Wild Fisheries

Establishing ecosystem-based management for the South Australian Sardine Fishery: developing ecological performance indicators and reference points to assess the need for ecological allocations

Final Report to the Fisheries Research and Development Corporation

Simon D. Goldsworthy, Brad Page, Paul Rogers
and Tim Ward

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FRDC PROJECT NO. 2005/031

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1 NON TECHNICAL SUMMARY

2005/031	**Establishing ecosystem-based management for the South Australian Sardine Fishery: developing ecological performance indicators and reference points to assess the need for ecological allocations**

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OBJECTIVES

1. To identify species of key marine predators that consume significant quantities of sardine and could potentially be used to assess the need for ecological and/or spatial allocations in the SA Sardine Fishery.
2. To identify population parameters for these key marine predators, such as measures of foraging and/or reproductive success, that are likely to be affected by changes in the distribution and abundance of sardine, and which could potentially act as ecological performance indicators for the fishery.
3. To examine the spatial and temporal scales at which these performance indicators vary in order to develop reference points that could be used to assess the need (if any) to establish ecological allocations in the fishery.
4. To use the results of this study to revise the management plan and establish cost effective systems for ongoing monitoring and assessment of the ecological effects of the SA Sardine Fishery.

OUTCOMES ACHIEVED TO DATE

This report provides information that is needed to ensure that the South Australian Sardine Fishery is managed according to the principles of Ecologically Sustainable Development (ESD). Information in the report will be used to address recommendation in the strategic assessment of the fishery, as required by the Commonwealth Department of Sustainability, Environment, Water, Population and Communities. Data from this report will provide a basis for PIRSA Fisheries to assess the suitability of establishing Ecological Performance Indicators for the fishery.

The South Australian Sardine Fishery (SASF) was established in 1991 and is Australia's largest fishery by weight. Like all South Australian fisheries, the SASF is managed according to the principles of Ecologically Sustainable Development (ESD), which means that fisheries management decisions must balance ecological, economic, social and inter-generational equity considerations. Entry is limited to 14 licence holders. There are input controls, including limitations on net size, and a Total Allowable Catch (TAC), with 14 equal Individual Transferable Quotas (ITQ) that are set annually. The formal Management Plan identifies the biological, ecological, economic and social objectives of the SASF and outlines the framework of the performance indicators, reference points and decision rules that have been established. Fishery-independent stock assessments are undertaken annually or biannually using the Daily Egg Production Method (DEPM) or an age-structured population model. The baseline TAC of 30,000 t is maintained while estimates of spawning biomass remain between the limit reference points of 150,000 and 300,000 t (exploitation rates of 10-20%).

Because data on ecosystem processes are expensive to collect and difficult to incorporate into fishery models, management typically has a single-species focus, aimed at ensuring that fish stocks provide the optimal yield. This approach is used effectively in many fisheries, including the SASF, but there is increased recognition that improved knowledge of ecosystem processes will reduce the risk to populations of predators that use the fisheries species. The SASF has been operating since the early 1990s, with most of its catch (annual catch ~30,000 t since 2006) from southern Spencer Gulf. In 2004, the SASF licence holders, the fishery managers and Australian scientists initiated a broad ecological study, which aimed to assess the impact (if any) of the fishery on the natural predators of sardines, to determine whether an explicit ecological allocation of sardines was required.

We describe the diets and habitats of several ecologically- and/or economically-important species of pelagic fishes, squids, marine mammals and seabirds, which could potentially be used to assess the need for ecological and/or spatial allocations in the SASF. We also include catch data for the SASF in our analyses, to facilitate comparisons with the consumption patterns of natural predators. Overall, the most important prey were krill, followed by sardines, anchovies, arrow squid and other crustaceans, and in total, these five prey groups accounted for 52% ± 21 (se)% of the consumption of the 37 predator groups considered. The importance of sardines to several predators highlights the need for the ongoing monitoring of ecosystem processes in this region.

The foraging ranges of five species of land-based marine predators were assessed to determine the extent of their ranges and to assess their suitability as ecological performance indicators. We estimated the distributions of foraging effort from more than 300 marine predators, including New Zealand fur seals, Australian sea lions, crested terns, little penguins and short-tailed shearwaters. Overall, sardines only made up about 1% of the total prey biomass consumed by the five apex predators, and only 2% of the total fish biomass consumed. The total estimated consumption of sardines by these predators (753 t/y), is very small (3%) relative to the current annual TACC (~30,000 t) of the SA sardine fishery. The catch of sardines by the fishery exceeds the consumption by these predators wherever fishing effort occurs, but there are also large areas where consumption of sardines by these five apex predators exceeded that of the fishery.

We provide an ecosystem perspective of the SASF, by placing its establishment and growth in the context of other dynamic changes in the ecosystem, including those from other fisheries, apex predator populations and meteorological and oceanographic change. We used the *Ecopath* with *Ecosim* software to develop a trophic mass-balance model of the eastern Great Australian Bight ecosystem off the South Australian coast, which includes continental shelf waters to 200m depth between 132° and 139.7° longitude; a region of about 154,084 km^2. We investigated the potential impacts of the sardine fishery on high tropic level predators, especially land-breeding seals and seabirds. Despite the rapid growth of the sardine fishery since 1991, sensitivity analyses, based on mixed trophic impacts, detected negligible fishery impacts on other groups, but Ecosim indicated that many of these groups were sensitive to changes in sardine biomass. This finding suggests that current levels of fishing effort are not impacting negatively on the ecosystem function. Of the land-breeding marine predators, crested terns demonstrated the greatest sensitivity to reductions in sardine biomass both in direction (negative) and magnitude, followed by Australasian gannets. Little penguins also demonstrated reductions in biomass in response to reduced sardine biomass. The trophodynamic modelling developed in this study provides the ability to resolve and attribute potential impacts from multiple fleets and environmental changes that are needed to develop and assess potential Ecological Performance Indicators for the SASF.

Despite the global interest in identification and development of ecological performance indicators to fulfil the requirements of international and regional conventions, few fisheries use ecological performance indicators to inform management decisions. In this study, we identified ecological performance indicators from natural sardine predators, such as measures of foraging and/or reproductive success, which are likely to be affected by changes in the distribution and abundance of sardines, and which could potentially act as ecological performance indicators for the fishery. The further development of these long-term monitoring datasets would provide opportunities to assess human impacts and environmental forcing on the health of this exploited ecosystem.

KEYWORDS: Pilchard, sardine, eastern Great Australian Bight ecosystem, trophodynamics.

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2 BACKGROUND: ASSESSING THE NEED FOR ECOLOGICAL ALLOCATIONS IN AUSTRALIA’S LARGEST SMALL PELAGIC FISHERY

Rogers PJ, Goldsworthy SD, Ward TM, Page B, Wiebkin A

Ecosystem-based Fisheries Management (EBFM)

Ecosystem-based fisheries management (EBFM) is based on the principle whereby exploitation of a target species is managed as part of the broader ecosystem. The key aim of EBFM is to maintain healthy ecosystems, as they ultimately provide the environmental basis for long-term sustainability of fisheries resources (Pikitch *et al.* 2004). Of importance when adopting this approach is the integration of relevant socio-economic, and ecological information by agencies chartered with management, conservation and consultation with fishery stakeholders. It is also fundamental that robust assessment methods for target-species are developed as they provide the framework of indicators for monitoring both the performance of the fishery, and ecosystem responses to management measures.

Industrial fishing is an extractive process that has the potential to have direct and in-direct ecological consequences. In Australia, legislation requires that the use of these resources is equitably shared by the broad community, indigenous peoples and future generations. Management agencies are legally and socially obligated to reduce the risks to ecosystems from which these resources are extracted. Pikitch et. al. (2004) defined some of the central tenets of EBFM as: i) avoid degradation of ecosystems, ii) minimise risk of irreversible change to species groups and ecosystems, iii) maintain socio-economic benefits, while minimising risk to ecosystem integrity; iv) generate understanding of the ecological processes and impacts of human activities; v) adopt the precautionary approach where knowledge is limited. Additional components of EBFM include the identification and management of regions that form critical foraging and breeding habitats of top predators that are listed by the International Union for Conservation of Nature (IUCN) Redlist and/or are likely to have important ‘top-down’ roles in ecosystems. Removal of top predators from marine ecosystems through bycatch, targeted fishing or competition for resources has the potential to inflict cascading impacts on lower trophic levels and fisheries (Pauly *et al.* 1998; Baum *et al.* 2003; Myers and Worm 2003; Myers *et al.* 2007; Heithaus *et al.* 2008). The complicated nature of trophic interactions means that these impacts can be difficult to quantify, yet they can be predicted using freely available ecosystem modelling tools (Jennings and Kaiser 1998; Pauly *et al.* 2000; Stevens *et al.* 2000; Kitchell *et al.* 2002).

Extension of EBFM to include consideration of the roles of top predators requires understanding and monitoring of: indicators of ecosystem health; the critical habitats that support their populations; the magnitude of variability in life history and foraging metrics. An example of the effects of removal of top predators was demonstrated by the occurrence of trophic collapses in an ecosystem where cod (*Gadus morhua*) and other large fish species were formerly dominant (Frank *et al.* 2005). In this case, declines in the abundance of these top predators were followed by 'freeing up' of productivity of benthic invertebrates and small pelagic fish species and exponential growth in a grey seal (*Halichoerus grypus*) population (Frank *et al.* 2005). This indicates that the implications of not managing fisheries at the 'ecosystem' level can be unpredictable, and manifest as changes at trophic levels with two or more 'degrees of separation'.

Global, regional and national treaties and legislation supporting adoption of EBFM in Australia

Numerous international agreements and policies support the adoption of EBFM in Australia. Scandol *et al.* (2005) provided a summary of some of the global environmental laws and treaties relating to EBFM of which Australia is a signatory and therefore obligated or expected to act upon as a member of the United Nations (UN). Aqorau (2003) provided similar information in a broader context. In short, some of the most relevant of these international laws and treaties include: i) The UN Convention on the Law of the Sea (UNCLOS), which makes reference to protection and preservation of marine ecosystems, IUCN listed threatened and endangered species and their habitats; ii) The UN Convention on Biological Diversity (CBD), which suggests that states "integrate consideration of the conservation and sustainable use of biological resources into national decision making", and develop "legislation and provisions for threatened species"; iii) The UN Food and Agriculture Organisation's (FAO, Code of Conduct for Responsible Fisheries, which proposes that UN affiliated countries have fisheries management strategies that adequately conserve and protect biodiversity, habitats and ecosystems (FAO 1995), and iv) The UN Fish Stock Agreement that is linked to UNCLOS and contains guidelines for State and Commonwealth managed fisheries. This agreement requires consideration of the integrity of fished ecosystems and the impacts of fishing on competing predatory species (Scandol *et al.* 2005). Modern management plans for Australian State and Commonwealth fisheries have strong connections to these international treaties, yet examples of adoption of measures based on the underlying principles are uncommon.

