed (Caretta and Barlow 2011, Dawson *et al.* 2013). Interestingly, harbour porpoise bycatch rates in gillnet fisheries along the northeastern US were more than 2.5 times higher in nets equipped with some pingers but not a full complement, compared with nets without any pingers, which indicates that partial use of pingers may be worse than not using any pingers (Palka *et al.* 2008). For common bottlenose dolphins, studies of pinger use to reduce depredation in gillnet fisheries generally show small and inconsistent improvements in fish catches and some reduced damage to nets, but pingers do not appear to effectively reduce bycatch of bottlenose dolphins (Dawson *et al.* 2013).

The use of pingers has been extended to trials and use in some pelagic trawl fisheries, but in these relatively noise-saturated pelagic trawler operations the effectiveness of pingers in reducing bycatch of cetaceans is unclear (Werner *et al.* 2006, Zeeberg *et al.* 2006), with mixed results reported from different studies (reviewed in Elgin Associates unpublished (b)). Pingers were trialled in the PFTIMF but were found to be ineffective in keeping common bottlenose dolphins out of the trawl net, and pingers were therefore rejected as a dolphin bycatch mitigation method (Stephenson and Wells 2006). Further trials with larger, louder pingers in the PFTIMF have commenced but the results of these trials are not yet known (Allen *et al.* 2014). Allen *et al.* (2014) concluded that pingers are unlikely to deter common bottlenose dolphins from interacting with trawl nets or to mitigate bycatch for this species.

Extensive testing of a range of pingers has been done in the UK bass pair trawl fishery and the adjacent European fishery (Morizur *et al.* 2008, Northridge *et al.* 2011). Trials have evaluated the sound source levels, pulse durations, immersion depths, placement within trawl gear, and distance from dolphin groups in relation to their behavioural responses. Trials have produced mixed results, with significant reductions in bycatch rates observed when loud pingers have been used in some pelagic trawls, however insufficient numbers of control tows during these trials prevent confidence in the results of some experiments (reviewed in Elgin Associates unpublished (b)).

In the UK pair trawl fishery, preliminary trials using standard pingers designed for gillnets showed no reduction in bycatch of common dolphins, however, after the introduction of more powerful DDD pingers (peak source levels approximately 165 dB re 1 μPa at 1 m) as a mitigation device in 2006–07, a substantial 77 per cent reduction in observed bycatch of common dolphins was reported (Table 5.4, Northridge *et al.* 2011, de Boer *et al.* 2012). The overall observed bycatch rate in tows with DDDs during 2007–2009 was 0.178 common dolphins per tow, compared with an overall observed bycatch rate of 0.772 common dolphins per tow for the seasons 2001–06 prior to the use of DDDs (Northridge *et al.* 2011). The lower bycatch rate may be attributed to the use of pingers, but interpretation of these results is complicated by the absence of a significant number of control tows without DDDs with associated dolphin bycatch, hence it is possible that, after 2006, the bycatch rate declined independently of the use of pingers (Elgin Associates unpublished (b)).

Table 5.4 Observed bycatch of common dolphins by season in the UK bass pair trawl fishery before and after the introduction of DDD pingers in the winter of 2006–07

WINTER SEASON ENDING	DAYS	TRIPS	HAULS	DOLPHINS	RATE PER TOW
2001	57	10	92	52	0.565
2002	50	14	91	9	0.099
2003	76	16	113	27	0.239
2004	98	26	136	169	1.243
2005	133	39	176	176	1.000
2006	61	21	53	77	1.453
2007	15	5	34	8	0.235
2008	0	0	0	0	0.000
2009	23	10	28	2	0.071
2010	133	41	188	28	0.149
Totals	646	182	911	548	0.602

Source: Northridge *et al.* (2011).

The vessels involved in the bass pair trawl fishery voluntarily requested pingers and observers each season in recent years to ensure that detailed records were maintained of dolphin bycatch and the deployment patterns and functioning of the pingers (Northridge *et al.* 2011). Some pinger malfunctioning occurred in the 2009–10 season and 28 common dolphins were recorded as bycatch, but Northridge *et al.* (2011) estimated that about 39 fewer dolphins died in bass pair trawls than would have occurred if pingers had not been used. Monitoring of pinger deployment and bycatch in this fishery over three years showed encouraging results, but three potential problems with the use of pingers were identified: pingers were not always fully charged or working when deployed, pingers were sometimes placed in a suboptimal position and needed to be deployed in more than 10 m of water, and pingers can degrade after three years and may not be able to hold adequate charge (Northridge *et al.* 2011). Accordingly, it was recommended that a code of best practice in the fishery should address these issues and ensure that DDDs are fully charged, functioning and deployed on the lower wing ends or bridles of the trawl gear (Northridge *et al.* 2011). In summary, Northridge *et al.* (2011) concluded that DDDs appear to be effective in reducing bycatch of common dolphins, but they noted that there are still important challenges to address including determining the most effective configuration of pingers for mid-water trawls.

Trials with various types of pingers have been completed to test their effectiveness in reducing bycatch of common dolphins by French trawlers in the European bass trawl fishery in the Bay of Biscay (Van Canneyt *et al.* 2007, Morizur *et al.* 2007, 2008). Initial tests with seven models of commercial pingers in August 2005 and August 2006 showed that four models were ineffective at deterring common dolphins, whereas three DDD models caused an obvious change in behaviour but with a variable response level (Van Canneyt *et al.* 2007). Trials were also done using Cetasaver pingers that were developed to mitigate common dolphin bycatch in this fishery as part of the European 'Necessity' project; the Cetasaver uses a conical direction beam that is directed towards the opening of the trawl with an averaged sound level of 178 dB, which results in 139 dB sound level at the entrance to the trawl (Morizur *et al.* 2008). Directivity tests using Cetasaver 3 on six groups of common dolphins showed that when the dolphin swimming direction and the emitted sound direction were 180 ° apart, the dolphins reacted at distances of up to a 200 m (Van Canneyt *et al.* 2007). The Cetasaver 3 system created an acoustic barrier that was effective on all the dolphin groups tested and they did not approach within 200 m in frontal experiments (Van Canneyt *et al.* 2007). French trawler operators prefer to have the Cetasaver set on the rear part of the trawl rather than use the DDD set on the wings of the trawls, because there is less interference with the netsonder because of the geometry of the beams (Morizur *et al.* 2007).

A modified Cetasaver 7 pinger was trialled during winter 2006–07 in the fishery and observers noted that the bycatch of common dolphins decreased by 80 per cent, but the number of test trawls was too low to be conclusive (Morizur *et al.* 2007). More extensive and rigorous trials with Cetasaver pingers were completed in 2007 and 2008, using commercial pelagic trawls in the bass fishery and usually in the presence of scientific observers (Morizur *et al.* 2008). The tests involved a total of 121 tows using the Cetasaver pinger in which six common dolphins were caught in five tows, and a total of 129 tows without the pinger in which 20 dolphins were caught in 10 tows (Morizur *et al.* 2008). The results indicated a reduction in recorded common dolphin bycatch of about 70 per cent during the two years of trials, but analyses showed that twice the number of observations would be needed for the trials to provide statistically significant differences to fully evaluate the effectiveness of Cetasaver pingers (Morizur *et al.* 2008).

At-sea trials off Ireland indicated that Cetasaver pingers may not provide a consistently effective deterrent signal for all groups of common dolphins (Berrow *et al.* 2009), which may explain why bycatch was not suppressed in all trawls in the Bay of Biscay (Morizur *et al.* 2007, 2008). Pingers have been deployed in the New Zealand jack mackerel trawl fishery but it is not known whether the use of pingers has significantly reduced the mortality of common dolphins in this fishery (Mr R. Wells, Deepwater Group Ltd New Zealand pers. comm. in Elgin Associates unpublished (b)).

Another area of uncertainty associated with the use of pingers in trawl fisheries is whether or not dolphins or other cetaceans could become habituated to pingers over time, which could cause a decline in effectiveness. A decline in the effectiveness of pingers in reducing bycatch could be difficult to detect and distinguish from other factors operating in trawl fisheries, and would require detailed studies and long-term assessments of bycatch rates involving observer programmes such as those developed for managing gillnet fisheries (reviewed in Caretta and Barlow 2011, Dawson *et al.* 2013). The widespread use of more powerful pingers such as DDDs to overcome noise from trawl vessels and gear could potentially exclude some cetaceans from important feeding, resting and breeding or nursery areas and may reduce their foraging success and ultimately affect their survival (Zollett and Rosenberg 2005, Northridge *et al.* 2011). Experiments using DDDs to test their potential to exclude cetaceans have produced mixed results, although there was some evidence of decreased cetacean activity up to at least 1.2 km from the DDD and possibly up to a distance of 3 km or more (Northridge *et al.* 2011). Preliminary tests using a less powerful Aquamark 100 pinger appeared to have an effect up to about 400 m (Northridge *et al.* 2011).

Fishing behaviour and codes of practice

A range of voluntary or mandated management measures can be used in trawl fisheries to modify or adapt fishing practices to reduce interactions with cetaceans and other non-target species. These include the use of observer programs and fishers' logbooks to assess the levels and patterns of interactions to enable altered fishing practices, altered timing and depths of trawls, haulback procedures and managing vessel turns.

Independent observer programs are very important for assessing fisheries management options because they can provide more reliable data on interactions and bycatch mortality with non-target species including protected species. Independent observer data can result in higher bycatch mortality estimates than those based on fishers' logbooks that may under-report rates of interactions and bycatch mortality (e.g. Stephenson *et al*. 2008, Hamer *et al*. 2008, Moore *et al*. 2010, Allen *et al*. 2014). Therefore, observer programs can provide more effective data to estimate bycatch within a fishery over time, and for estimating total cetacean bycatch mortality across all fishing effort within a region (e.g. Tuck *et al*. 2013). Reliable data on interactions and bycatch mortality also enable fishing practices to be altered to reduce these risks through adaptive management. Thompson *et al*. (2013) noted that trawl fisheries are often characterised by high fishing effort with low rates of observer coverage, which prevents reliable estimates of bycatch. In New Zealand waters this is particularly important for estimating and trying to reduce the key threat of bycatch in gillnet and trawl fisheries of the endangered endemic Hector's dolphin and the critically endangered Maui's dolphin subspecies *Cephalorhynchus hectori* ssp. *maui*, which are characterised by relatively small and decreasing populations (e.g. Dawson and Slooten 2005, Slooten *et al*. 2006, Slooten and Dawson 2010).