Four legislative components have been fundamental in the evolution of the EBFM principle in Australian fisheries management jurisdictions. These include the overarching principles of Ecologically Sustainable Development (National Strategy for ESD, 1992), the National ESD reporting framework for fisheries (Fletcher *et al.* 2002; Fletcher *et al.* 2005); the Commonwealth Government's *Environmental Protection and Biodiversity Conservation Act, 1999* (*EPBC Act*), and the Australian Ocean Policy. The *EPBC Act*, 1999 has reporting requirements including those under the Ecologically Sustainable Management of Fisheries guidelines, which require Australian fisheries to undergo extensive ecological assessment processes relating to impacts on ecosystems and threatened species, before their products can be approved for export. In addition to these requirements, the Commonwealth Government's Australian Ocean Policy is linked closely to the principles outlined in the National Strategy for ESD, 1992, and provides a framework of guidelines for EBFM and regional marine planning processes.

Ecological allocation and spatial limitation

Two tools that are available for adoption of EBFM measures include ecological allocation and spatial limitation. Ecological allocation refers to the proportion of the Total Allowable Commercial Catch (TACC) not allocated to fishery production, but reserved in consideration of the importance of the targeted prey to top predators. Key information required in this decision-making process includes the spawning stock biomass of the prey species, the distribution of key predator species and their foraging ranges, overlap with the historical range of the fishery, prey consumption rates, and the effects of variability in oceanographic and climatic processes on prey distribution and abundance. Spatial limitations are part of this allocation and can be defined as the portion of the potential fishery that is temporarily or permanently restricted to fishing in recognition of the importance of that area for foraging by key predators. An important ecological consideration of this approach is that systematic displacement of fishing effort to lessen the impact on one species group has the potential to cause a 'bulge effect' and have unforeseen negative impacts on other vulnerable species, ecosystem or fishery.

Case studies of ecological allocation and spatial limitations

A long-term example of the development of an EBFM approach is that of the Convention of the Conservation of Antarctic Marine Living Resources (CCAMLR), which is part of the Antarctic Treaty System. CCAMLR originated in 1977, with the aim to prevent the over-exploitation of the Antarctic krill (*Euphausia superba*) biomass. It extended to manage the potential for negative impact on top predators, including seabirds, whales and seals (Constable *et al.* 2000). Following the early development of a 'conservative approach' for the management of krill stocks in the

1980s and 1990s, CCAMLR developed precautionary decision rules that aimed to allow rates of 65–75% of the median pre-exploitation krill biomass to be allocated to the ecosystem (Hewitt *et al.* 2004). Key considerations of this process included the potential impacts on land-breeding and pelagic predators and their vastly different foraging strategies (Constable *et al.* 2000); the spatial range of the krill fisheries; their overlap with the foraging ranges predators; and that intensive fishing during periods of low prey abundance may lead to negative localised effects in important breeding and foraging areas (Constable and Nicol 2002; Hewitt *et al.* 2004).

Spatial allocation measures have been implemented in Alaska due to competition between the Steller sea lion (*Eumetopias jubatus*) that is Endangered (IUCN Redlist, 2008; US *Endangered Species Act*), the groundfish fishery for pollock (*Theragra chalcogramma*) and Atka mackerel (*Pleurogrammus monopterygius*) (Witherell et. al. 2000). These fish species constitute important prey for *E. jubatus* and the total allowable catches for the fishery were spatially allocated to reduce the risk of localised depletion and direct competition with seals. Fishing for pollock was also spatially restricted near Steller sea lion haulouts. Similarly, the targeting of capelin (*Mallotus villosus*) and krill off Alaska were prohibited in 1997 due to their high ecological importance (Witherell et. al. 2000)

The sandeel-seabird inter-relationship in the North Sea is also an interesting case-study. Sandeels (*Ammodytes matinus*) are important prey for many top predators (Camphuysen *et al.* 2006), and off the east coast of Scotland the Sandeel Fishery is the biggest single-species fishery, with a total catch of ~2.5 M tonnes (Dunne 2003). Following scientific advice from the ICES Study Group on Effects of Sandeel Fishing regarding the trophic importance of sandeels, spatial catch limits were implemented along the coast of Scotland to protect seabird breeding colonies which were heavily or totally reliant on these forage fish (Dunne 2003). This management action followed the occurrence of high profile seabird mortality events, a substantial body of scientific evidence, and pressure from media and non-government organizations. This included a major incident of mass seabird mortality due to poor prey availability and competition with the sandeel fishery in 1983; and evidence that the reproductive success of Arctic terns (*Sterna paradisaea*) (Monaghan *et al.* 1989) and black legged kittiwake (*Rissa tridactyla*) were correlated with sandeel abundance (Furness, 2002). Specifically, it was suggested that the sandeel stock must be estimated to be ~50,000 t for the kittiwake population to have a breeding success of >0.5 chicks per nest (Furness 2006). These types of parameters can be used by managers as reference points, but need to be considered as dynamic and responsive to other environmental and ecological pressures.

Oceanography and Ecology of the Great Australian Bight

The Great Australian Bight (GAB) is located on Australia's south facing coastline and is characterised by a broad continental shelf that is up to 200 km wide. It is the location of the world's only northern boundary current upwelling ecosystem (Middleton and Cirano 2002; Ward *et al.* 2006). Shelf waters of the eastern GAB (EGAB) and the interface with southern Spencer Gulf form a complex oceanographic system (Middleton and Cirano 2002). Thermal and salinity fronts form at the gulf mouth and limit exchange between the cool, low salinity Southern Ocean water masses and the warmer, higher salinity gulf waters (Bruce and Short 1990). This system is thought to play an important ecological role in this region, but the mechanisms that are responsible remain poorly resolved. The broad area around this frontal zone is where the South Australian Sardine Fishery (SASF) focuses a high proportion of its fishing effort (Ward et. al. 2008). Spencer Gulf and Gulf St Vincent are unique in the Southern Hemisphere and represent the only semi-protected, 'seasonally subtropical systems' at this otherwise temperate latitude (35° S).

During summer and autumn, shelf waters of this region are characterised by coastal upwelling that occurs between the Bonney Coast in south-eastern South Australia and the eastern GAB during summer-autumn (Kaempf *et al.* 2004), and thermoclines that form during periods of lower wind stress. These processes are coupled with the South Australian and Flinders Currents at the continental shelf margins (Middleton and Cirano 2002) and intrusion of the tropical Leeuwin Current water mass in early winter. This complex interaction of oceanographic processes supports a regionally productive marine ecosystem inhabited by a diverse suite of marine predators that have high global conservation significance and substantial economic value to local communities. This unique region supports significant levels of planktonic production during some upwelling seasons, and suitable environmental conditions for spawning, survival and growth of a diverse small pelagic fish assemblage comprising ten key species belonging to six families. These are Clupeidae, Engraulidae, Scombridae, Carangidae, Emmelichthyidae and Scomberesocidae. Members of the family Clupeidae are dominant and five species occur in this region. Small pelagic fish species found in South Australia include the Australasian sardine (*Sardinops sagax*), Australian anchovy (*Engraulis australis*), round herring (*Etrumeus teres*), sandy sprat (*Hyperlophus vittatus*), blue sprat (*Spratelloides* spp.), mackerels (*Trachurus declivis* and *T. novaezelandiae*), blue or slimy mackerel (*Scomber australasicus*), redbait (*Emmelichthys nitidus*) and saury (*Scomberesox saurus*). The presence of this small pelagic fish assemblage partly explains why this region supports: the world's most important feeding ground for juvenile southern bluefin tuna (*Thunnus maccoyii*); snapper (*Pagrus auratus*) breeding and feeding areas; the Australian salmon (*Arripis* spp.) migration; ~1.3 million pairs of short-tailed shearwaters (*Puffinus tenuirostris*); populations of white-faced storm petrels (*Pelagodroma marina*), Australasian gannet (*Morus serrator*), several albatross and tern species and little penguin (*Eudyptula minor*); >75% of the global population of Australian sea lion (*Neophoca cinerea*) and almost 80% of the Australian population of New Zealand fur seal (*Arctocephalus forsteri*); a pygmy blue whale (*Balaenoptera musculus brevicauda*) feeding and migration pathway; populations of toothed whales and several IUCN listed shark species, including gulper sharks (*Centrophorus* spp.), white shark (*Carcharodon carcharias*), shortfin mako (*Isurus oxyrinchus*), and common thresher (*Alopias vulpinus*).

The South Australian Sardine Fishery (SASF)

Overview

The SASF is the largest fishery by volume in Australia and has a total annual catch of up to 42,475 tonnes (Figure 2.1). The fishery is based near Port Lincoln, on the western side of the entrance to Spencer Gulf and adjacent to the eastern region of the GAB (Figure 2.2). This area supports the largest known Australian sardine *Sardinops sagax* spawning population (up to ~263,747 t) (Ward *et al.* 2008). Most of the catch consists of sardine however in years following two mass mortality events (1995 and 1998) anecdotal evidence suggested that Australian anchovy comprised a higher portion of catches. Sardine is mainly a ‘low price-high volume’ product with values ranging between 40– 80 c per kg for tuna fodder, $2.50 per kg for recreational fishing bait and $10 kg for human consumption. Most (94%) of the annual sardine catch is used as fodder by the southern bluefin tuna mariculture industry based in Port Lincoln. Additional markets for human consumption are also being expanded. The tuna mariculture industry is based around wild-caught southern bluefin tuna taken in the Australian Fisheries Management Authority (AFMA) managed tuna purse-seine fishery, which operates in the GAB under global quota arrangements.

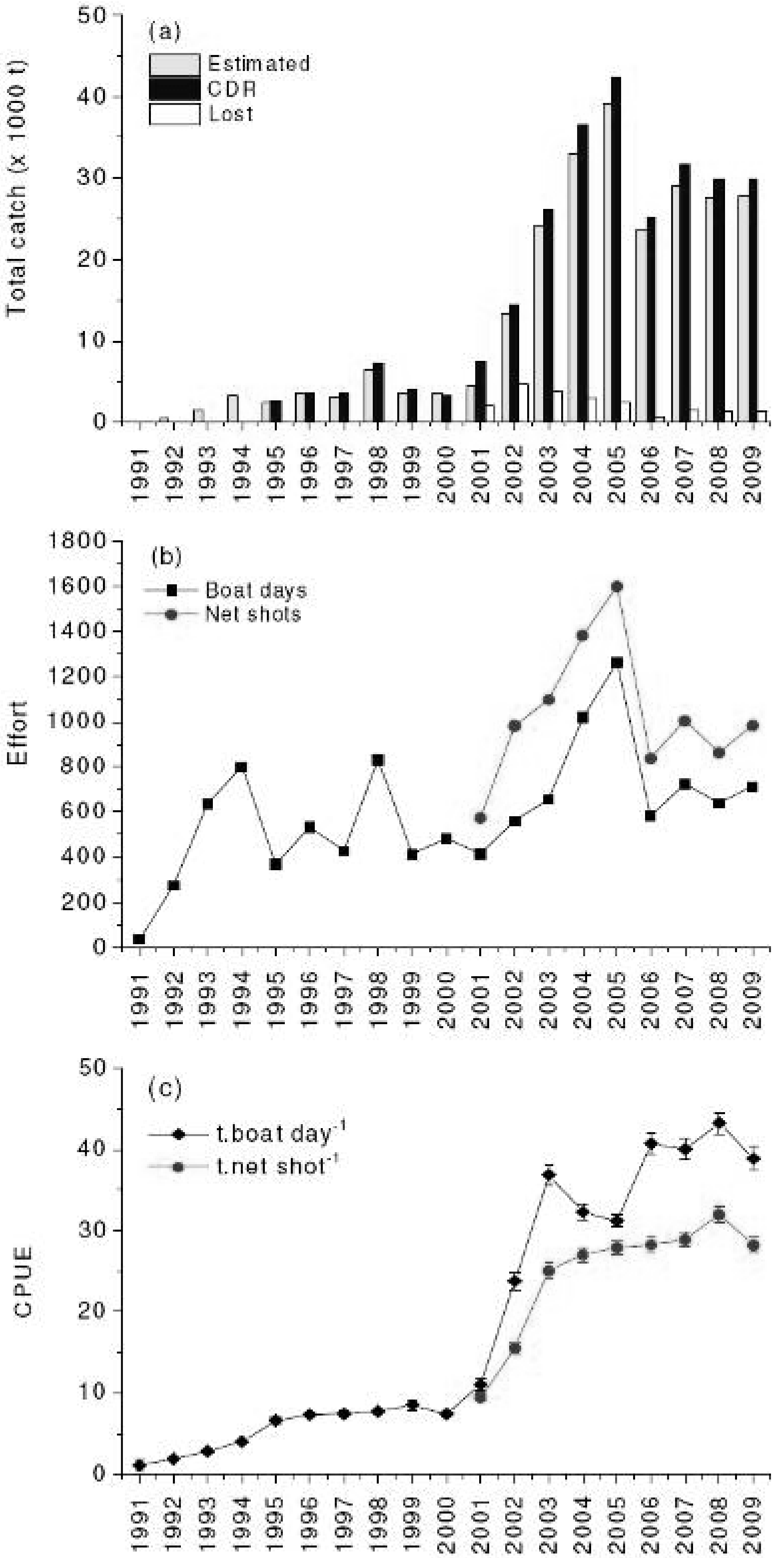

Figure 2.1. (a) Total catches of sardine from logbook records, catch disposal records (CDR) and estimated lost catches, (b) fishing effort (boat-days and net sets), and (c) mean annual catch per unit effort (tonnes per boat-days and net sets +/- SE) (Ward *et al.* 2010).

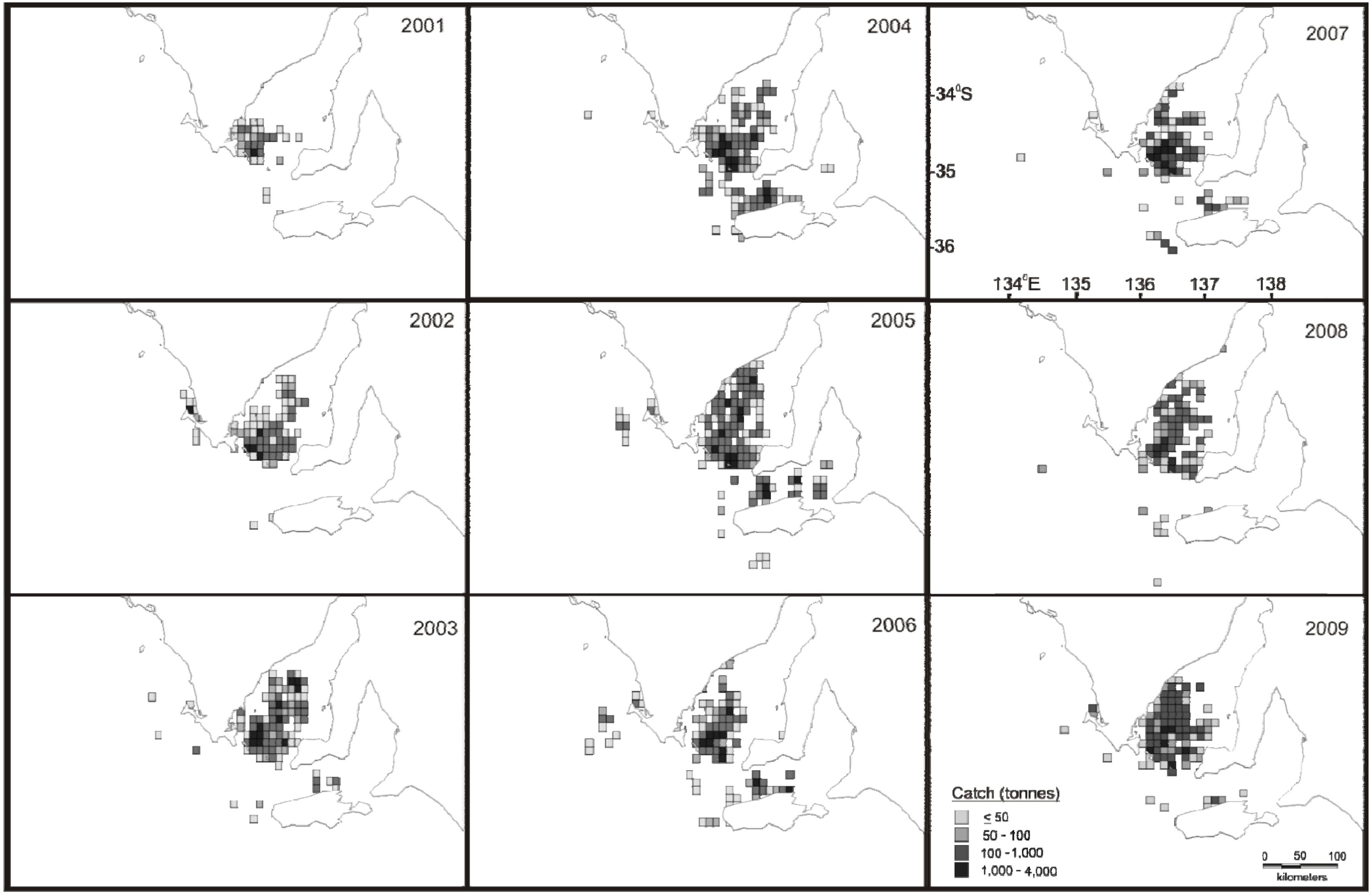

Figure 2.2. Spatial trends in annual catches of sardine between 2001 and 2009 (Ward *et al.* 2010).

Management of the SASF

The SASF is managed by the State Government's Primary Industries and Resources South Australia (PIRSA Fisheries) in accordance with the *Fisheries Management (Marine Scalefish Fisheries) Regulations* 2006 under the *Fisheries Management Act* 2007. The costs of policy development, compliance and research undertaken by SARDI Aquatic Sciences are recovered from annual licence fees. There are also several input and output controls. There are currently 14 licence holders and several fishing companies operate more than one licence. A fishery working group was established early in the development of the fishery to facilitate the consultation process between PIRSA Fisheries, SARDI researchers and licence holders, and to ensure equitable allocation of the sardine resource.

The main management performance indicator that is used to monitor the status of the sardine stock exploited by the SASF is conservative estimates of spawning biomass obtained using the Daily Egg Production Method (DEPM) (Lasker 1985). From 1999 to 2006, the TACC (Figure 2.3) for the following calendar year fluctuated widely and was set as a proportion of the spawning biomass (i.e. 10.0–17.5%, depending on the magnitude of the spawning biomass estimates) (Figure 2.4). More recently, the TACC has been set at 30,000 t and this will be maintained, while the estimate spawning biomass remain between 150,000 and 300,000 t; reflecting exploitation rates of 20 and 10%, respectively (Figure 2.3). Up to 4,000 t of additional TACC has recently been allocated to be taken outside the traditional fishing grounds of the SASF (Figure 2.3, Ward *et al.* 2010).

Since 1999, the fishery has mostly operated in southern Spencer Gulf and undergone significant economic expansion following the recovery of the spawning biomass from the second mass mortality event in 1998 (Figure 2.4, Ward *et al.* 2001 a,b) and proportional increases in TACC (Figure 2.3). During 2002, catches were mostly taken in Spencer Gulf and Investigator Strait, and a small proportion was taken in southern Gulf St Vincent and off the west coast of Kangaroo Island (Figure 2.2). In 2003 and 2004, the expansion of the TACC saw spatial expansion of the fishery from the traditional fishing grounds east and north-east of Dangerous Reef, to include more effort in Investigator Strait (Figure 2.2). During 2004 and 2005, concerns were expressed regarding increased prevalence of juvenile sardine in catches, and the possibility of localised depletion in the fished area. In 2006, the reduction in the TACC to ca. 25,000 t (Figure 2.3) saw a spatial retraction back to traditional grounds in the central region of Spencer Gulf and Investigator Strait (Figure 2.2).

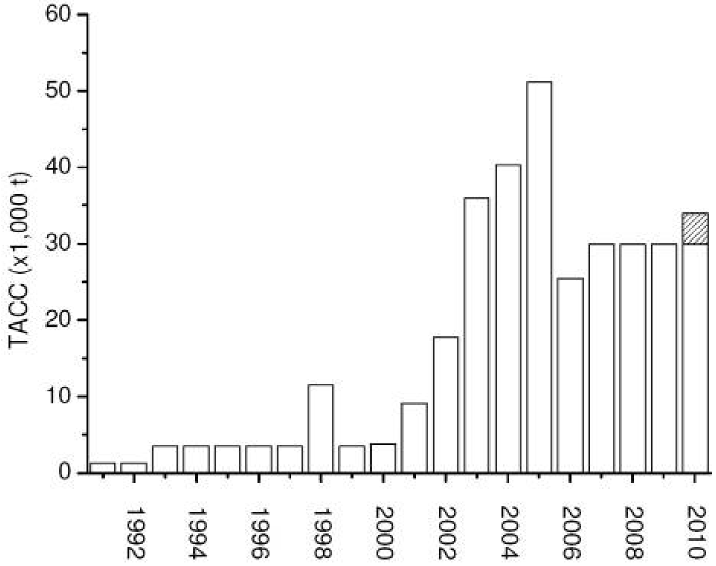

Figure 2.3. TACC for the SA sardine fishery between 1991-2009 (hatched TACC for 2010 is to be caught outside the traditional fishing area) (Ward *et al.* 2010).