Thompson *et al*. (2013) analysed patterns of bycatch of 135 common dolphins observed captured in trawl fisheries in New Zealand from 1995–96 to 2010–11, including 119 common dolphins captured in the mackerel trawl fishery on the west coast of the North Island. All captures occurred with vessels longer than 90 m, with the majority of common dolphin bycatch occurring with vessels longer than 100 m (Thompson *et al*. 2013). The highest number of captures (70 per cent) occurred during tows where the headline depth (depth of the headline below the surface) was between 10–40 m, with reduced dolphin captures recorded when the headline depth was lower in the water column down to 110 m (Thompson *et al*. 2013). Nine common dolphins were captured at fishing depths between 115 m to 184 m, during the period 2001–05 (Du Fresne *et al*. 2007). About 73 per cent of dolphin bycatch occurred during tows that were between two and six hours duration. Headline depth and trawl duration were the most important variables that best explained patterns of bycatch of these dolphins. Statistical modelling indicated that increasing the headline depth by about 21 m would halve the probability of capture of common dolphins, and decreasing trawl duration would also reduce the probability of capture (Thompson *et al*. 2013). Du Fresne *et al*. (2007) noted that common dolphin bycatch mortality occurred when total winch time of trawls exceeded 24 minutes.

Geographic location, light conditions and lunar phase also influenced patterns of common dolphin bycatch in trawls, with 80 per cent of captures associated with trawls at night particularly during lunar phases with no moonlight (Du Fresne *et al*. 2007, Thompson *et al*. 2013). Diurnal patterns in trawling effort and the extent of moonlight or light spill from trawlers have also been observed to influence cetacean behaviour in other regions and increase the extent of interactions with trawl gear and bycatch at night for some species (e.g. Fertl and Leatherwood 1997, Zollett and Rosenberg 2005, Zeeberg *et al*. 2006). In contrast, most common bottlenose dolphins are caught during daylight hours in the PFTIMF (Stephenson *et al*. 2008, Allen *et al*. 2014). Other factors influencing cetacean bycatch include location of trawling on or near the continental shelf edge, and the season of fishing (Couperus 1997, Zeeberg *et al*. 2006, Fernández-Contreras *et al*. 2010).

Cetaceans are susceptible to capture during different operational phases of trawling. They may become trapped during shooting or haulback if they fail to abandon the net or if the net mouth collapses, and can die if the nets remain submerged in the water for long periods before they are checked (Fertl and Leatherwood 1997, Fernández-Contreras *et al*. 2010). Sharp vessel turns and changes in speed may increase the risk of bycatch (Couperus 1997, Zollett and Rosenberg 2005), although dolphin bycatch can also occur when trawlers are travelling in a straight line at an even towing speed (Northridge *et al*. 2005). Some trawlers use auto-trawl systems that use self-tensioning winches to maintain the shape of the trawl gear when turning, to ensure that the entrance to the net remains open at all times. Net monitoring systems are also designed to maintain net geometry through monitoring and controlling the trawl doors via telemetry and sensors (Elgin Associates unpublished (b)). It has been suggested that the use of auto-trawl systems might reduce the likelihood of marine mammal entrapment from net collapse and therefore may provide some mitigation of the risk of bycatch for pinnipeds and cetaceans (Wakefield *et al*. 2014, Mr G. Geen Seafish Tasmania Pty Ltd pers. comm. in Elgin Associates unpublished (b)). However, auto-trawl systems have not been specifically evaluated as an approach to mitigating marine mammal bycatch, hence their effectiveness in mitigating cetacean bycatch is uncertain (Elgin Associates unpublished (b)).

Wakefield *et al*. (2014) reported on discussions with fishers in the PFTIMF in relation to the potential for entrapment of dolphins following collapse of the net mouth resulting from reduced trawl speed or sharp turning of the vessel during hauling. It was suggested that a small number of the 14 dolphin mortalities recorded in statutory logbooks during a six-month observer program may have resulted from a few instances of net collapse that occurred when a relief skipper, unfamiliar with the operation of the auto-trawl system, was on board. Wakefield *et al*. (2014) noted that development of a vessel operating code of practice may help prevent net collapse, and documenting other standard operational procedures would help maintain a consistent standard of mitigating dolphin interactions. Underwater video records within the trawl net indicated that common bottlenose dolphins may initially become stressed toward the mouth of the net, therefore *in situ* records of dolphin behaviour in this part of the net would enable better understanding of this issue and could lead to development of further mitigation strategies to reduce bycatch (Wakefield *et al*. 2014).

A code of practice was adopted for mid-water trawling in the SPF in 2005, following an incident involving 13 common dolphin mortalities (Mr G. Geen Seafish Tasmania Pty Ltd, pers. comm. 23 April 2013). Following meetings of the Cetacean Mitigation Working Group to develop plans to mitigate bycatch of TEPS, voluntary rules for mid-water trawl operations were implemented by SPF industry members in 2004 and 2005 (Tuck *et al*. 2013). The first rule stated that fishing must stop and the vessel must relocate if dolphins were seen following incidental dolphin captures, and the second rule involved conducting long wide turns to maintain net configuration rather than winching gear to blocks prior to turning (Tuck *et al*. 2013). As discussed in Section 3.2.4 all mid-water trawl vessels in the SPF are now required to have "effective mitigation approaches and devices to the satisfaction of AFMA to minimize interactions with dolphins … ". AFMA implements this requirement through vessel-specific VMPs, consistent with the provisions of the SPF Bycatch and Discard Workplan (AFMA 2011).

Hamer *et al*. (2012) noted that a 'move on' tactic has been used by some longline fishers to try to reduce depredation of fish catches but they concluded that "the success of this strategy seems to be ambiguous at best and is likely to be costly, thus affecting profit margins". Similarly, Tilzey *et al*. (2006) analysed the use of the 'move on' tactic for avoiding pinniped bycatch in the winter blue grenadier fishery off west Tasmania and they concluded it was only occasionally successful because depredating individuals were able to travel long distances to remain with the vessel.

Similar measures to reduce marine mammal bycatch in trawl fisheries in the north-east and mid-Atlantic region have been identified by the Atlantic Trawl Gear Take Reduction Team, which include reducing the number of turns made by the fishing vessel, decreasing tow times at night, and increasing communication between fishers about sightings or incidental takes of marine mammals (Zollett 2009).

Temporal and spatial closures

The only long-term conservation management measure that has been proven to reduce bycatch of small cetaceans in fisheries is the separation of nets and cetaceans in space and time (Reynolds 2008). Time and area closures can be effective in areas where the risks of bycatch are relatively high and consistent, but the utility of such closures are fishery-specific and require detailed knowledge of spatial and temporal use of habitats by cetaceans within the fishery area (Zollett and Rosenberg 2005). Time and area closures may have unintended consequences and cause a shift in the type of fisheries gear used or displace the fishing effort to other areas, which can impact on other populations or different species (Zollett and Rosenberg 2005, O'Keefe *et al*. 2014).

Hamer *et al*. (2012) concluded that the implementation of spatial closures using marine-protected areas to mitigate fisheries impacts on marine mammals has generally proven to be more effective for pinnipeds than for cetaceans because pinnipeds are central placed foragers, which enables their at-sea movements and population trends to be more effectively quantified. They noted that marine protected areas that effectively protect odontocete cetaceans are difficult to implement because: (1) determining where closures should be located is difficult in the absence of reliable data on odontocete movement or migration patterns; (2) protected areas are often smaller than required, due to stakeholder pressure to minimise their impact on fisheries using these areas; (3) monitoring compliance by fishers can be difficult due to the lack of capacity and resources; and (4) quantitative assessment of the performance of these closures is hampered by the statistical uncertainties associated with the limited and potentially unrepresentative data (Hamer *et al*. 2012).

Time and area closures have been used to protect threatened cetacean species from fisheries bycatch in some regions but with mixed success (reviewed in O'Keefe *et al*. 2014). For example, sanctuaries and fisheries closures have been implemented in parts of the range of the endangered Hector's dolphin and critically endangered Maui's dolphin subspecies in New Zealand (Dawson and Slooten 2005, Slooten *et al*. 2006, Slooten 2007, 2013) and for the critically endangered vaquita porpoise *Phocoena sinus* in the northern Gulf of California (Jaramillo-Legorreta *et al*. 2007, Gerrodette and Rojas-Bracho 2011, Senko *et al*. 2014). These closures have reduced the rate of bycatch and enabled the previously rapid rate of decline to be slowed in some areas, which proves that area-based management can work if applied at sufficiently large scales (Slooten 2013). However, because these closures have not encompassed the full range of areas used by these threatened species, their effectiveness in reducing overall population decline and risk of extinction has been compromised (Gerrodette and Rojas-Bracho 2011, Slooten 2013).

In the Australian region, bycatch records in the PFTIMF indicated that common bottlenose dolphins were caught in all fishing areas, across all depths and throughout the year, hence seasonal or spatial adjustments to fishing effort would be unlikely to significantly reduce dolphin bycatch in this fishery (Stephenson *et al*. 2008, Allen *et al*. 2014).

In the SPF area, the proposed Australian sea lion closure area for the DCFA would prevent fishing activity from the DCFA in shelf waters off SA out to a depth of 150 m (Section 5.2.3), and some restriction on the use of mid-water trawls are in place under the South-east Commonwealth Marine Reserves Network occurring within the area of the SPF (see Section 3.2.4). The South-east Commonwealth Marine Reserves Management Plan 2013–23 authorises mid-water trawling in multiple use zones provided that the gear does not come into contact with the seabed, whereas mid-water trawling is not allowed in all other zones (see Figure 3.4 in Section 3.2.4). In the Great Australian Bight Commonwealth Marine Reserve (see Figure 3.5 in Section 3.2.4), mid water trawling is authorised in the Marine Mammal Protection Zone and Benthic Protection Zone provided it does not come in contact with the seafloor; however, all vessel access is prohibited (including all forms of fishing) in the Marine Mammal Protection Zone between 1 May and 31 August.

In the SPF area off SA, a gillnet fishing closure was in force from September 2011 to August 2014 in the Coorong Dolphin Zone (see Figure 5.12), to minimise dolphin bycatch within the GHAT sector of the SESSF (AFMA 2014d). This closure resulted from a significant increase in dolphin bycatch reported by fishers using bottom-set gillnets, mainly in the Coorong Zone east of Kangaroo Island (AFMA 2014d). During late 2010 to September 2011 a total of 52 dolphins were reported to have been caught in gillnets resulting in 50 mortalities, with 18 dolphin mortalities and one live dolphin interaction reported in logbooks for the 2012-13 season (AFMA 2013f, 2014d). Of the 40 dolphins identified 38 were common dolphins and two were bottlenose dolphins (AFMA 2013f). AFMA temporarily closed the area to gillnet fishing while it consulted on a dolphin strategy for minimising gillnet bycatch (AFMA 2014d). AFMA also established a dolphin observation zone adjacent to the closed area (see Figure 5.12 in Section 5.2.3), in which all gillnet fishing was required to be monitored by observers or by e-monitoring systems. Gillnet fishers are now allowed to fish within the Coorong Dolphin Zone provided that they are compliant with the AFMA Dolphin Strategy (AFMA 2014d).