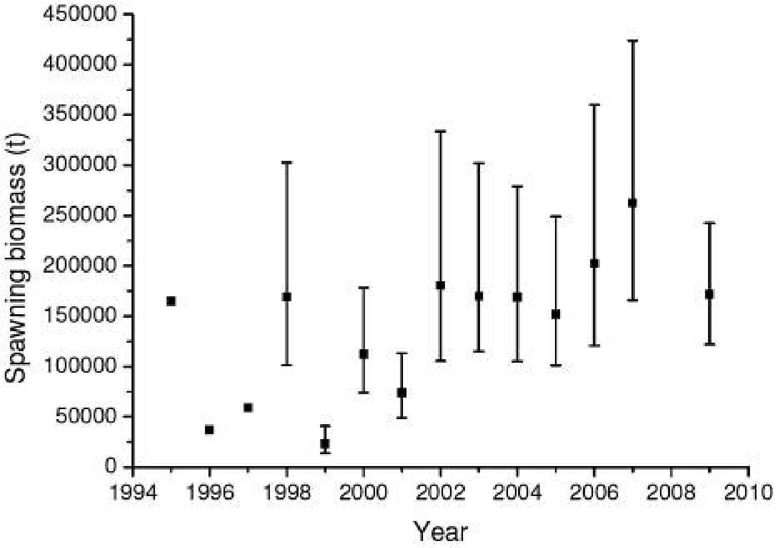

Figure 2.4. Daily Egg Production Method estimates of sardine spawning biomass in South Australian waters from 1995-2009. Error bars are 95% confidence intervals (Ward *et al.* 2010).

Need for EBFM in the SASF

The need to manage the SASF according to the principles of EBFM reflects the important ecological role of sardine and other small pelagic fishes throughout their global and Australasian distribution, the high economic value and size of the fishery, and the conservation status of many of the region's marine predators. There are also legislative requirements for all fisheries under the provisions of the *EPBC Act* 1999 to undertake strategic assessment and, if necessary, mitigate the ecological effects of fishing, including any potential trophic impacts. The role of PIRSA Fisheries is to ensure that the sardine resource is used in an ecologically sustainable and economically efficient manner on behalf of the broader community and future generations. The strategic assessment of the SASF was facilitated by PIRSA Fisheries against Australian Government ESD Guidelines for fisheries, and it identified a need to measure and minimize the impacts of the fishery on the "broader ecosystem" and "to review the current ecological management objectives, management strategies and performance indicators". This assessment led to the development of a pilot study to assess the need for ecological allocation in the SASF, which was funded by the Fisheries Research and Development Corporation in 2003 (Ward *et al.* 2005). The objectives of the pilot study were to establish suitable methods for: estimating primary and secondary production; measuring the importance of sardine in predator diets; comparing growth rates of predators in areas of high and low fishing pressure and sardine abundance; developing a preliminary trophodynamic model, and proposing a comprehensive study to assess the need for ecosystem allocation. This pilot study highlighted: the species richness and global importance of top predators in the GAB; a vast array of key knowledge gaps relating to their population demographics and ecology; and inadequacies in datasets required to properly assess the potential trophic impacts of the fishery on the broader ecosystem. This led to the development of a major trophodynamic study of the ecosystem that aimed to assess the need for ecological allocations in the SASF, and the results of which are presented in this report.

Objectives

The overall objectives of this study were:

1. To identify species of key marine predators that consumed significant quantities of sardine and could potentially be used to assess the need for ecological and/or spatial allocations in the SA Sardine Fishery.
2. To identify population parameters for these key marine predators, such as measures of foraging and/or reproductive success, that are likely to be affected by changes in the distribution and abundance of sardine, and which could potentially act as ecological performance indicators for the fishery.
3. To examine the spatial and temporal scales at which these performance indicators vary in order to develop reference points that could be used to assess the need (if any) to establish ecological allocations in the fishery.
4. To use the results of this study to revise the management plan and establish cost-effective systems for ongoing monitoring and assessment of the ecological effects of the SASF.

The approach

A multi-disciplinary approach was implemented to assess the need for an ecosystem-based approach to managing the SASF. This involved direct assessment of predator population and foraging parameters, including measures of dietary composition, foraging effort and reproductive success, as well as the use of 'state of the art' satellite tracking technologies. One of the main aims was to assess the importance of sardine and other small pelagic fish in the diets of key predators. Dietary assessments for seals and seabirds were done by analysis of regurgitates, scats, prey DNA, partially digested prey, and through reconstruction of prey via analysis of hard-parts, such as otoliths and cephalopod beaks. Prey biomass data were gleaned from a number of sources including direct stock assessment estimates (e.g. for sardine), and from relative abundance estimates based on fishery data. Estimation of foraging effort was achieved using electronic satellite tags and sensors, which were fitted to seals and seabirds. Reproductive success was monitored for the major seal and seabird colonies both within and adjacent to the fishery, via annual seal pup counts, and chick clutch size estimations.

Individual diet, foraging and fishery datasets were integrated using ecosystem modelling software packages available at www.ecopath.org, including ECOPATH and ECOSIM (Pauly *et* *al.* 2000). These models partition the ecosystem into species groups, and given sets of parameters as inputs, produce estimates of mean annual biomass, annual production and annual consumption for each group. The models also simulate changes in ecosystems at different temporal scales (ECOSIM). Diet composition, fishery data, assimilation, migration and biomass accumulation were also model inputs. Modelling scenarios were used to explain the relative ecological importance of sardine and other small pelagic fishes in the ecosystem, to consider suitable triggering actions that are responsive to changes in abundance of sardine, and to assess the need for developing a suite of tangible and cost-effective ecological reference points for the SASF. Use of these modelling tools to simulate various scenarios in upwelling ecosystems off Africa and South America has indicated that interactions between small pelagic fishes (sardine and anchovy), zooplankton and predators are important in explaining ecosystem perturbation (Shannon *et al.* 2008).

This was the first large-scale ecological study in the GAB and Spencer Gulf to attempt to understand the ecosystem dynamics, to objectively investigate the ecological value of an internationally important prey species, and to assess the need to partition the ‘harvest’ of this species fairly and responsibly among all ‘users’. Over the past decade, this has involved the collection of a vast array of baseline data on the distribution and abundance of the prey fields of key predatory species and on critical foraging and breeding areas that constitute oceanographic ‘hotspots’. It led to the development of a predictive ecosystem model for the GAB and Spencer Gulf that allows scenarios to be run to investigate the potential impacts of different environmental, ecological and fishery interactions and scenarios.

One of the critical assumptions of this study was that through dietary analysis of a suite of predators, we could glean enough information to be able to explain the importance of a particular prey species in the coastal-shelf ecosystem off South Australia. Conceptually, this seemed reasonable, but retrospectively, it was an enormous task, given the relatively short timescale of the project (initially 3 yrs, with extension), the complexity of the trophic interactions, and the inherent spatial and temporal variability associated with the distribution and abundance of different prey taxa. For example, within the relatively short timescale of the project, there have been several periods characterised by different environmental regimes (e.g. strong and weak wind regimes that either favour or don’t favour coastal upwelling, respectively), and sardines have experienced substantial inter-annual variation in their distribution and abundance in the GAB, and in the main fishing area in Spencer Gulf. Trying to decouple the environmental and human-impact related dynamism that influences these factors and the complex population level responses by predators was especially difficult.

One of the major challenges that faces researchers, fishery and natural resource managers at the beginning of studies of marine ecosystem state and functioning is that ecosystems have often already experienced extensive human impact. Therefore, the ‘comparative baseline’ or ‘unexploited state’ prior to the study is typically unobtainable. For example, some marine mega-fauna that inhabit or periodically visit South Australian waters have only recently begun to recover from over-exploitation (whales, seals and school shark). Therefore, our trophodynamic model was developed based on what was already likely to be a substantially altered system, in terms of the functioning of trophic linkages and energy transfer, predator-prey interactions, and competition for resources. Hence, implementation of spatial or ecosystem allocations would need to consider this. It would also need to be sensitive to a range of tangible biological, and socio-economic indicators, as well as identify potential flow-on impacts of displaced fishing effort. A framework for EBFM that was recently developed for the Norwegian Sea suggested that measures should also include assessment of the present and future impacts of fishery and industrial processes, and their interactions (Ottersen et. al. in press). This new plan emphasised the importance of identifying suitable indicators, while accounting for uncertainty in the degree to which the functioning of the supporting ecosystem was understood (Ottersen et. al. in press). Future development of an EBFM framework for the unique Southern Ocean ecosystem examined during our study would also need to be sensitive to the continual advances in our understanding of the state, functioning and dynamics of the oceanographic and ecological processes that ultimately support the sustainability of the SASF.

Need

Provisions of the *Commonwealth Environment and Biodiversity Conservation Act 1999* require strategic assessment and, if necessary, mitigation of the ecological effects of fishing, including trophic impacts. The strategic assessment of the South Australian Sardine Fishery identified the need to measure and minimize the impacts of the fishery on the "broader ecosystem" and "to review the current ecological management objectives, management strategies and performance indicators". However, operational ecological performance indicators and mitigating strategies have not yet been established for any pelagic fishery in Australia, and there is no agreed scientific framework for establishing these tools.

In recognition of :
1) the high profile of the SA Sardine Fishery (as Australia's largest pelagic fishery);
2) the important ecological role of sardines in the Flinders Current Ecosystem;
3) the high economic value and conservation significance of the region's marine predators;
4) and the sophisticated (single-species) stock assessment procedures and management arrangements that have been established,

members of the SA Sardine Fishery identified the need to establish "world's best practices" for managing the potential ecological impacts of the fishery. In response to this need, fishers initiated research to develop ecological performance indicators and reference points for the fishery.

Prior to this study, there was no scientific framework to assess whether the management arrangements that were established for the SA Sardine Fishery were sufficiently conservative to ensure that the fishery was managed according to the principles of ESD (i.e. that fishing does not significantly affect the status of other components of the ecosystem, Fletcher *et al.* 2002).

This project addresses the pressing need to develop a scientific framework for establishing ecological performance indicators and reference points for pelagic fisheries. The focus on the SA Sardine Fishery is appropriate, as such a large and complex undertaking could only be contemplated in a large and valuable fishery that has sophisticated stock assessment procedures and management arrangements in place, and can thus allocate significant resources to consider the establishment of an ecosystem-based management system.