Bycatch trigger limits

As outlined in Section 5.2.3, maximum allowable bycatch limits or PBR limits may be set for a fishery to ensure that bycatch levels allow marine mammal populations to be maintained at a sustainable level or recover from depletion (Hall *et al*. 2000, Roman *et al*. 2013). The IWC has adopted a precautionary approach to cetacean bycatch, recommending that bycatch should not exceed one half of the maximum growth rate of a population. Most odontocete species are considered to be at risk from bycatch mortality in more than one fishery due to their extensive ranges that overlap with multiple fisheries such as gillnet and longline fisheries and in some cases purse seine and trawl fisheries (Shaughnessy *et al*. 2003, Bilgmann *et al*. 2008, 2014, Hamer *et al*. 2008, 2012). Hence, PBR models need to take into account all forms of anthropogenic mortality including impacts from bycatch in all relevant fisheries. However, most cetacean species that occur in the SPF area and more broadly in Australian waters are assessed as Data Deficient because there is insufficient information to determine their population size and trends and current conservation status (reviewed in Ross 2006, Woinarski *et al*. 2014). Therefore, until further detailed research is done on genetic structure (e.g. Bilgmann *et al*. 2008) and the abundance and trends of relevant subpopulations of cetaceans, it is not feasible to determine PBR bycatch limits for cetacean species within the SPF.

Reducing vessel strike of cetaceans

Vessel strike is a threat to some cetacean species and particularly for threatened large whale species with depleted populations (e.g. Laist *et al*. 2001, Kemper *et al*. 2008, Silber *et al*. 2009). As noted in Section 5.3.2, vessel strikes are thought to be relatively common in Australian waters including the SPF area but these are not well documented, and the incidence of vessel strikes is likely to increase in future as some whale populations continue to increase following severe depletion from whaling. The risk of vessel strike from large fishing vessels in the SPF area is uncertain, but international data indicate that most severe or lethal vessel strikes are caused by vessels that are 80 m or longer and which travel at speeds greater than 14–15 kilometres per hour (Laist *et al*. 2001, Vanderlaan and Taggart 2007). Silber *et al*. (2009) noted that reducing the co-occurrence of whales and vessels is the only certain means of reducing vessel strikes, but that this is not possible in many situations, particularly where major shipping routes overlap with whale aggregation areas (Redfern *et al*. 2013). Identification of key feeding grounds or aggregation areas such as the Bonney Upwelling and Perth Canyon, major migration routes, and important calving and nursery grounds for whales and other cetaceans in the SPF area would allow the risk of vessel strike to be assessed in marine spatial planning. A diverse range of cetaceans have been recorded from the Bonney Upwelling region including blue, fin, sei, sperm, killer, pilot and beaked whales, and various dolphin species (Gill 2002, Gill *et al*. 2011, Miller *et al*. 2012); hence this upwelling region has high cetacean activity that increases the risk of vessel strike and other interactions with large trawlers and other vessels. Identification of important habitats for cetaceans can inform marine spatial planning to assess the need for altered shipping routes to reduce the risks of vessel strike (Redfern *et al*. 2013). Alternatively, reduced vessel speed zones could be used to reduce the likelihood of fatal vessel strikes in identified high-risk areas (Redfern *et al*. 2013).

Marine mammal observers can be used to alert vessel crew to the presence of cetaceans and other large marine mammals in the vicinity or path of vessels, but visual detection of marine mammals is often difficult especially in poor weather and low light conditions and at night. Observer programs can be expensive and detection may occur too late to avoid a collision. Potential technological solutions to reduce the risk of vessel strike such as remote sensing using acoustic detections from passive acoustic or sonar devices, radar, and thermal imaging, may improve detection of whales and other cetaceans, but these technologies are not yet proven to be capable of significantly reducing vessel strikes and require further development and testing (Silber *et al*. 2009).

Summary: management of interactions

- *Management and mitigation measures that have some potential for reducing direct interactions with and associated bycatch of some species of cetaceans include excluder devices and other gear modifications, acoustic deterrent pingers, modified fishing practices, temporal and spatial closures, bycatch triggers and move-on rules.*
- *Excluder designs tested to date have not been consistently effective in reducing cetacean bycatch in trawls, and at present there is no solution to filter or deter cetaceans from entering the net opening.*
- *Excluder devices have reduced bycatch mortality of some marine megafauna in some trawl fisheries in Australian waters and internationally, but these need to be carefully designed and optimised for each fishery and for different species of cetaceans.*
  - *Underwater cameras have shown very high rates of interaction between dolphins and trawl operations in some fisheries, and further research and monitoring is needed to understand the behaviour of cetaceans in trawl nets. Common dolphins and bottlenose dolphins may behave differently when constrained within nets and may require different excluder designs and location of escape holes, which complicates the development and optimisation of excluder devices in the SPF area where both species occur.*
- *Acoustic pingers have been effective in reducing bycatch of some cetaceans in some gillnet fisheries, but their effectiveness in reducing cetacean bycatch in relatively noise-saturated pelagic trawl fisheries is unclear, with mixed results reported in different studies in Australia and overseas.*
  - *Some studies have reported significant reductions in bycatch mortality of common dolphins. But pingers appear unlikely to deter common bottlenose dolphins from interacting with trawl nets or effectively mitigate bycatch for this species.*
- *Codes of practice to reduce the risk of interactions include suspension of fishing and relocation to another area following bycatch events, but the success of the 'move on' tactic for cetaceans is uncertain.*
- *Spatial and temporal fishing closures can reduce interactions and bycatch mortality of cetaceans where the risks of interactions and bycatch are relatively high and consistent and where closures encompass sufficient parts of the range. However, effective planning of closures requires detailed knowledge of spatial and temporal use of habitats, which is lacking for most cetacean species in the SPF area.*

- *Data on population size and trends, genetic structure, and mortality from fisheries bycatch and other anthropogenic threats are lacking for most cetacean species in the SPF area. This precludes the development of population demographic models needed to determine sustainable biological removal limits for these species and bycatch trigger limits for cetaceans in the SPF mid-water trawl fishery.*
- *Independent observer programs are very important for assessing fisheries management options because they provide more reliable data on cetacean interactions and bycatch mortality, enabling adaptive management to reduce the risks of interactions.*

Proposed management of direct interactions of the DCFA and cetaceans

To manage and mitigate direct interactions with dolphins, the DCFA would have been subject to the provisions of the SPF Management Plan, SPF Harvest Strategy and SPF Bycatch and Discard Workplan and to provisions (a), (b), (f) and (g) of Condition 1 in the Schedule to the accreditation of the SPF made by the Environment Minister, as follows:

(a) Prior to fishing, have in place demonstrably effective and scientifically proven mitigation approaches and devices to the satisfaction of AFMA to minimise interactions with dolphins, seals and seabirds, including gear handling and net setting rules. These mitigation devices must, as a minimum, include best practice seal excluder devices with top opening escape hatches or equivalent mechanisms

(b) In the event of one or more dolphin mortalities as a result of the mid-water trawl fishing activities:

   i. suspend fishing;

   ii. consult with any AFMA observer onboard and review the effectiveness of mitigation measures; and

   iii. not recommence fishing within 50 nm of the mortality event.

(f) Ensure that there is an on-board observer at all times with 24 hour monitoring of mid-water trawl fishing activities and there is an underwater camera record of the operation of any bycatch excluder device at all times, and reviewed by an observer each day. The requirements under this Condition will apply to 1 November 2013 with monitoring arrangements to apply after this date to be determined following a review by AFMA and the Department.

(g) When fishing, report daily to AFMA on the level of protected species interactions, including mortalities.

As noted in Section 3.2.4, all mid-water trawl vessels in the SPF are now subject to the following conditions.

- Prior to fishing, mid-water trawl vessels must have in place effective mitigation approaches and devices to the satisfaction of AFMA to minimise interactions with dolphins, seals and seabirds.
- AFMA requires that at least one observer be deployed on each new mid-water trawl vessel for the first 10 fishing trips with additional observer coverage or other monitoring implemented as appropriate, following scientific assessment of the SPF.

Furthermore, the DCFA would also have been subject to conditions proposed in the *FV Abel Tasman* Seal and Dolphin Management Plan (Box 5.1), and those requirements relevant to dolphins (and potentially to other cetaceans) are evaluated in Section 5.3.5 together with assessment of their likely effectiveness.

5.3.4 Assessment of the likely nature and extent of direct interactions by the DCFA with cetaceans

The panel's assessment of the likely nature and extent of direct interactions of dolphins and other cetaceans with the DCFA is based on the review of the available information on the 21 cetacean species in the SPF area that are considered to be most relevant to this assessment (Section 5.3.1), and the review of available information on interactions between cetaceans and trawl fisheries in Australia and internationally (Section 5.3.2).

As noted above, there is considerable uncertainty with respect to the nature, extent and risks to cetaceans from interactions with mid-water trawl operations in the SPF and therefore considerable uncertainty with respect to interactions with the DCFA. This uncertainty arises from a number of issues including the lack of information or insufficient knowledge about the distribution, abundance and current conservation status of most cetacean species within the SPF area; the location and seasonal use of important feeding, breeding and nursery grounds and migration pathways for most cetacean species within the SPF fishing area; the extent to which dolphins and other cetacean species feed on or rely on small pelagic fish in their diets; the range of dolphin and other cetacean species that would be likely to directly interact with the DCFA, and the extent to which some may become habituated to fishing activities; and the risks and effects of bycatch mortality or vessel strikes on populations of these cetaceans (e.g. Ross 2006, Bannister 2008, Bilgmann *et al*. 2008, Woinarski *et al*. 2014). In addition, the spatial and temporal distribution of fishing effort of the DCFA cannot be predicted with any confidence.

Cetaceans are a diverse group of intelligent marine mammals, and some species are readily attracted to fishing operations which increases their risk of injury or mortality (e.g. Fertl and Leatherwood 1997, Broadhurst 1998, Zollett and Rosenberg 2005, Allen *et al*. 2014). Therefore, the nature and extent of direct interactions with the DCFA is likely to vary significantly among cetacean species, and these interactions include feeding within nets, behavioural changes leading to increased interactions with fisheries, injury or mortality from incidental bycatch, injury or mortality from vessel collision, and acoustic disturbance (see Section 5.3.2). The risks associated with these interactions will also vary among species, ranging from higher risk species such as short-beaked common dolphins and bottlenose dolphins that are known to feed on small pelagic fish and interact extensively with trawl fisheries leading to intermittent bycatch, through to larger whale species that do not usually feed on small pelagic fish but may be at higher risk from vessel strike or acoustic disturbance (Bannister *et al*. 1996, Ross 2006, Bannister 2008, Woinarski *et al*. 2014).

Based on the previous records of dolphin and other cetacean bycatch mortality with mid-water trawls in the SPF and elsewhere, the panel considers that some degree of cetacean bycatch would be likely to occur with the DCFA. Direct interactions, injuries and mortality are likely to be higher in areas of increased cetacean abundance that overlap with zones of increased fishing intensity, and particularly for cetaceans that are opportunistic feeders able to take advantage of herding and aggregation of fish resulting from trawling activities.

The panel noted that very little information is available on direct interactions between cetaceans and mid-water trawls in the SPF area. Based on the limited recorded bycatch mortality of 25 dolphins including common bottlenose dolphins and possibly short-beaked common dolphins in mid-water trawls off Tasmania in the SPF during 2001–10 (Lyle and Willcox 2008, Tuck *et al*. 2013), cetacean interactions resulting in bycatch have been considered to be relatively rare and unpredictable events. However, a large mid-water trawl freezer vessel such as that used in the DCFA would enable fishing to potentially occur throughout the SPF area wherever target fish are available; hence a greater range of habitats and increased numbers of cetacean species may be encountered compared with previous trawling operations in the SPF off Tasmania. This could result in a greater range of cetacean species interacting with the DCFA, increasing the uncertainty about the likely nature and extent of direct interactions with cetaceans. Spatial and temporal patterns of distribution and abundance of cetacean species are highly variable throughout the SPF area (see Section 5.3.1). Some species exhibit marked changes in distribution and abundance resulting from seasonal aggregations and use of feeding grounds, annual or multi-year breeding cycles and migratory movements (Ross 2006, Bannister 2008, Woinarski *et al*. 2014), which further increases the uncertainty about the likelihood of interactions with cetaceans in the DCFA.

The extent to which mortality from bycatch or vessel strike arising from interactions with the DCFA may affect cetacean populations is unknown, because most cetacean species in the SPF area are categorised as Data Deficient based on insufficient knowledge of their population size and trends and other relevant parameters required to assess their conservation status (Woinarski *et al*. 2014).

Summary: likely nature and extent of direct interactions by the DCFA with cetaceans

- *There is considerable uncertainty about the nature, extent and risks to dolphins and other cetaceans in the SPF area from interactions with mid-water trawl operations under the DCFA.*
- *Interactions would vary among species, with higher risk species such as bottlenose dolphins and short-beaked common dolphins known to prey on small pelagic fish and interact extensively with trawl fisheries, whereas some larger whale species may be at higher risk from vessel strike or acoustic disturbance.*

- *The risks of mortality from bycatch or vessel strike for the cetacean species likely to interact with the DCFA are uncertain, but some bycatch mortality could be predicted to occur based on previous reported bycatch of dolphins in mid-water trawls in the SPF area and interactions with trawl fisheries reported in other regions.*
- *The DCFA would enable fishing to occur throughout the SPF area, which would increase the range of cetacean species likely to be encountered.*

5.3.5 Assessment of the effectiveness of proposed management measures to mitigate interactions by the DCFA with cetaceans

The proposed mitigation measures for the DCFA as specified in the Seal and Dolphin VMP (Box 5.1) are assessed below. The panel noted that the VMP refers specifically to dolphins, whereas a range of other cetacean species that occur in the SPF area are at risk of interacting with the DCFA (refer to Sections 5.3.1 and 5.3.4) and therefore mitigation measures specified in the VMP are potentially relevant to other cetaceans.

Mandatory gear requirements: exclusion device

The Seal and Dolphin VMP specifies that the DCFA must have an AFMA approved seal and dolphin excluder device installed within the net at all times while conducting fishing operations.

The panel supports the requirement to have an excluder device installed within the net at all times while fishing. However, the panel noted that excluder devices are often designed to mitigate bycatch of seals or dolphins rather than being optimised for both groups of marine mammals. Hence, a single type of excluder device may mitigate bycatch for pinnipeds but may be less successful for cetaceans, and vice versa. For example, dolphins may need a different spacing of bars in the excluder grid to reduce the risk of their tail flukes becoming trapped, compared with an optimal grid design for excluding seals. There is some evidence indicating that different dolphin species may react differently to excluder devices and their location in relation to the mouth of the net (see Section 5.3.3). Furthermore, the panel noted that a range of designs for dolphin excluder devices have been developed and tested in various fisheries with varying degrees of success, hence at present the optimum design for effectively mitigating bycatch of dolphins and other cetaceans is uncertain. The efficacy of using a SED for mitigating bycatch of dolphins is highly uncertain, as dolphins and seals may behave differently within trawl nets and in response to excluder devices and the location of the escape hole (see Section 5.3.3). Furthermore, previous research has shown that excluder devices and their position within trawl nets may need to be varied for different species of dolphins (e.g. Northridge *et al.* 2005, Stephenson *et al.* 2008, Wakefield *et al.* 2014), which complicates the design process and optimisation of an excluder device for different types of marine mammals.

Under Condition 1(a) (see Section 5.3.3), a best practice seal excluder device with top opening escape hatch or equivalent mechanism would be required to be used; however the panel considers that the efficacy of such a device in mitigating dolphin bycatch is uncertain and may provide only opportunistic or suboptimal mitigation of interactions with dolphins and other species of cetaceans likely to be encountered in the DCFA. What constitutes best practice remains open to question.

The VMP requires that the escape hole on the excluder device is upward opening and has a hood attached to reduce any potential fish loss and to not allow any large animals to fall out of the net if immobile and the net is inverted. The panel noted that there is evidence of some dolphins escaping through upward opening escape holes; however some captive female bottlenose dolphins have been reported to prefer downward opening escape holes and the behaviour of most cetacean species in trawl nets is unknown. Therefore, the efficacy of this approach for mitigating interactions with different dolphin species or other cetaceans likely to interact with the DCFA is uncertain, and requires further research. The panel supports the requirement to attach a hood to the excluder device to increase the chance of retaining any bycatch of dolphins and other species.

As noted in Section 5.2.5, the SED proposed for use in the *FV Abel Tasman* was designed by Maritiem and was intended to have a soft-fibre grid rather than a hard grid, with a top opening escape hole and a hood cover to retain bycatch. However, the efficacy of this excluder device in mitigating seal and dolphin interactions is uncertain, as a soft grid has previously proved less effective than a hard metal grid for directing seals towards the escape hole (see Section 5.2.5), and this new SED design would require extensive testing at sea and monitoring to optimise its efficiency over an extended developmental period (e.g. Northridge *et al.* 2005, Zeeberg *et al.* 2006, Lyle and Willcox 2008, Stephenson *et al.* 2008).

Mandatory fishing operation requirements

The panel supports the requirement to have an AFMA observer onboard the DCFA at all times, noting that it interprets this as requiring that there must be AFMA observer coverage (observer onboard and/or electronic) of all fishing activities to ensure full observer coverage and electronic monitoring of all shots and hauls.

The panel supports the requirement to ensure the boat has underwater cameras operational at all times while undertaking fishing and that those cameras constantly record any take of bycatch and/or behaviours near the excluder device. The panel noted the importance of recording all interactions with dolphins or other cetaceans including their behaviour around the excluder grid and escape hole, and recording the number, condition and timing of dolphins and potentially other cetaceans that use the escape hole or are taken as bycatch. Multiple underwater cameras would need to be used to ensure coverage of interactions near the excluder device and the escape hole. Recording of dolphin behaviour within the net nearer the net mouth would also be beneficial based on observations in the PFTIMF that common bottlenose dolphins appear to become distressed initially near the net mouth (e.g. Wakefield *et al*. 2014).

The panel supports the requirement to allow the on-board AFMA observer access to review any footage recorded by the underwater camera at least once every 24 hours for the duration of each fishing trip. However, the panel noted the importance of monitoring and assessing the extent of interactions with dolphins and potentially other cetaceans on as near a real-time basis as possible, so that Condition 1(b) to suspend fishing (see Section 5.3.3) can be complied with in the event of one or more dolphin mortalities occurring. The panel interprets this requirement to indicate that all recorded footage would be archived for future reference so all fishing operations can be reviewed, noting that it may be possible only to monitor and review a subset of trawls and hauls onboard. Therefore, if one or more dolphin mortalities occurred but were not quickly detected through real-time monitoring, a subsequent review of the footage would occur too late to suspend fishing in the immediate area in which the mortality event occurred and consequently the risk of further bycatch would not be mitigated.

The panel supports the requirement to not deploy any trawl nets if dolphins are sighted around the boat by any crew member or on-board observers, until the dolphins have dispersed of their own accord or the boat has steamed away and are no longer in sight of the boat. This requirement refers only to dolphins but it is likely that other cetaceans may be intermittently present in the vicinity of the vessel. The panel also noted that visual observations of dolphins and other cetaceans near the vessel by crew members and on-board observers will be dependent upon sighting efficiency and detectability that will vary during vessel operations and are highly dependent on time of day (significantly lower during low light and at night), sea state and weather conditions, which introduces further uncertainty in assessing this requirement. Most trawls in the SPF commence at night when visual detection of cetaceans would be very limited. Remote sensing technology could be considered for detecting cetaceans in the vicinity of the vessel using passive acoustic or sonar devices, radar, or thermal imaging. However, Silber *et al*. (2009) noted that these technologies need further development and testing and are likely to be costly. Dolphins and other cetaceans may remain submerged below the surface for some time so detectability will vary depending upon their behaviour states and dive duration. It is unlikely that an on-board observer would be able to continually view all areas around the vessel to ensure dolphins are not present prior to and during deployment and hauling of nets. The time required for dolphins and other cetaceans to disperse sufficient distance away from the vessel to reduce the risk of interactions is unknown, and the distance the vessel should steam away from the area of dolphin sightings to reduce the risk of interactions is uncertain. The panel noted the requirement that trawl nets should not be deployed if seals are sighted within 300 m of the vessel, but the distance between the vessel and dolphins is not similarly specified.

The panel supports the requirement that observers are notified prior to the deployment and/or the recovery of trawl nets, day and night, in order to allow the observers to be present to detect any seals which become enfolded or caught at the surface, so the animals can be rapidly and humanely released. The panel noted that this requirement is also relevant for detection and rapid and humane release of any dolphins and potentially other cetaceans that may become enfolded or caught in the net.

The panel supports the requirement to take all reasonable steps to ensure that, as far as practicable, if a seal or dolphin is captured in a trawl net as a result of fishing operations, the mammal is released alive and unharmed, and noted that this measure is also relevant to other cetaceans.

The panel supports the requirement to record any interaction with any protected species in a logbook onboard the boat and notify AFMA in writing, detailing any interactions with protected species, including any mortalities, every 24 hours for the duration of each fishing trip. The panel noted that the degree to which all interactions with protected species are recorded in a logbook and reported to AFMA every 24 hours will be dependent upon the coverage of the underwater cameras and the extent to which all shots and hauls are reviewed within each 24-hour period. The ability to identify and report on the species of dolphin or other cetacean interacting with the DCFA would be determined by the extent of training in species identification for observers and crew. Identification of cetacean species at sea is often difficult. If dead dolphins or other cetaceans are retained in the net or hood and brought onboard, there is increased scope for detailed examination and identification by trained observers. Further, the collection of small biopsy samples for genetic analysis would enable the species identification to be confirmed. Accurate identification and reporting of interactions with dolphin and other cetacean species would enable management measures to be more effectively adapted to account for different species.