Aims and objectives

1. To identify species of key marine predators that consume significant quantities of sardine and could potentially be used to assess the need for ecological and/or spatial allocations in the SA sardine Fishery;
2. To identify population parameters for these key marine predators, such as measures of foraging and/or reproductive success, that are likely to be affected by changes in the distribution and abundance of sardine, and which could potentially act as ecological performance indicators for the fishery;
3. To examine the spatial and temporal scales at which these performance indicators vary in order to develop reference points that could be used to assess the need (if any) to establish ecological allocations in the fishery;
4. To use the results of this study to revise the management plan and establish cost effective systems for ongoing monitoring and assessment of the ecological effects of the SA Sardine Fishery.

Format of the report

This report addresses each of the program objectives in distinct chapters. Chapter 3 examines the diets of the abundant marine predators in the region used by the SA sardine fishery. Chapter 4 details the foraging distributions of the abundant and tractable marine predators in the region, including crested terns, short-tailed shearwaters, little penguins, New Zealand fur seals and Australian sea lions. In Chapter 5, we use Ecopath with Ecosim software to develop a trophic mass-balance model of the ecosystem used by the SA sardine fishery. Chapter 6 identifies the species of abundant and tractable marine predators in the region that have the greatest potential as ecological performance indicators for the SA sardine fishery. Chapters 7-10 provide recommendations for further research, benefits and adoption, planned outcomes and conclusions.

The chapters of this report are presented as a series of papers that are will be submitted to journals before the FRDC report is published. As a consequence, some of the material is duplicated, particularly in the chapter introductions.

3 THE DIETS OF THE MARINE PREDATORS IN SOUTHERN AUSTRALIA: ASSESSING THE NEED FOR ECOSYSTEM-BASED MANAGEMENT OF THE SOUTH AUSTRALIAN SARDINE FISHERY

Page B, Goldsworthy SD, McLeay L, Wiebkin A, Peters K, Einoder L, Rogers P, Braley M, Gibbs S, McKenzie J, Huveneers C, Caines R, Daly K, Harrison S, Baylis A, Morrice M, Gill P, McIntosh R, Bool N, Ward T.

Introduction

Competition for resources applies evolutionary pressure on competing species resulting in at least one species diversifying its behaviour, prey resources or habitat use (MacArthur and Levins 1967). Over time, competition then becomes less intense and, on evolutionary time scales, closely related species can coexist and use similar niches (reviewed in den Boer 1986, Loveridge and Macdonald 2003). Studies of resource competition rarely indicate that coexisting species feed on completely different prey, rather co-existing species typically utilise similar prey in different proportions (e.g. Loveridge and Macdonald 2003, Gonzalez-Solis *et al.* 2000). Resource competition is therefore a key driver of the evolution of the different sizes, behaviours, ranges and prey preferences among species.

To simplify the structure and dynamics of complex ecosystems, species that are ecologically-similar are often grouped into guilds, with the assumption that resource competition among members of a guild is more intense than competition between different guilds (Root 1967). For example, in Southern Ocean ecosystems, commercial fisheries, baleen whales, cephalopods, Antarctic seals and small pelagic fishes may be part of the same guild, because they all target krill (Barlow *et al.* 2002; Murphy *et al.* 1988; Reid *et al.* 1999). In contrast, different age and sex groups of some species have been described as ecological species, because of the different resources they use (Polis 1984). Therefore, these age and sex groups may be members of different guilds, which may reduce intra-specific competition for resources (Polis 1984, reviewed in Andersson 1994, Isaac 2005). Such results and many recent studies suggest that these age and sex groups should be separated in food web models.

This study describes the diets and habitats of several ecologically- and/or economically-important species of pelagic fishes, squids, marine mammals and seabirds in southern

Australia (Table 3.1). These comprise 7 species of small pelagic fish, 7 species of large pelagic fish, 2 squid species, 5 shark species, 6 species of seabird, 2 fur seal species, 1 sea lion species, 2 dolphin species and 1 whale species. Many of the fishes and squids are commercially and/or recreationally exploited in this region, whereas the marine mammals and seabirds were exploited in the past, but their populations are now mostly recovering or stable (e.g. Ling 1999). We also include catch data for the South Australian sardine fishery in our analyses, to facilitate comparisons with the consumption patterns of natural predators.

A considerable amount of dietary and life history data have been published and/or summarised for the key marine species in southern Australia (see Table 3.1 for data summaries and details of each predator and prey species). No studies have combined these data from the region to investigate dietary relationships among the fishes, squids, seals and seabirds. The following sections summarise the diet data included in this study.

Pelagic fishes and sharks

Daly (2007) analysed the diets of 7 small pelagic fish species, most of which were collected in or near Spencer Gulf, with crustaceans, principally *Nyctiphanes australis*, being the most common prey. Caines (2005), Ward *et al.* (2006) and Page (unpublished data) summarised the diets of large pelagic fish, which had been captured throughout southern Australia over the preceding 7 years, and indicated that these predators used a broad range of pelagic prey. Fish species included in the studies were southern bluefin tuna *Thunnus maccoyii*, bonito *Sarda australis* (referred to as *Sarda australis* throughout, to distinguish it from *Sepioteuthis australis*), kingfish *Seriola lalandi*, barracouta *Thyrsites atun*, Western Australian salmon *Arripis truttacea*, snook *Sphyraena novaehollandiae*, and swallowtail *Centroberyx lineatus*. Diet data from common thresher sharks *Alopias vulpinus*, bronze whaler sharks *Carcharhinus brachyurus*, shortfin mako sharks *Isurus oxyrinchus*, blue sharks *Prionace glauca* and hammerhead sharks *Sphyrna* sp. were collected by Rogers *et al.* (unpublished data) from continental shelf and gulf waters off South Australia; they indicated the importance of large and small pelagic fish prey, southern bluefin tuna, squids and benthic cephalopods.

Squids

The diets of two squid species, which were collected in or near Spencer Gulf from trawl fishery bycatch (calamary squid *Sepioteuthis australis*, hereafter referred to as *S. australis*) or plastic lures (Gould's squid *Nototodarus gouldi*), were assessed by Roberts (2005, 2007) and Braley *et al.* (2010). *Sepioteuthis australis* and *N. gouldi* typically used fish, crustaceans and other cephalopods, but *N. gouldi* used relatively more pelagic prey such as *Nyctiphanes australis*.

Seabirds

The diets of penguins *Eudyptula minor* (Wiebkin *et al.* unpublished data), short tailed shearwaters *Puffinus tenuirostris* (Einoder *et al.* unpublished data), crested terns *Sterna bergii* (McLeay *et al.* 2009a,b) and silver gulls *Chroicocephalus novaehollandiae* (Harrison, unpublished data) were each assessed at several colonies off South Australia. The primary prey of *E. minor* was anchovy, while *P. tenuirostris* used crustaceans, including *N. australis*, before their chicks hatched and switched to fishes and cephalopods during the chick-rearing period. *Sterna bergii* typically used anchovy, sardine *Sardinops sagax* and blue sprat *Spratelloides robustus*. The diet of *C. novaehollandiae* indicated that they consumed food from pelagic, benthic and terrestrial habitats (Harrison unpublished data).

The diet of Australasian gannets *Morus serrator* was taken from Bunce (2001), who studied them in Port Phillip Bay, southeastern Australia, and indicated that they typically feed on inshore, pelagic schooling fish. The diet data for shy albatross *Thalassarche cauta* were from southeastern Australia (Hedd and Gales 2001), indicating that the most common prey were jack mackerel *Trachurus declivis* and cuttlefish *Sepia* sp.

Marine mammals

The diets of several age classes of New Zealand fur seal *Arctocephalus forsteri* and adult male Australian fur seal *Arctocephalus pusillus doriferus* were reported by Page *et al.* (2005 and unpublished data) from several colonies in southern Australia. Juvenile *A. forsteri* typically fed on Myctophidae from oceanic waters, whereas adults typically fed on Ommastrephid squids, penguins and redbait. Adult male *A. p. doriferus* typically consumed redbait and leatherjackets (Monocanthidae). The diet of female *A. p. doriferus* was not studied, because they have only recently been recorded breeding in the region (Shaughnessy *et al.* 2007). The diet of Australian sea lions *Neophoca cinerea* was assessed from 3 colonies in southern Australia whilst the diets of common dolphins *Delphinus delphis* and bottlenose dolphins *Tursiops aduncus* were assessed from individuals that were found dead in southern Australia. *Neophoca cinerea* and *T. aduncus* both primarily used *Octopus* sp., *Sepia* sp. and *S. australis* (*N. cinerea*: McIntosh *et al.* 2006, Peters *et al.* unpublished data and Page *et al.* unpublished data, *D. delphis* and *T. aduncus*: Kemper and Gibbs 2001, Gibbs *et al.* unpublished data). *Delphinus delphis* used pelagic fish such as *S. sagax*, *E. australis* and *T. declivis*. The presence of *N. australis* in the diet of blue whales *Balaenoptera musculus brevicauda* in southeastern Australia was reported by Jarman *et al.* (2002) and Gill (2002). They indicated that *N. australis* are a common prey and other crustaceans and Salpidae are also used.

SA Sardine Fishery (SASF)

The composition of the SASF catch was summarised by Shanks (2005) based on samples of catch over four years, indicating that 99% of it comprised sardines.

The SASF has operated since the early 1990s, with most catch coming from southern Spencer Gulf. The fishery uses purse seine nets and has 14 licence holders, which currently share the annual allowable catch of 30,000 t (Ward *et al.* 2008). Australian Government legislation encourages the implementation of ecosystem-based fisheries management, which requires an understanding of how fisheries compete with and affect other components of the ecosystem, rather than only quantifying impacts on target and bycatch species (Fletcher *et al.* 2002).

In 2004, the SASF licence holders, the fishery managers and Australian scientists initiated this current ecological study, which aims to assess the impact of the fishery on the natural predators of *S. sagax* to determine whether an ecological allocation of *S. sagax* is required. The objective of this project is to ensure that natural predators of *S. sagax* are not adversely affected by the fishery by: 1) compiling the diet data collected in recent years from key predators in the eastern Great Australian Bight; 2) describing the different guilds in the ecosystem used by these predators and the fishery; and 3) describing the prey species that are responsible for the differences between the different guilds. This study ultimately aims to identify species of key marine predators that consume significant quantities of sardines and could potentially be used to assess the need for ecological and/or spatial allocations in the SASF.

Materials and methods

This study used diet data that had been summarised using two indices, which estimate importance of prey taxa in different ways: 1) numerical abundance and 2) biomass (Table 3.1). Percentage numerical abundance is the proportion of the total number of prey items made up by each prey taxon. Such data were available for silver gulls, Australian sea lions, common dolphins and bottlenose dolphins. Because the importance of a prey taxon in diet studies is best represented by volumetric data (Hyslop 1980), the most appropriate assessments are based on the percentage biomass contribution. If dietary samples permit such analyses, prey mass is typically estimated based on the proportion of each prey taxon in the sample, or by using regression equations that describe relationships between otolith/beak measurements and fish/cephalopod mass. Percentage biomass data were available for the other 33 predator groups examined.

Some diet studies present summaries of the prey species of fish, cephalopods and others separately (e.g. Hume *et al.* 2004), because some researchers hypothesise that recovery rates of these different groups are not reflective of the consumption rates. This is thought to be the case for animals such as marine mammals, which may retain the remains of large prey, such as cephalopods, in their stomachs for longer than the remains of small prey, such as fish (Gales and Cheal 1992). Data that were available for *T. aduncus* and *D. delphis* had separate summaries of fish and cephalopod abundance (Table 3.1). We combined these data by assuming that fish and cephalopods contributed equally to the diets of these dolphins. These assumptions and the potential biases they introduce are discussed below.

Where possible prey were identified to species, but this was not possible for many taxa, which were grouped by genus, family or broader groups. Taxonomic information for all prey species is provided in Table 3.2. The remains of unidentifiable prey were recorded in the diets of many predator groups, which is a common problem in diet studies. We allocated these remains proportionally to the prey taxa groups that were identifiable following the methods of Page *et al.* (2005).

Most diet summaries assess differences by several factors, such as season, sex, age, breeding stage and/or location. For some species, such as *A. forsteri* and *P. tenuirostris*, some of these factors were available, so we separated them in our analyses. For *A. forsteri*, we incorporated data that split adult female, adult male and juvenile seals into different predator groups (Page *et al.* 2005). For *P. tenuirostris,* the data were split into behavioural groups, according to whether the individual had just completed a long foraging trip (>3 days duration) or short foraging trip (<3 days duration). Long foraging trips are typically conducted in waters to the south of the Australian continental shelf, whereas short trips are typically conducted over continental shelf waters (Einoder 2010). For other predator groups for which such data were not available, we used a single predator group. As a result, most predator groups are individual species, but others are classified by genus (Table 3.1). To summarise the diet data, we followed the methods of Trites *et al.* 1999, Page *et al.* 2005, Karpouzi *et al.* 2007 and Stobberup *et al.* 2009 to standardise each season, sex and/or breeding stage, so that each factor, e.g. different sexes, contributed equally to the diet for a location, and these data were subsequently standardised with other locations, so that each location contributed equally to the diet for the predator group.

Additional insights into resource competition among predator groups can be gained by classifying predator groups based on the habitat typically used by their prey. Prey were classified as benthic if they typically remain on or near the sea floor (Gomon *et al.* 2008, www.fishbase.org). Prey were classified as pelagic if they typically remain in the water column throughout the day and night, or if they migrate into the water column to feed during either the day or the night (Gomon *et al.* 2008, www.fishbase.org). Some prey were classified as terrestrial. We weighted the habitat classifications by the relative importance of each prey group in the diet and standardised the resulting value, to estimate the percentages of each prey that were benthic, pelagic and terrestrial. We then classified each predator group as either BENTHIC, PELAGIC or TERRESTRIAL, based on which category contributed the greatest percentage.

To simplify the results of our multivariate analyses, we excluded uncommon prey taxa, which were defined as groups that did not contribute at least 1% to the diet of at least one predator group. Biomass and numerical abundance data were not transformed because these data provided proportional representation and we did not want to down-weight dominant taxa. The Bray-Curtis association measure was used for the analysis based on the findings of Beals (1984). To quantify diet variation among groups of predators we used the software program called PRIMER (version 6, PRIMER-E Ltd., Plymouth, UK). The similarity profile test (SIMPROF) and CLUSTER procedures in PRIMER were used to test for evidence of structure (guilds) among predators, based on their diets and the habitats of their prey. SIMPROF provides objective rules to determine guilds, by comparing similarities in the diet/habitat data for each predator group and splitting groups of predators that have different diet/habitat profiles. The similarity profiles are analogous to the niche overlap index developed by Schoener (1968).

Means are presented as ± standard deviation and all statistical tests are two-tailed, unless stated, with the α level of statistical significance set at 0.05.

Results

In total, diet profiles were available for 37 predator groups (Table 3.1). Prey taxa included 66 fish, 12 cephalopod, 3 bird, 11 crustacean and 8 other taxa groups, representing 52%, 18%, 25%, 2% and 3% of the total consumption, respectively (Table 3.2). Overall, the most important prey were *N. australis*, followed by *S. sagax*, *E. australis*, *N. gouldi* and other crustaceans, and in total, these five prey groups accounted for 52.4% ± 21.4% of the consumption of the 37 predator groups. *Nyctiphanes australis* accounted for 100% of the diet of some predators, but was not eaten by others, with a mean of 18.3% ± 31.5%. The same was found for *Sardinops sagax*, with a mean of 12.0% ± 23.4%. *Engraulis australis* accounted for 0-75.7% of the diet of all predators, with a mean of 9.0% ± 17.6%. *Nototodarus gouldi* accounted for 0-29.9% of the diet of all predators, with a mean of 6.8% ± 9.0%. Other crustaceans accounted for 0-86.5% of the diet of all predators, with a mean of 6.3% ± 16.6%.

Table 3.1. Predator group and sample sizes compiled in this study. The source of the data and method used to summarise the data (based on biomass estimation or numerical abundance) are indicated. Data are grouped by guild (see Figure 3.1) and within guild into alphabetical order.

Predator species	**Predator common name**	**Sample size (n)**	**Summary method**	**Reference**
Alopias vulpinus	Common thresher shark	13	Biomass estimation	Rogers et al. (unpublished data)
Arripis truttacea	West Australian salmon	117	Biomass estimation	Caines (2005), Page et al. (unpublished data)
Delphinus delphis	Common dolphin	64	Numerical abundance	Kemper and Gibbs (2001), Gibbs and Kemper (unpublished data)
Eudyptula minor	Little penguin	465	Biomass estimation	Wiebkin et al. (unpublished data)
Sarda australis	Bonito	4	Biomass estimation	Caines (2005), Page et al. (unpublished data)
Morus serrator	Australasian gannet	131	Biomass estimation	Bunce (2001)
SA sardine fishery	Sardine fishery	4	Biomass estimation	Shanks (2005)
Sphyraena novaehollandiae	Snook	181	Biomass estimation	Caines (2005), Page et al. (unpublished data)
Sterna bergii	Crested tern	2938	Biomass estimation	McLeay et al. (2009)
Thunnus maccoyii	Southern blue fin tuna	94	Biomass estimation	Caines (2005), Ward et al. 2005, Page et al. (unpublished data)
Thysites atun	Barracouta	71	Biomass estimation	Caines (2005), Page et al. (unpublished data)
Arctocephalus pusillus doriferus	Australian fur seal	106	Biomass estimation	Page et al (2005)
Arctocephalus forsteri	New Zealand fur seal: juvenile	321	Biomass estimation	Page et al (2005), Page et al. (unpublished data)
Arctocephalus forsteri	New Zealand fur seal: adult female	1156	Biomass estimation	Page et al (2005), Page et al. (unpublished data)
Arctocephalus forsteri	New Zealand fur seal: adult male	648	Biomass estimation	Page et al (2005), Page et al. (unpublished data)
Nototodarus gouldi	Gould's squid	215	Biomass estimation	Roberts (2007), Braley et al. (2010)
Puffinus tenuirostris	Short-tailed shearwater: long trip	117	Biomass estimation	Einoder et al. (unpublished data)
Puffinus tenuirostris	Short-tailed shearwater: short trip	347	Biomass estimation	Einoder et al. (unpublished data)
Sepioteuthis australis	Calamary squid	87	Biomass estimation	Roberts (2005)
Seriola lalandi	Kingfish	43	Biomass estimation	Caines (2005), Page et al. (unpublished data)
Thalassarche cauta	Shy albatross	586	Biomass estimation	Hedd and Gales (2001)
Carcharhinus brachyurus	Bronze whaler	23	Biomass estimation	Rogers et al. (unpublished data)
Isurus oxyrinchus	Shortfin mako shark	12	Biomass estimation	Rogers et al. (unpublished data)
Neophoca cinerea	Australian sea lion	149	Numerical abundance	McIntosh et al. (2006), Peters et al. (unpublished data)
Prionace glauca	Blue shark	4	Biomass estimation	Rogers et al. (unpublished data)
Sphyrna sp.	Hammerhead shark	16	Biomass estimation	Rogers et al. (unpublished data)
Tursiops aduncus	Bottlenose dolphin	51	Numerical abundance	Kemper and Gibbs (2001), Gibbs and Kemper (unpublished data)
Chroicocephalus novaehollandiae	Silver gull	108	Numerical abundance	Harrison (unpublished data)
Balaenoptera musculus	Blue whale	1	Biomass estimation	Jarman et al. (2002), Morrice and Gill (unpublished data)
Centroberyx lineatus	Swallowtail	174	Biomass estimation	Caines (2005), Page et al. (unpublished data)
Emmelichthys nitidus	Redbait	54	Biomass estimation	Daly (2007)
Engraulis australis	Anchovy	15	Biomass estimation	Daly (2007)
Etrumeus teres	Maray	12	Biomass estimation	Daly (2007)
Sardinops sagax	Sardine	218	Biomass estimation	Daly (2007)
Scomber australasicus	Blue mackerel	16	Biomass estimation	Daly (2007)
Spratelloides robustus	Blue sprat	17	Biomass estimation	Daly (2007)
Trachurus declivis	Jack mackerel	40	Biomass estimation	Daly (2007)

Table 3.2. Percent contribution of prey taxa to the diet of each predator. Guilds of predators and habitats of prey taxa (benthic (B), pelagic (P) or terrestrial (T) in the prey taxa column) are also indicated.