Mandatory interaction requirements

The VMP requires that, if fishing operations conducted by the method of mid-water trawling result in the death of one or more dolphins in any one shot the holder must:

- suspend fishing immediately
- notify the AFMA observer onboard of the dolphin mortalities and with the assistance of the AFMA observer review the effectiveness of mitigation measures used in fishing operations
- not recommence fishing within 50 nm of the event.

The panel supports the requirement to suspend fishing immediately when a mortality is detected. However, it noted that it is likely that some mortalities will not be immediately known to the crew or the observer, but may subsequently be identified on review of video recordings which would reduce the efficacy of this requirement, i.e. by the time the mortality has been identified from recordings the vessel will no longer be in the area.

The panel supports the requirement to notify the AFMA observer of any mortality event. However, the panel noted that in practice there will be considerable uncertainty with respect to how the effectiveness of mitigation measures used in fishing operations could be reviewed at sea to decrease the risk of further interactions and mortalities.

The panel supports the move-on rule in principle, but noted that the rationale for requiring the vessel to move away a distance of 50 nm is unclear. This distance may reduce the likelihood of further interactions by some smaller cetaceans, however, some cetaceans are known to follow fishing vessels for extended periods once depredation events have occurred so whether 50 nm is too large or too short a distance to move is uncertain. As with pinnipeds, it is possible that in randomly moving a set distance, fishing operations could move to an area with higher cetacean densities where interactions are more, rather than less likely. Further, as noted above, mortalities detected by the underwater camera are unlikely to be identified on a real-time basis and some mortality may remain undetected until the gear is hauled, so the likely effectiveness of the move-on rule is uncertain.

Panel assessment and advice: effectiveness of proposed measures and actions to avoid, reduce and mitigate impacts of direct interactions by the DCFA with cetaceans

<u>Assessment: effectiveness of proposed measures</u>

- *The efficacy of the proposed mitigation measures for dolphin interactions with the DCFA is highly uncertain, primarily because these measures have not proven to be consistently effective at mitigating bycatch of dolphins and other cetaceans in other fisheries, nor specifically in the SPF. In particular:*
  - *there is no currently accepted optimum excluder device for mitigating interactions with dolphins and there is evidence to suggest that a single excluder device may not effectively mitigate bycatch of both seals and dolphins, or different species of dolphins*
  - *effective enforcement of measures related to dolphin mortality and suspension of fishing and move-on rules is limited by the capacity for rapid detection of all mortalities using delayed review of underwater recordings that would reduce the efficacy of these measures*
  - *the effectiveness of measures that require a response to sightings of dolphins is questionable given that most fishing in the SPF takes place at night and visual detection of dolphins is highly variable at other times.*

Advice: actions to avoid, reduce and mitigate adverse environmental impacts of the DCFA

- *Use an excluder device only after its operation has been optimised for the vessel, fishery and different dolphin species , including both bottlenose and short-beaked common dolphins, under a scientific permit with the required level of performance developed in consultation with experts, noting that excluder designs tested to date have not been consistently effective in reducing cetacean bycatch in trawls, and at present there is no solution to filter or deter cetaceans from entering the net opening.*
- *Use underwater video to monitor dolphin behaviour within the net and around the excluder device to determine the efficacy of the excluder device and levels of cryptic mortality.*
- *Introduce a bycatch rate trigger limit for dolphin species for the fishery or fishing areas, or a total mortality trigger for a fishing season and/or fishing areas on a precautionary rather than evidentiary basis.*
- *Replace the 50 nm move-on rule in response to a single dolphin mortality, with a requirement to move to an area where interactions with cetaceans are less likely, based on available data on estimated at sea density distributions.*
- *Assess the efficacy of acoustic deterrent pingers (during rigorous controlled trials under scientific permit with the required level of performance developed in consultation with experts), and temporal and spatial closures, that have been shown elsewhere to have potential to reduce the risk of interactions for some cetacean species, including dolphins.*
- *Prohibit the discard of any biological waste (excluding the release of any protected fauna) noting that this was a requirement of the proposed seabird VMP.*
- *Ensure 100 per cent observer coverage of fishing operations and, if trigger limits are used in conjunction with move-on rules or requirements to review mitigation measures, provide sufficient observer capacity to ensure that underwater video footage is monitored at the end of each shot to maximise response times to mortalities.*
- *In addition to the above actions to mitigate impacts on dolphins, ensure that monitoring and agreed management responses are in place to allow a timely management response if other cetacean species interact with the DCFA.*

5.3.6 Monitoring and research

The previous Sections have highlighted the considerable uncertainties associated with assessing the likely nature and extent of direct interactions of cetaceans with the DCFA (Section 5.3.4), and the efficacy of the proposed management measures to mitigate interactions with cetaceans and the DCFA (Section 5.3.4). These uncertainties require further monitoring and research to improve knowledge so that these issues can be more effectively addressed. The key questions that arise from assessment of the DCFA in relation to likely interactions with cetaceans, and the rationale for these, are outlined below.

1. What regions in the SPF area are important habitats used by cetaceans that have increased risk of interactions with the DCFA?

As noted in Section 5.3.1, remarkably little information is available on the distribution and abundance and important habitat areas used by most cetaceans in the SPF area for aggregating, feeding, breeding and nursery areas (Ross 2006, Bannister 2008, Woinarski *et al.* 2014). Important seasonal aggregation and feeding habitat areas are known for some larger whales such as the Bonney Upwelling and Perth Canyon where endangered blue whales and a diverse range of other cetaceans have been recorded. Major migratory pathways for humpback whales and some other cetaceans, and breeding or aggregation grounds for southern right whales and the depleted population of sperm whales, occur in various locations along southern Australia within the SPF area. There is insufficient knowledge of important habitat areas for the depleted south-east subpopulation of southern right whales, and the location of important habitats and areas of increased abundance for most other cetaceans in the SPF area are unknown. This is particularly problematic for smaller cetaceans including bottlenose dolphins and short-beaked common dolphins that are known to be at particularly high risk of interactions and bycatch from trawl fisheries. Improved knowledge and identification of important habitats for cetaceans occurring in the SPF area is essential to enable hotspots of increased cetacean abundance and activity to be identified, so that the degree of overlap with potential areas of increased fishing activity likely to occur from the DCFA could be determined. This would allow areas of increased risk of interactions between cetaceans and the DCFA to be identified, which in turn would enable more effective spatial planning for management of the DCFA. Risk-based management would enable assessment of the need for, and likely efficacy, of seasonal spatial closures for the DCFA to reduce the likelihood of interactions with cetaceans and adverse outcomes arising from net feeding, entanglement and bycatch, noise disturbance and vessel strike. Further research using satellite tracking of cetaceans would provide essential information for identifying important habitats and seasonal and migratory movements within the SPF area, and the potential for overlap with fishing activities in the DCFA.

2. <u>What levels of mortality arising from interactions with the DCFA could be sustained by cetacean populations in the SPF area?</u>

The most recent comprehensive assessment of the conservation status of Australian mammals (Woinarski *et al*. 2014) concluded that blue whales are Endangered with Antarctic blue whales Critically Endangered, fin whales and sei whales are Endangered, and sperm whales are Vulnerable, hence these species are at more obvious risk of anthropogenic impacts. As the abundance and population trends of most cetacean species occurring in the SPF area are unknown and most of these species are assessed as Data Deficient, assessment of the likely impacts of fishing and other anthropogenic threats is seriously impaired (Ross 2006, Woinarski *et al*. 2014). Therefore, further monitoring and research on population size and trends, and research on genetic structure to identify management units within these populations are essential to provide the information needed to develop population models that can be used to assess the likely impacts of mortality arising from interactions with the DCFA. At present it is not possible to effectively determine PBRs for fishing-related mortality for cetacean species at risk of interactions with the DCFA, which hinders assessment of the potential for adverse environmental impacts arising from the altered fishing practices associated with the DCFA. This information is necessary to assess the DCFA against Part 13 of the EPBC Act in relation to interactions with EPBC Act listed species in Commonwealth waters, which requires that "the fishery does not, or is not likely to, adversely affect the conservation status of protected species or affect the survival and recovery of listed threatened species".

Adequate monitoring and reporting of interactions and bycatch mortality are necessary for calculating sustainable PBRs in relation to the DCFA and other fisheries in this region that are known to cause mortality of cetaceans. The panel also considers that a total mortality trigger leading to suspension of fishing operation to enable a detailed review of the fishing operations and mitigation measures is needed (refer to Section 5.3.5). Therefore, the panel reaffirms the importance of ensuring full independent observer coverage and monitoring of fishing activities at all times, as part of the proposed mandatory fishing operation requirements (see Section 5.3.5). As noted in Section 5.3.3 independent observer programs are important because they provide more reliable data on interactions with TEPS and bycatch mortality (e.g. Stephenson *et al*. 2008, Hamer *et al*. 2008, Tuck *et al*. 2013, Allen *et al*. 2014). Where there is sufficient observer coverage, data on interactions and bycatch mortality can be used to effectively assess the performance of fisheries operations and management measures to reduce interactions and bycatch. Tuck *et al*. (2013) concluded that a number of measures that have been introduced to reduce interactions with TEPS and bycatch in Australia's Commonwealth-managed fisheries have been successful to varying degrees, but that data availability or precision are insufficient for some fisheries to judge the effectiveness of these measures. Adequate monitoring and reporting of interactions is also required for estimating total cetacean bycatch mortality across all fishing effort within a region (Tuck *et al*. 2013), and therefore for identifying the extent to which the DCFA might contribute to this total fishing mortality. An important issue related to adequate monitoring and reporting of interactions and bycatch mortality, is to ensure that observer training and fisher education programmes include training in identification of cetacean species and other protected species to improve records of interactions and bycatch, and to more effectively identify species of concern.

3. What modifications to the proposed fishing gear and operations of the DCFA are needed to improve management and reduce the potential for interactions including bycatch of cetaceans?

The panel concludes that substantial research and monitoring are needed to assess the effectiveness of the proposed management measures to mitigate cetacean interactions with the DCFA (see Section 5.3.5). In relation to the SED proposed for use in the DCFA, detailed research and monitoring using a quantitative experimental design and trials conducted at sea would be needed to evaluate SED performance and provide the data needed to optimise its effectiveness in mitigating bycatch. Different designs of excluder devices have been developed for reducing bycatch of common dolphins and bottlenose dolphins (see Section 5.3.3). An upward opening escape hole seems to be preferred by some small cetaceans, but some captive female common bottlenose dolphins have been reported to prefer a downward opening escape hole. These issues complicate the design process; hence it may not be possible to optimise an excluder device to effectively mitigate bycatch of all cetacean species and pinnipeds that could potentially interact with the DCFA. The performance of the proposed auto-trawl system to maintain net integrity and SED performance also needs to be tested during trials at sea.