				Predator guild and taxa										
				1	1	1	1	1	1	1	1	1	1	1
Prey type	Prey family/group	Prey common name	Prey taxa	*T. atun*	*Sarda australis*	*A. vulpinus*	*S. bergii*	*M. serrator*	*E. minor*	*A. truttacea*	Sardine fishery	*S. novaehollandiae*	*T. maccoyii*	*D. delphis*
Fish	Apogonidae	Cardinal fish	*Vincentia* sp^{B}	0	0	0	0.1	0	0	0	0	0	0	0
		Wood's siphon fish	*Siphaemia cephalotes*^{B}	0	0	0	0	0	0	0	0	0	0	0
	Arripidae	Herring and salmon	*Arripis georgianus & A. truttacea*^{P}	0	0	7.2	2.5	2	8.9	2.4	0	0	0.7	1.7
	Atherinidae	Silversides	Atherinidae^{P}	0	0	0	0.3	0	1.3	31.6	0	0	0	0
	Berycidae	Swallowtail	*C. lineatus*^{B}	0	0	0	0	0	0	0	0	0	0.1	0
	Callorhinchidae	Elephant fish	*Callorhinchus milii*^{B}	0	0	0	0.1	0	0	0	0	0	0	0
	Carangidae	Jack and yellowtail mackerel	*Trachurus* sp.^{P}	7.2	0	0	0.2	0	0	0	0	0	3.2	11.4
		Trevally	*Pseudocaranx* sp.^{P}	0	0	0	0	0.8	0	0	0	0	0	6.4
	Clupeidae	Sardine	*Sardinops sagax*^{P}	42.8	100	17.1	22.9	16	3.5	9.5	99	12.7	37.4	20.8
		Sprats	Clupeidae^{P}	0	0	0	7.5	0.8	2.8	0	0	0	3	0.6
		Maray	*Etrumeus teres*^{P}	0	0	0	1.4	0	0	0	0.2	3.6	0	0
	Cyttidae	Silver dory	*Cyttus australis*^{P}	0	0	0	0	0	0	0	0	0	0	0
	Dasyatidae	Stingray	Dasyatidae^{B}	0	0	0	0	0	0	0	0	0	0	0
	Dinolestidae	Long-finned pike	*Dinolestes lewini*^{P}	0	0	0	0	0	0	0	0	0	0	0
	Diodontidae	Porcupine fish	Diodontidae^{B}	0	0	0	0.1	0	0	0	0	0	0	0
	Elasmobranchii	Shark - other	Elasmobranchii^{B}	0	0	0	0	0	0	0	0	0	0	0
	Emmelichthyidae	Redbait	*Emmelichthys nitidus*^{P}	18.2	0	0	0	0	4.4	4.1	0	0	12.1	0
	Engraulidae	Anchovy	*Engraulis australis*^{P}	7.5	0	75.7	30.7	9	61.6	18.6	0.8	15.4	21.4	43.3
	Gempylidae	Barracouta	*Thyrsites atun*^{P}	0	0	0	5.7	37.6	0	0	0	0	0	0
		Western gemfish	*Rexea solandri*^{P}	0	0	0	0	0	0	0	0	0	0	0
	Gerreidae	Silverbelly	*Parequula melbournensis*^{B}	0	0	0	0.1	0	0	0.9	0	0	0	0.2
	Gobiidae	Goby	Gobiidae^{B}	0	0	0	0	0	0	0	0	0	0	2.1
	Gonorynchidae	Beaked salmon	*Gonorynchus greyi*^{P}	0	0	0	0.1	0	0	0	0	0	0	0
	Hemiramphidae	Southern sea garfish	*Hyporhamphus melanochir*^{B}	0	0	0	4.9	10	10.8	4.8	0	0	0	0.9
	Labridae	Wrasse - other	Labridae^{B}	0	0	0	0	3	0	4.3	0	0	0	0
		Blue throat wrasse	*Notolabrus tetricus*^{B}	0	0	0	0	0	0	0	0	0	0	0
	Latridae	Trumpeter	*Latris lineata*^{B}	0	0	0	0	0	0	0	0	0	0	0.7
	Microstomatidae	Microstomatid	*Nansenia macrolepis*^{P}	0	0	0	0	0	0	0	0	0	0	0
	Monocanthidae	Leatherjacket - other	Monocanthidae^{B}	0	0	0	0	0	0.5	0	0	0	0	0
		Degan's leatherjacket	*Thamnaconus degeni*^{B}	0	0	0	12.9	0	0	0	0	0	0	0
		Velvet Leatherjacket	*Meuschenia scaber*^{B}	0	0	0	0	0	0	0	0	0	0	0
		Bridled leatherjacket	*Acanthaluteres spilomelanurus*^{B}	0	0	0	0.1	0	0	0	0	0	0	0
	Moridae	Red rock cod	*Pseudophycis bachus*^{B}	0	0	0	0	0	0.1	0	0	0	0	0
	Mugilidae	Yellow-eye mullet	*Aldrichetta forsteri*^{P}	0	0	0	0	0.8	0	0	0	0	0	0
	Mullidae	Goatfish	*Upeneichthys* sp.^{B}	0	0	0	0.1	5	0	0	0	0	0	0

Table 3.2. (cont.)

				Predator guild and taxa										
				1	1	1	1	1	1	1	1	1	1	1
Prey type	Prey family/group	Prey common name	Prey taxa	*T. atun*	*Sarda australis*	*A. vulpinus*	*S. bergii*	*M. serrator*	*E. minor*	*A. truttacea*	Sardine fishery	*S. novaehollandiae*	*T. maccoyii*	*D. delphis*
Fish (cont.)	Myctophidae	Lanternfish	*Symbolophorus* sp.^{P}	0	0	0	0	0	0	0	0	0	0	0
		Lanternfish	Myctophidae^{P}	0	0	0	0	0	0	0	0	0	0	0
		Belted lanternfish	*Electrona paucirastra* ^{P}	0	0	0	0	0	0	0	0	0	0	0
		Jensen's lanternfish	*Diaphus jenseni* ^{P}	0	0	0	0	0	0	0	0	0	0	0
		Carlsberg's lanternfish,	*Electrona carlsbergi* ^{P}	0	0	0	0	0	0	0	0	0	0	0
		Lanternfish	*Krefftichthys anderssoni* ^{P}	0	0	0	0	0	0	0	0	0	0	0
		Lanternfish	*Gymnoscopelus robustus* ^{P}	0	0	0	0	0	0	0	0	0	0	0
	Neosebastidae	Gurnard perch	Neosebastidae^{B}	0	0	0	0	0	0	0	0	0	0	0
	Notocheiridae	Surf sardine	*Iso rhothophilus* ^{P}	0	0	0	0	0	0	0	0	0	0	0
	Odacidae	Weed whiting	Odacidae^{B}	0	0	0	5.4	0	0	0	0	0	0	0
	Oreosomatidae	Smooth oreo	*Pseudocyttus maculatus* ^{P}	0	0	0	0	0	0	0	0	0	0	0
	Pempherididae	Bullseye	*Pempheris* sp.^{B}	0	0	0	3.1	0	0.7	21.4	0	58.5	0	0
	Platycephalidae	Flathead	*Neoplatycephalus* sp^{B}	0	0	0	0	0	0	0	0	0	0	0
		Tiger flathead	*Neoplatycephalus richardsoni* ^{B}	0	0	0	0	0	0	0	0	0	0	0
	Scomberesocidae	Saury	*Scomberesox saurus* ^{P}	0	0	0	0	0	1	0	0	0	3	0
	Scombridae	Blue Mackerel	*Scomber australasicus* ^{P}	6.1	0	0	0.4	11	0.6	0	0	0	0.7	0.3
		Southern bluefin tuna	*Thunnus maccoyii* ^{P}	0	0	0	0	0	0	0	0	0	0	0
	Scorpaenidae	Scorpionfish	Scorpaenidae^{B}	0	0	0	0.1	0	0	0	0	0	0	0
	Scorpididae	Sweep	*Scorpis aequipinnis* ^{P}	0	0	0	0	0	0	0	0	0	0	0
	Seriolella	Spotted warehou	*Seriolella punctata* ^{P}	0	0	0	0	0	0	0	0	0	0	0
	Serranidae	Perch	Serranidae^{B}	0	0	0	0	0	0	0	0	0	0	1.4
	Sillaginidae	Eastern school whiting	*Sillago flindersi* ^{B}	0	0	0	0	0	0	0	0	0	0	0
		Western school whiting	*Sillago bassensis* ^{B}	0	0	0	0.1	0	0	0	0	0	0	0
		King George whiting	*Sillaginodes puncata* ^{B}	0	0	0	0	0	0	0	0	0	0	0
	Sparidae	Snapper	*Chrysophrys auratus* ^{B}	0	0	0	0	0	0	0	0	0	0	0.2
	Sygnathidae	Pipefish	Sygnathidae^{B}	0	0	0	0.1	0	0.5	0	0	0	0	0
		Sea horse	Sygnathidae^{B}	0	0	0	0.1	0	0.1	0	0	0	0	0
	Terapontidae	Western striped trumpeter	*Pelates octolineatus* ^{B}	0	0	0	0.1	0	0	0	0	0	0	0
	Tetraodontidae	Toads	Tetraodontidae^{B}	0	0	0	0	0	0	0	0	0	0	0
	Trachichthyidae	Roughy	Trachichthyidae^{P}	0	0	0	0	0	0	0	0	0	0	0
	Unknown fish eggs	Unknown fish eggs	Unknown fish eggs^{P}	0	0	0	0	0	0	0	0	0	0	0
	Uranoscopidae	Stargazer	Uranoscopidae^{B}	0	0	0	0	0	0	0	0	0	0	0

Table 3.2. (cont.)