Specific aspects of the excluder device that require research and monitoring include the performance of the proposed soft grid versus the use of a hard grid, the optimal placement of the excluder within the net, the spacing of the grid bars and angle of the grid, and the position and size of the escape hole to facilitate safe exit of cetaceans and other TEPS. The performance of the hood in retaining bycatch would also need to be evaluated during trials at sea. Underwater video cameras and archived recordings of cetacean behaviour within the net and near the excluder device are needed to monitor and evaluate the efficacy of the excluder device, and to facilitate adaptations of the design. Monitoring the health status of cetaceans and other TEPS that exit from the escape hole and post-release survival of any identified individuals would help to reduce uncertainties about cryptic mortality rates. As excluder devices are not consistently effective at reducing bycatch, underwater monitoring of the behaviour of dolphins and other cetaceans during night versus day trawls and in the forward part of the net would be beneficial in assessing the extent to which cetaceans become distressed within different areas of the net, leading to reduced probability of survival.

Other mitigation measures that could be assessed in relation to reducing interactions with cetaceans include the use of acoustic deterrent pingers such as DDDs, and physical deterrents to prevent cetaceans entering the net (see Section 5.3.3). Further research is required to develop monitoring systems that will enable remote detection of cetaceans in the vicinity or path of the fishing vessel to reduce the risk of vessel strike and other interactions, particularly when visual observation is impaired.

Research on aspects of fishing operations that would be beneficial in assessing rates of cetacean interactions and risks of bycatch mortality include comparisons of night versus day trawls, variation in net headline depth, tow duration and speed, and the distance required for the vessel to move away from an area in which bycatch mortality occurred to reduce the risks of interacting cetaceans remaining with the vessel during relocation to a new fishing area.

Panel advice: research and monitoring to reduce uncertainties

Research that addresses the following questions could help to reduce uncertainties about the potential for adverse environmental impacts of the DCFA on protected cetacean species:

- *What regions in the SPF area are important habitats used by cetaceans that have increased risk of interactions with the DCFA?*
- *What levels of mortality arising from interactions with the DCFA could be sustained by cetacean populations in the SPF area?*
- *What modifications to the proposed fishing gear and operations of the DCFA are needed to improve management and reduce the potential for interactions including bycatch of cetaceans?*

In addition, observer data on all cetacean interactions should be collected, analysed and published.

5.4 Seabirds

Little information or data specific to seabird bycatch in large mid-water trawl freezer vessels was located. The panel's assessment is, therefore, based largely on information available on the impacts on seabirds of trawling more generally, and mid-water trawling, in particular, including in the SPF.

5.4.1 Species

There are 89 protected species of seabirds that occur within the SPF area (see Appendix 3). Seabird species richness overlaid with mid-water trawl effort in the SPF is shown in Figure 5.20. Of those, the groups most impacted by direct interactions with fisheries are albatrosses and petrels (Baker *et al.* 2002, Croxall 1998 cited in Elgin unpublished (a)). The ERA for the SPF mid-water trawl sector (Daley *et al.* 2007b) assessed 76 bird species of which 53 were albatrosses and petrels. The remainder comprised penguins, cormorants, gannets, boobies, tropicbirds, skuas, gulls and terns, which are considered likely to be of lower risk from mortality in trawl fishing operations (Elgin Associates unpublished (a)).

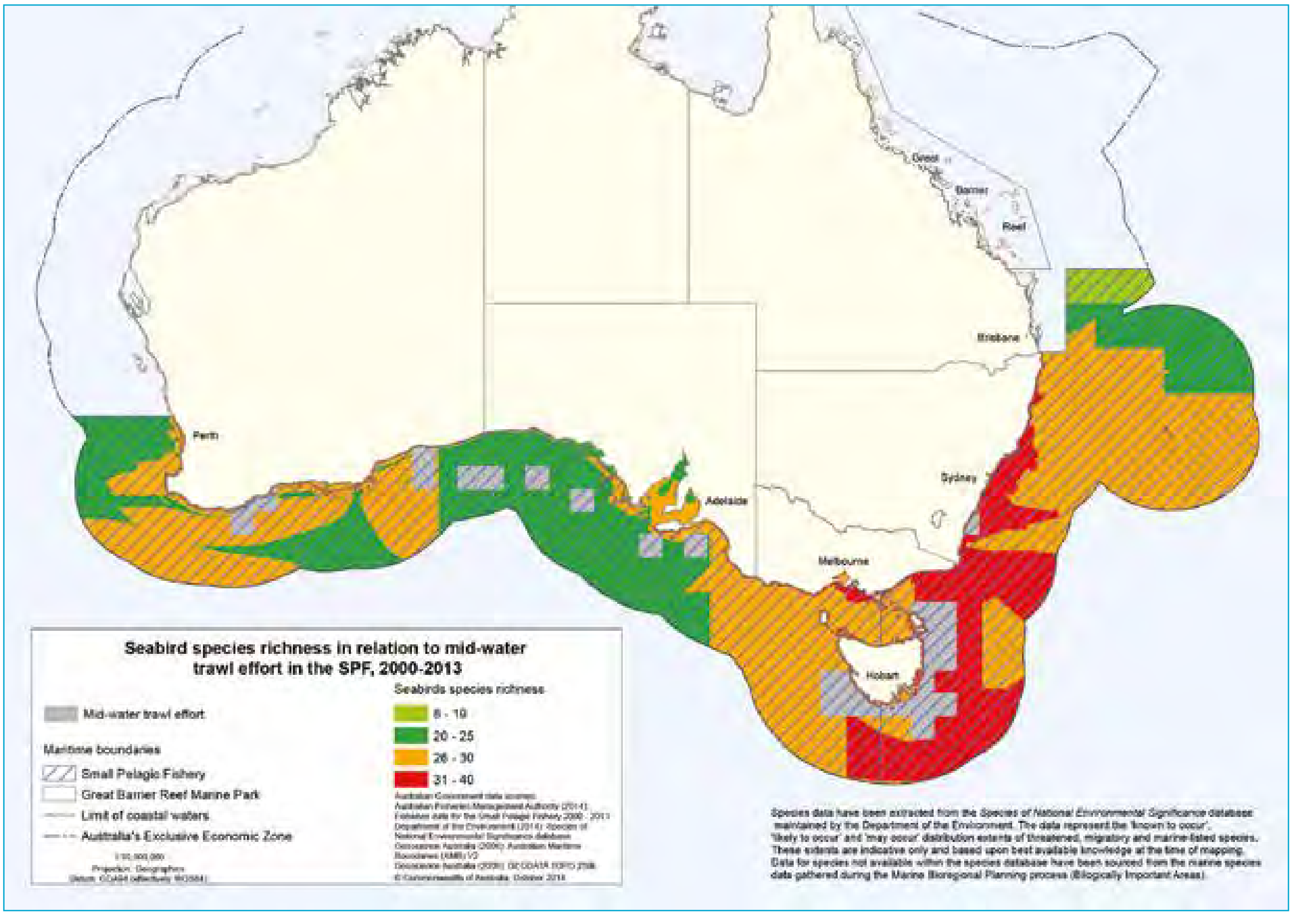

Figure 5.20 Seabird species in relation to the total area of waters fished in the SPF using mid-water trawl during 2000–2013. **Source:** Map produced by ERIN using unpublished AFMA data.

Of the 76 bird species assessed in the ERA, only three (shy albatross *Thalassarche cauta*, Chatham albatross *T. eremita* and black-browed albatross *T. melanophris*) were assessed at 'high' risk (Daley *et al.* 2007b). These assessments were reduced to 'medium' risk as a result of the residual risk assessment (AFMA 2010b). Thus, as a result of the ecological risk assessment and management processes, all bird species assessed were found to be at medium (43) or low (33) risk from mid-water trawl operations in the SPF.

5.4.2 Nature, extent and management of seabird interactions in the SPF

Extent of interactions in the SPF to date

The National Recovery Plan for Threatened Albatrosses and Giant Petrels 2011–2016, identifies the most critical foraging habitat for these species to be those waters south of 25°S where most species spend the majority of their foraging time (DSEWPaC 2011). The entire area of the SPF is south of 24°S.

The most comprehensive compilation of data on interactions with protected species in the mid-water trawl sector of the SPF has been compiled by Tuck *et al.* (2013). They report that, between 2001 and 2011, there were 37 recorded seabird interactions with mid-water trawl gear in the SPF (Tuck *et al.* 2013). Of those, 36 occurred in the first half of 2006 and involved shearwaters; 24 flesh-footed shearwaters *Puffinus carneipes*, eight short-tailed shearwaters and four unidentified shearwater species. Of those, 22 mortalities were recorded. Both flesh-footed and short-tailed shearwaters have an ecological risk management rating of 'medium' risk.

As noted in Section 5.1, the data reported in Tuck *et al.* (2013) exclude any impacts on seabirds resulting from acoustic disturbance and behavioural changes brought about by habituation to fishing operations. The panel found no evidence to suggest that acoustic disturbance from fishing vessels was likely have an adverse impact on seabirds. However, there is evidence to suggest that discarding of fish and waste from processed fish (offal) does result in habituation to fishing operations. This issue is discussed below. While the panel acknowledges that neither fishers nor observers are able to detect or report on the extent of habituation in the same way that they report on collisions with the vessel or gear or captures of seabirds in nets, the omission of the impact of habituation from data on 'interactions' with seabirds necessarily understates the extent and impact of 'interactions' as defined for the purposes of this assessment.

In its *Small Pelagic Fishery Management Arrangements Booklet 2014-15* (AFMA 2013e), AFMA defines interactions with protected species as "any physical contact an individual (person, boat or gear) has with a protected species that causes, or may cause death, injury or stress to the species". In relation to seabirds, AFMA (2013e) provides the following example of what is and what is not an interaction:

"An interaction includes:

- where a seabird has to be assisted back into the water
- when heavy contact occurs with the boat/gear, causing the bird to be dragged underwater or to deviate from its course
- any collisions with the fishing boat, fishing gear (i.e. warps, wheel house)
- a bird gets snagged on loose or protruding wire ends (e.g. splice ends)
- a high speed collision with boat/gear
- a bird gets caught in the net or snagged on the net while attempting to feed (on 'stickers') and has to be assisted back into the water or air.

An interaction does not include:

- seabirds landing on a boat or diving into/onto a net of fish and swimming or flying off uninjured and without assistance
- where a bird is flying and has light contact with boat/gear, and the bird does not deviate from its course
- a bird floating on the water, and has light contact with boat/gear
- where a bird 'hitches a ride' on the trawl arms for a period of time and then flies away unassisted."

The panel considered that this advice is inconsistent with the definition of interactions agreed in the MoU between the Department of the Environment and AFMA (see Section 2.2.3). The MoU, correctly in the panel's view, includes "any physical contact" as an interaction, while the examples cited by AFMA exclude seabirds landing on the boat or diving into the net of fish and other forms of 'light' contact with the vessel or gear.

The definition of interactions agreed with AFMA by the Department of the Environment is narrower than that considered appropriate by the panel. In addition, in applying that definition, AFMA further constrains its interpretation by excluding certain types of interactions. As a result, the panel considered that seabird 'interactions' as interpreted by the panel, are likely to be underreported by both observers and in fishers' logbooks. The report by Tuck *et al*. (2013), which relies on these sources of information, is therefore also likely to underestimate interactions.

Nature of interactions in the SPF to date

The causes of mortality of seabirds in trawl fisheries have been summarised by the Agreement on the Conservation of Albatrosses and Petrels (ACAP) (2013a) as: "Varied and dependent on the nature of the fishery (pelagic or demersal), the species targeted and fishing area. Mortalities may be categorised into two broad types: (1) cable-related mortality, including collisions with net-monitoring cables, warp cables and paravanes; and (2) net-related mortality, which includes deaths caused by net entanglements." Tuck *et al*. (2013) provide no indication of the cause/nature of the interactions with seabirds reported in the first half of 2006 in the mid-water trawl sector of the SPF.

The panel found that there was widespread agreement that there is a strong link between the discharge of biological material from trawl vessels and seabird interactions. This is evidenced by the following:

- ACAP's assessment that: "Seabird interactions have been demonstrated to be significantly reduced by the use of mitigation measures that include protecting the warp cable, managing offal discharge and discards, and reducing the time the net is exposed on the surface of the water … In all cases the presence of offal and discards is the most important factor attracting seabirds to the stern of trawl vessels, where they are at risk of cable and net interactions. Managing offal discharge and discards while fishing gear is deployed has been shown to reduce seabird attendance" (ACAP 2013a).
- Elgin Associates (unpublished (a)), in a review of the impacts on EPBC Act protected species by large mid-water trawl vessels conducted for the panel, concluded that, in the mid-water trawl sector of the SPF, "there is a risk of incidental mortality for seabirds that follow fishing vessels and attempt to feed on discards and offal through warp strike and entanglement in trawl gear" and "the concentration of prey items during or following fishing activities is known to attract feeding seabirds. It is possible that reliance on offal or discards from fishing operations may affect breeding success".
- The South Pacific Regional Fisheries Management Organisation's (SPRFMO) Conservation and Management Measure for minimising bycatch of seabirds exempts trawl vessels that discharge no biological material from the seabird mitigation specification for trawl fishing (SPRFMO 2014a).
- The New Zealand National Plan of Action for Seabirds noted that: "Warp strikes are uncommon when fish waste and discards are not being discharged. Vessels fishing without discharging fish waste and discards therefore present less danger to seabirds of warp strike" (Ministry for Primary Industries, 2013).
- "With seabirds, the biggest risk factor is offal and fish waste" (Mr R. Wells, ResourceWise Ltd, pers. comm. 28 April 2014).
- In New Zealand, there are a variety of regulatory and non-regulatory measures to mitigate impacts on seabirds including the mandatory use of bird scaring devices (tori lines and bird bafflers) for trawl vessels greater than 28 m length overall to keep seabirds away from trawl warps through to non-regulatory VMPs, specific to each vessel, to control factors such as offal management (Mr D. Turner, Ministry for Primary Industries New Zealand *in litt*. 25 June 2014). Mr Turner noted that: "The effectiveness of these measures depends on the type of interaction. Tori lines are effective at reducing warp strikes with long-winged seabirds such as albatrosses, however there is little mitigation that can be employed to stop short-winged birds diving on or into the net to take fish during hauling. Offal management, when done well, can be very effective."
- Seabirds are attracted to trawlers due to offal, smell and history (Mr F. Drenkhahn and Mr S. Boag *in litt.* 28 October 2013).

Summary: nature and extent of interactions with seabirds in the mid-water trawl sector of the SPF

- *The SPF area is known to be important to many seabird species, and interactions with the mid-water trawl sector have occurred.*

- *Despite issues with regard to interpretation of 'interactions', the panel formed the view that the rate of interactions of SPF mid-water trawl operations with seabirds is likely to have been low.*
- *It is likely that the relatively low level of seabird interactions in the SPF can be at least partly explained by the low level of discharge of biological material that would attract seabirds.*

Management of seabird interactions in the SPF

There are no specific seabird mitigation measures in place for mid-water trawl vessels in the SPF. However, Part 13 accreditation of the SPF under the EPBC Act requires that mid-water trawl boats must have in place effective mitigation approaches and devices to minimise interactions with seabirds. AFMA enforces this by requiring the development and implementation of an approved seabird VMP. These plans are developed by AFMA in consultation with the Department of the Environment and industry. All SPF mid-water trawl operators are required to comply with and enforce them onboard. The VMP sets out individually tailored mitigation measures for the boat that minimise seabird interactions. These include requirements for physical devices to minimise interactions. The VMP may also include measures to manage the discharge of biological waste from boats to reduce seabird attraction and move-on provisions for any interactions (AFMA 2013e). The application of this policy was apparent in relation to the proposal to introduce the *FV Abel Tasman* to the SPF in 2012. The VMP that was proposed to apply to that vessel is described in Box 5.2.

Box 5.2 Proposed *FV Abel Tasman* Seabird Management Plan

Seabird Hazard Summary:

Hazard	Threat to Seabirds
Net	Entanglement on hauling and setting
Warp Wire/Net Sonde Cable	Contact through mid-air collisions Injury or drowning by warps/cables from surface Snagging on warp sprags

Boat Specific Mitigation:
The agreed mitigation actions employed by the skipper and crew of the [FV mid-water trawler]

Mitigation measures	Details
Discharge management	The holder must retain all biological material and not discharge into the water while gear is in the water
Cleaning net before deployment of fishing gear	The holder must clean the nets prior to deployment of gear removing all accessible entangled fish ("stickers")
Bird bafflers	The holder must deploy bird bafflers[15] while gear is in the water
Warp maintenance	The holder must maintain warps and remove all sprags
Warp Deflectors[16]	The holder must deploy warp deflectors from both warps while gear is in the water.

Handling Practices:
If seabirds are incidentally caught and are still alive:

- Make every reasonable effort to ensure that seabirds are released alive:
- When possible, attempt to remove seabirds from netting or meshes without jeopardizing the life of the bird; and
- Always wear gloves, long sleeves and protective eyewear when handling seabirds because they have sharp beaks and are capable of serious bites.

15 A 'bird baffler' comprises two booms attached to both stern quarters of a vessel. Two of these extend out from the sides of the vessels and two from the stern. Dropper lines are attached to the booms to create a curtain to deter seabirds from the warp-sea interface zone (ACAP, 2013a).

16 Warp deflectors (scarers) comprise weighted devices attached to each warp with clips or hooks allowing the device to slide up and down the warp freely and stay aligned with each warp, creating a protective area around the warp.

Crew Awareness:

- Crew and boat safety remains paramount. In this context and in line with this VMP, all reasonable care should be taken to minimise seabird interactions
- Ensure crew are briefed on the seabird mitigation procedures and fully understand the actions required
- Crew need to be aware of the seabird activity around the boat and report any additional observed risks to seabirds to the skipper, who will inform AFMA
- Ensure skippers are informed of any mitigation gear failures immediately so they can be addressed rapidly or of potential improvement that may increase seabird mitigation effectiveness
- Any Occupational Health and Safety issues arising from the use of seabird mitigation measures or procedures must be reported immediately to the skipper to be forwarded to AFMA.

Reporting Requirements:

- Provided an operator is fishing in accordance with your SPF Management Plan accredited under Part 13 of the EPBC Act it is not an offence to have an interaction with a protected species. However, failure to report an interaction in your daily fishing log is an offence.
- All seabirds are protected under Australian law and as such seabird interactions must be recorded in the Listed Marine and Threatened Species Form at the back of your daily fishing log and submitted to AFMA with the relevant fishing log sheets.
- Notes on the effectiveness of the mitigation devices should be recorded in the comments section of your log page.
- Try to identify seabirds that are captured. All boats should have a copy of the protected species identification guide onboard.
- If a tagged/banded seabird is captured, operators should record the band number with as many details as possible in the Listed Marine and Threatened Species Form, noting the condition in which it was released.

Source: Dr J. Findlay, AFMA, *in litt.* 19 April 2013.

5.4.3 Nature and extent of direct interactions by the DCFA with seabirds

The likely nature and extent of interactions of the DCFA with seabirds will depend on the fishing practices adopted, fishing effort, the spatial and temporal pattern of fishing and the seabird mitigation measures used. It is the panel's view that all these factors may differ under a DCFA compared to previous mid-water trawl operations in the SPF. This limits the extent to which the nature and extent of seabird interactions in these previous operations can inform an assessment of the DCFA.

Fishing practices

The information available to the panel suggested that the configuration of the gear on the vessel used in the DCFA is likely to differ markedly from that used previously on mid-water trawl vessels in the fishery. However, the fishing scenario of the DCFA (Box 2.1) excludes the disposal of biological material. This is a point of difference between the DCFA and previous mid-water trawl operations in which discarding of biological material was permitted, although only low levels of discarding are recorded (Tuck *et al.* 2013). Based on the discussion in Section 5.4.2, the panel considered that the practice of no discards of biological material is likely to have a mitigating effect on the potential for impacts on seabirds through habituation and through physical interactions by way of cable strike or net entanglement.

The potential impact of the practice of pumping fish from the net to the vessel on seabird interactions had been identified, initially, by the Department of Sustainability, Environment, Water, Population and Communities, as an uncertainty related to large mid-water trawl freezer vessels (DSEWPaC 2012b) due to the possibility that fish in the net may be available at the surface during the pumping operation and therefore attractive to seabirds. Ultimately, however, the Department reached the conclusion that: "The nature of interactions between seabirds and other types of trawl vessels is fairly well known. The department considers that based on the advice provided by Seafish about the depth at which the cod-end will be left in the water, together with the application of a seabird management plan, the impact on seabirds of large mid-water trawl freezer vessels entering the fishery may be less than for other trawl methods and therefore there is little or no uncertainty about the potential environmental impacts on seabirds." (Logan 2014). While the panel's Terms of Reference do not identify uncertainties in relation to impacts of the DCFA on seabirds in particular, the panel decided that it was appropriate to reach its own conclusions on this matter.

The panel noted that pumping has been used in previous mid-water trawl and purse seine operations in the SPF (Seafish Tasmania Pty Ltd *in litt*. 16 October 2012). The panel was advised that, during pumping, the bag and codend of the trawl net hang vertically beneath the vessel and the net is fully submerged to a depth of 50–70 m (Seafish Tasmania Pty Ltd *in litt*. 16 October 2012 and pers. comm. 23 April 2013) and that the higher pumping capacity likely to be on a vessel involved in the DCFA compared to vessels previously operated in the SPF, would reduce the time taken for the codend to be emptied.

Fishing effort

The highest annual catch taken by mid-water trawling since 2000 was nearly 9000 t in 2003 (AFMA unpublished data).

Catches in the SPF in recent years have been significantly lower than the available TACs (Table 3.1). It is claimed (for example, Mr A. Ciconte *in litt*. 15 October 2012; Ms M. Valente *in litt*. 16 October 2012; Mr F. Drenkhahn, on behalf of eight SPF SFR holders, *in litt*. 16 October 2012) that the limited range of the wet boat fleet of vessels that has fished in the SPF to date has restricted the fishery's ability to catch the available TACs in an economically efficient way. The proposal for a large-scale mid-water trawl operation was a response to this situation. The panel considered that to be economically viable, substantial proportions of the available TACs would need to be taken by the DCFA and that it is, therefore, reasonable to assume the DCFA would result in increased trawl shots and increased catches compared to those of recent years.

A significant increase in fishing effort might be expected to result in an increase in the number of interactions with seabirds. However, whether the rate of interactions with seabirds under a DCFA, would necessarily increase from the relatively low rate of the past, will depend on other factors that are discussed below.

Spatial and temporal pattern of fishing

The panel considered that the spatial and temporal pattern of fishing under a DCFA is likely to differ markedly from that of previous mid-water trawl activities in the SPF since the rationale for the introduction of a large mid-water trawl freezer vessel into the SPF relies on the ability to fish areas of the fishery that have not been previously accessible due to their distance from ports, the ability to stay at sea for longer periods and the greater capacity to fish to the available TACs.

Figure 5.20 shows that the fishery has been operating in the area of the SPF where the highest species richness of seabirds occurs, however, the abundance and distribution of birds overall is unknown. Central place foragers are more likely to be vulnerable to interactions if fishing occurs in close proximity to their rookeries. The most abundant seabird is the short-tailed shearwater, which numbers approximately 23 million, although there has been substantial decline over the past few decades (BirdLife International 2014). There are more than 280 rookeries situated on numerous, relatively inaccessible offshore islands with the largest of more than 2.8 million individuals, on Babel Island in eastern Bass Strait (Patterson *et al*. unpublished). Little penguins are also abundant with about 35,000 birds distributed throughout southern Australia and the largest colony on Gabo Island. There have been no recorded interactions with little penguins in the SPF. The population of Australasian gannets was estimated to be about 20,000 pairs in 1999–2000, with the largest colony approximately 12,000 on Black Pyramid Rocks in Tasmania (Bunce *et al*. 2002 in Patterson *et al*. unpublished). There are no recorded interactions of gannets with the SPF mid-water trawl fishery. Short-tailed shearwaters and Australasian gannets forage on the shelf for SPF species but can range approximately 200 km from their rookeries during breeding season while little penguins are restricted to around 30 km during breeding season and are less dependent on the SPF species.

These species are amongst the most numerous CPFs in the SPF but did not appear to be particularly vulnerable to direct interactions in the SPF previously. Tuck *et al*. (2013) reported 36 shearwater interactions from observed trips in 2002 and 2006, (eight of which were confirmed short-tailed shearwaters) resulting in 22 fatalities. However, while this suggests a very low mortality rate, the level of observer coverage is low therefore the real rate of interaction and mortality in the SPF may be higher. Daley *et al*. (2007b) in assessing marine birds in the ERA found: "two [of three] of the high risk bird species are large species observed in high numbers on the fishing grounds: black-browed albatross and shy albatross. No captures of these birds have been recorded in the SPF but albatross have been killed in other Commonwealth mid-water trawl fisheries through warp strikes which are a concern overseas, particularly in New Zealand and other southern hemisphere countries." They also note that the third bird species was rated high because of lack of data and that there are no sustainable mortality rates estimated for these species. Observer records for seabird interactions are very patchy throughout the SESSF and not robust enough for detailed analysis (Tuck *et al*. 2013), but in the South East Trawl component of the SESSF, the rate of seabird interactions for 2005 and 2006 was approximately 0.67 to two birds per tow (i.e. approximately between 700 and 1600 interactions per year respectively). Fifteen mortalities were recorded in 2005 and none in 2006. The panel considered that there is some uncertainty about the potential rate of direct interactions by the DCFA with seabirds but that the level of mortality would likely to be low.

Panel assessment: likely nature and extent of direct interactions by the DCFA with seabirds

- *It is likely that the rate of interactions with seabirds with mid-water trawl vessels in the SPF has been low, despite most operations having been in areas of high seabird species richness.*
- *In the context of the DCFA, the practice of pumping from the codend does not pose a specific risk to seabirds and may mitigate the risk, on a shot-by-shot basis, compared to the same practice applied by smaller vessels with reduced pumping capacity.*
- *Since it is not possible to predict with any certainty where or when the DCFA might fish or the intensity of that fishing, it is not possible to provide any firm conclusions on the likely differential impacts on seabirds that might arise from the DCFA. However, if the DCFA operated in areas or at times of the year that have not been fished previously by mid-water trawl vessels it is reasonable to expect that:*
  - *the rate of interaction might vary in comparison to previous mid-water trawl operations*
  - *the species involved in such interactions may differ from those of the past*
  - *the risk profile of those species could vary compared with those encountered in previous mid-water trawl operations.*
- *These matters constitute ongoing uncertainties associated with the operation of the DCFA.*

Seabird mitigation measures

The panel has used the VMP for seabirds that was proposed to be applied to the *FV Abel Tasman* (see Box 5.2) as a basis for consideration of the actions that could be taken by regulatory authorities to avoid, reduce and mitigate adverse environmental impacts of the DCFA on seabirds.

The panel considered the most recent advice from ACAP for reducing the impact of pelagic trawl gear on seabirds (ACAP 2013a, b, c) represents current best practice in the area of seabird mitigation. To inform its assessment of the likely effectiveness of the mitigation measures required by the VMP, the panel has compared the measures contained in the VMP to the ACAP advice (Table 5.5).

Table 5.5 Comparison of proposed Seabird VMP and ACAP best practice advice

MITIGATION MEASURES	DETAILS IN VMP	ACAP ADVICE (ACAP 2013A, B, C)	COMPARISON OF VMP AND ACAP ADVICE
Discharge management	The holder must retain all biological material and not discharge into the water while gear is in the water.	Avoid any discharge during shooting and hauling. Where possible and appropriate, convert offal into fish meal and retain all waste material with any discharge restricted to liquid discharge/sump water to reduce the number of birds attracted to a minimum. Where meal production from offal and full retention are not feasible, batching waste (preferably for two hours or longer) has been shown to reduce seabird attendance at the stern of the vessel. Mincing of waste has also been shown to reduce the attendance of large albatross species.	Proposed measures consistent with ACAP advice.
Cleaning net before deployment of fishing gear	The holder must clean the nets prior to deployment of gear removing all accessible entangled fish ('stickers').	Clean nets after every shot to remove entangled fish ('stickers') and benthic material to discourage attendance during gear shooting.	Proposed measure consistent with ACAP advice
Time net on surface		Minimise the time the net is on the water surface during hauling through proper maintenance of winches and good deck practices.	ACAP advice not relevant to the DCFA since fish to be pumped from the net to the vessel rather than the net be hauled.
Net binding		For pelagic trawl gear, apply net binding to large meshes in the wings (120–800 mm), together with a minimum 400 kg weight incorporated into the net belly prior to setting.	Measure not included in the proposed VMP and therefore inconsistent with ACAP advice.
Bird bafflers	The holder must deploy bird bafflers while gear is in the water.	Generally, bird bafflers are not regarded as providing as much protection to the warp cables as bird scaring lines or warp scares. ACAP has insufficient evidence to recommend bird bafflers, noting that there were a variety of bird baffler designs and trials would be needed to demonstrate the efficacy of a particular design.	Based on ACAP advice it is not clear that the measure proposed by the VMP would be effective unless the VMP specifies a proven bird baffler design.
Warp maintenance	The holder must maintain warps and remove all sprags.	Not specified.	Not inconsistent with ACAP advice.
Warp deflectors	The holder must deploy warp deflectors from both warps while gear is in the water.	Insufficient evidence to recommend this measure. Warp scarers have been shown to reduce contact rates but not to significant levels and were not as effective bird scaring lines.	Based on ACAP advice it is not clear that the measure proposed by the VMP would be effective.
Bird scaring line		Deploy bird scaring lines while fishing to deter birds away from warp cables and net monitoring cable. Recommended even when appropriate offal discharge and fish discard management practices in place.	Measure not included in the proposed VMP and therefore inconsistent with ACAP advice.
Snatch block		Install a snatch block at the stern of a vessel to draw the net monitoring cable close to the water to reduce its aerial extent.	Measure not included in the proposed VMP and therefore inconsistent with ACAP advice.

Panel advice: effectiveness of proposed measures and actions to avoid, reduce and mitigate adverse environmental impacts on seabirds

- *The requirements in the proposed VMP regarding discharge of biological material, the removal of stickers and warp maintenance should be consistent with or equivalent to the ACAP advice.*
- *Adopt the ACAP advice regarding net binding, bird scaring lines and the use of a snatch block noting that the use of bird scaring lines and net binding are part of the seabird VMP for Australia's winter blue grenadier fishery.*
- *If bird bafflers and warp deflectors are to be used, develop and optimise the design under scientific permit, noting that seabird captures in the SESSF have been reduced by 75 per cent using 'pinkies' (Pierre et al. 2014).*
- *Direct deck lighting inboard and keep to the minimum level necessary for the safety of the crew.*
- *Develop advice on the correct interpretation of 'interactions' with seabirds in consultation with the Department of the Environment to ensure it is consistent with the intent of the MoU between the Department and AFMA and ensure that DCFA operators and crew are familiar with this advice.*
- *Ensure that the seabird VMP for the DCFA meets the requirements of the National Recovery Plan for Threatened Albatrosses and Giant Petrels 2011–2016 (DSEWPaC 2011).*
- *If unacceptable levels of interactions with protected seabird species occur, suspend fishing immediately and adopt one of the following options:*
  - *time and area closures, noting that these will rely on knowledge of spatial and temporal uses of habitats that overlap with the fishery*
  - *trigger limits and move-on rules*
- *Consistent with the measures suggested above for pinnipeds and cetaceans, ensure 100 per cent observer coverage of all fishing activity.*

5.4.4 Monitoring and research

Given the uncertainties identified above in relation to the potential for changes in the spatial and temporal pattern of fishing under a DCFA to alter the nature and extent of past interactions with seabirds in the mid-water trawl sector of the SPF, it is imperative that full observer coverage apply to a DCFA.

The panel heard of the potential risk posed by uninitiated crews (Mr R. Wells, ResourceWise Ltd pers. comm. 28 April 2014) and the importance of education of the crew in ensuring that mitigation measures were properly implemented (e.g. Mr F. Drenkhahn and Mr S. Boag *in litt.* 28 October 2013 in Elgin Associates unpublished (a)).

Panel advice: research and monitoring to reduce uncertainties

The following proposals for monitoring and research could help to reduce uncertainties about the potential for adverse environmental impacts of the DCFA on protected seabird species:

- *Identify key ecologically sensitive seabird species, areas and times where spatial management strategies may be appropriate to mitigate direct interactions if required.*
- *Collect, analyse and publish observer data on all seabird interactions, including on the levels and causes of seabird bycatch, focusing especially on recording of warp interactions and trawl entanglement.*
- *Use electronic monitoring via video camera/s to assist in quantifying warp strikes.*
- *Ensure crews are properly trained in the use of the required seabird mitigation and on reporting requirements.*