Prey type	Prey family/group	Prey common name	Prey taxa	Predator guild and taxa										
				1	1	1	1	1	1	1	1	1	1	1
				T. atun	*Sarda australis*	*A. vulpinus*	*S. bergii*	*M. serrator*	*E. minor*	*A. truttacea*	Sardine fishery	*S. novaehollandiae*	*T. maccoyii*	*D. delphis*
Cephalopod	Argonautidae	Southern argonaut	*Argonauta nodosa* ^{P}	0	0	0	0	0	0	0	0	0	0	0
	Histioteuthidae	Jewel squid	*Histioteuthis* sp.^{P}	0	0	0	0	0	0	0	0	0	0	0
	Loliginidae	Calamary squid	*Sepioteuthis australis* ^{P}	0	0	0	0.4	2	0	0	0	0	0	6
	Octopodidae	Octopus - other	Octopodidae^{B}	0	0	0	0	0	0	0	0	0	0	0.3
		Maori octopus	*Octopus maorum* ^{B}	0	0	0	0	0	0	0	0	0	0	0
		Southern keeled octopus	*Octopus berrima* ^{B}	0	0	0	0	0	0	0	0	0	0	0
	Ommastrephidae	Gould's squid	*Nototodarus gouldi* ^{P}	18.2	0	0	0.2	2	2.7	2.4	0	8	18.3	1.2
		Arrow squids	Ommastrephidae^{P}	0	0	0	0	0	0	0	0	0	0	0
		Southern Ocean arrow squid	*Todarodes filippovae* ^{P}	0	0	0	0	0	0	0	0	0	0	0
	Sepiidae	Giant cuttlefish	*Sepia apama* ^{B}	0	0	0	0	0	0	0	0	0	0	0
		Cuttlefish - other	*Sepia* sp.^{B}	0	0	0	0	0	0	0	0	0	0.1	2.5
	Sepiolidae	Southern dumpling squid	*Euprymna tasmanica* ^{B}	0	0	0	0	0	0	0	0	0	0	0
Crustacean	Amphipoda	Amphipod	Amphipoda^{P}	0	0	0	0	0	0	0	0	0	0	0
	Crustacean - other	Crustacean - other	Crustacean^{P}	0	0	0	0	0	0.5	0	0	0	0	0
	Decapoda	Decapod	Decapoda^{P}	0	0	0	0	0	0	0	0	0	0	0
	Euphausiidae	Krill	*Nyctiphanes australis* ^{P}	0	0	0	0	0	0	0	0	0.9	0	0
	Isopoda	Isopod	Isopoda^{P}	0	0	0	0	0	0	0	0	0	0	0
	Maxillopoda	Copepod	Maxillopoda^{P}	0	0	0	0	0	0	0	0	0.9	0	0
	Ostracoda	Ostracod	Ostracoda^{P}	0	0	0	0	0	0	0	0	0	0	0
	Palinuridae	Southern rock lobster	*Jasus edwardsii* ^{B}	0	0	0	0	0	0	0	0	0	0	0
	Pandalidae	Pandalid prawn	Pandalidae^{B}	0	0	0	0	0	0	0	0	0	0	0
	Penaeidae	Western king prawn	*Melicertus latisulcatus* ^{B}	0	0	0	0.3	0	0	0	0	0	0	0
		Southern velvet shrimp	*Metapenaeopsis palmensis* ^{B}	0	0	0	0	0	0	0	0	0	0	0
Bird	Procellariidae	Shearwater	*Puffinus* sp.^{P}	0	0	0	0	0	0	0	0	0	0	0
	Seabird - other	Seabird - other	Seabird - other^{P}	0	0	0	0	0	0	0	0	0	0	0
	Spheniscidae	Little penguin	*Eudyptula minor* ^{P}	0	0	0	0	0	0	0	0	0	0	0
Other	Donacidae	Cockle	*Plebidonax deltoides* ^{B}	0	0	0	0	0	0	0	0	0	0	0
	Insecta	Insect	Insecta^{T}	0	0	0	0	0	0	0	0	0	0	0
	Mollusca	Mullusc	Mollusca^{B}	0	0	0	0	0	0	0	0	0	0	0
	Nereididae	Nereid Worm	Nereididae^{P}	0	0	0	0	0	0	0	0	0	0	0
	Plant - marine	Plant - marine	Plant - marine^{B}	0	0	0	0	0	0	0	0	0	0	0
	Plant - terrestrial	Plant - terrestrial	Plant - terrestrial^{T}	0	0	0	0	0	0	0	0	0	0	0
	Polychaeta	Worm	Polychaeta^{P}	0	0	0	0	0	0	0	0	0	0	0
	Salpidae	Salp	Salpidae^{P}	0	0	0	0	0	0	0	0	0	0	0

Table 3.2. (cont.)

				Predator guild and taxa									
				2	3	4	4	5	5	5	5	5	5
Prey type	Prey family/group	Prey common name	Prey taxa	*A. pusillus*	*A. forsteri* -JUV	*A. forsteri* -AF	*A. forsteri* -AM	*N. gouldi*	*S. australis*	*S. lalandi*	*T. cauta*	*P. tenuirostris* -LT	*P. tenuirostris* -ST
Fish	Apogonidae	Cardinal fish	*Vincentia* sp^{B}	0	0	0	0	0	0	0	0	0	0
		Wood's siphon fish	*Siphaemia cephalotes*^{B}	0	0	0	0.4	0	0	0	0	0	0
	Arripidae	Herring and salmon	*Arripis georgianus* & *A. truttacea*^{P}	0	1.7	0.1	2.2	0.8	10.5	7.4	0	0.1	0.1
	Atherinidae	Silversides	Atherinidae^{P}	0	0	0	0	0	0	0	0	0	0
	Berycidae	Swallowtail	*C. lineatus*^{B}	1.1	0	5.2	2.8	0	0	0	0	0	0
	Callorhinchidae	Elephant fish	*Callorhinchus milii*^{B}	0	0	0	0	0	0	0	0	0	0
	Carangidae	Jack and yellowtail mackerel	*Trachurus* sp.^{P}	5.4	6.6	4.3	7.7	10	39.5	7.1	46.8	27.1	10.4
		Trevally	*Pseudocaranx* sp.^{P}	0	0	0	0	0	0	15.5	0	0	0
	Clupeidae	Sardine	*Sardinops sagax*^{P}	0	2.1	0.9	1	11.8	0	5.3	0	0.9	1.1
		Sprats	Clupeidae^{P}	0	0.1	0	0	0	0	0	0	0.4	0.5
		Maray	*Etrumeus teres*^{P}	0	0	0	0	2.5	0	0	0	0	0
	Cyttidae	Silver dory	*Cyttus australis*^{P}	0	0	0	0.1	0	0	0	0	0	0
	Dasyatidae	Stingray	Dasyatidae^{B}	0	0	0	0	0	0	0	0	0	0
	Dinolestidae	Long-finned pike	*Dinolestes lewini*^{P}	0	0.1	0.1	0	0	0	0	0	0	0
	Diodontidae	Porcupine fish	Diodontidae^{B}	0	0	0	0	0	0	0	0	0	0
	Elasmobranchii	Shark - other	Elasmobranchii^{B}	0	0	0	0	0	0	0	0	0	0
	Emmelichthyidae	Redbait	*Emmelichthys nitidus*^{P}	44.8	1.8	23.3	12.3	4.7	0	8.7	2.9	3.9	4.8
	Engraulidae	Anchovy	*Engraulis australis*^{P}	0	1.3	0.2	0.5	3.6	11.2	0	0	19.4	13.7
	Gempylidae	Barracouta	*Thyrsites atun*^{P}	0	0.8	1.7	2	2.5	0	0	8	1.2	1.5
		Western gemfish	*Rexea solandri*^{P}	0	2.2	0.8	1.7	3.1	0	0	0	0	0
	Gerreidae	Silverbelly	*Parequula melbournensis*^{B}	0.3	0.3	0.4	0.1	0.3	0	0	0	0	0
	Gobiidae	Goby	Gobiidae^{B}	0	0	0	0	0	0	0	0	0	0
	Gonorynchidae	Beaked salmon	*Gonorynchus greyi*^{P}	0	0	0	0	0	0	0	0	0	0
	Hemiramphidae	Southern sea garfish	*Hyporhamphus melanochir*^{B}	0	0.9	0.4	0.3	0.3	2.7	0	0	0.2	0.1
	Labridae	Wrasse - other	Labridae^{B}	0	0	0	0	0	0	0	0	0	0
		Blue throat wrasse	*Notolabrus tetricus*^{B}	0	0	0	0	0	0	0	0	0	0
	Latridae	Trumpeter	*Latris lineata*^{B}	0	0	0	0	0	0	0	0	0	0
	Microstomatidae	Microstomatid	*Nansenia macrolepis*^{P}	0	0	0.1	0.1	0	0	0	0	0	0
	Monocanthidae	Leatherjacket - other	Monocanthidae^{B}	36.7	1.4	8.3	8	0	1.4	0	0	0	0
		Degan's leatherjacket	*Thamnaconus degeni*^{B}	0	0	0	0	0	0	0	0	0	0
		Velvet Leatherjacket	*Meuschenia scaber*^{B}	0	0	0	0	0	0	0	0	0	0
		Bridled leatherjacket	*Acanthaluteres spilomelanurus*^{B}	0	0	0	0	0	0	0	0	0	0
	Moridae	Red rock cod	*Pseudophycis bachus*^{B}	2.6	0.2	0.1	2	0.3	0	0	6.6	0	0
	Mugilidae	Yellow-eye mullet	*Aldrichetta forsteri*^{P}	0	0.1	0	0	0	0	0	0	0	0
	Mullidae	Goatfish	*Upeneichthys* sp.^{B}	0.5	0	0	0.1	0.7	0	0	0	0	0

Table 3.2. (cont.)

				Predator guild and taxa									
				2	3	4	4	5	5	5	5	5	5
Prey type	Prey family/group	Prey common name	Prey taxa	*A. pusillus*	*A. forsteri* -JUV	*A. forsteri* -AF	*A. forsteri* -AM	*N. gouldi*	*S. australis*	*S. lalandi*	*T. cauta*	*P. tenuirostris* -LT	*P. tenuirostris* -ST
Fish (cont.)	Myctophidae	Lanternfish	*Symbolophorus* sp.^{P}	0.1	10.9	0.9	0.4	0	0	0	0	0	0
		Lanternfish	Myctophidae^{P}	0	0	0	0	0	0	0	0	4.8	0.8
		Belted lanternfish	*Electrona paucirastra* ^{P}	0	0	0	0	0	0	0	0	4	0.6
		Jensen's lanternfish	*Diaphus jenseni* ^{P}	0	0	0	0	0	0	0	0	3.7	0.6
		Carlsberg's lanternfish,	*Electrona carlsbergi* ^{P}	0	0	0	0	0	0	0	0	3.5	0.6
		Lanternfish	*Krefftichthys anderssoni* ^{P}	0	0	0	0	0	0	0	0	2.7	0.4
		Lanternfish	*Gymnoscopelus robustus* ^{P}	0	1.1	0.2	0.2	0	0	0	0	0	0
	Neosebastidae	Gurnard perch	Neosebastidae^{B}	0	0	0	0	0	0	0	0	0	0
	Notocheiridae	Surf sardine	*Iso rhothophilus* ^{P}	0	0	0	0	0.7	0	0	0	0	0
	Odacidae	Weed whiting	Odacidae^{B}	0	0	0	0	0	0	0	0	0	0
	Oreosomatidae	Smooth oreo	*Pseudocyttus maculatus* ^{P}	0	0	0	0	0	0	0	0	0	0
	Pempherididae	Bullseye	*Pempheris* sp.^{B}	0	16.7	0.7	0.2	0	14.3	0	0	0	0
	Platycephalidae	Flathead	*Neoplatycephalus* sp^{B}	0	0	1.2	0	0	9.9	0	0	0	0
		Tiger flathead	*Neoplatycephalus richardsoni* ^{B}	2	0	0.1	0.2	0	0	0	0	0	0
	Scomberesocidae	Saury	*Scomberesox saurus* ^{P}	0	0	0.1	0	0	0	0	0	0	0
	Scombridae	Blue Mackerel	*Scomber australasicus* ^{P}	0	0	0.1	0.1	0.2	0	8.9	0	0.4	0.6
		Southern bluefin tuna	*Thunnus maccoyii* ^{P}	0	0	0	0	0	0	0	0	0	0
	Scorpaenidae	Scorpionfish	Scorpaenidae^{B}	0	0	0	0	0	0	0	0	0	0
	Scorpididae	Sweep	*Scorpis aequipinnis* ^{P}	0	0	0	0	0	0	0	0	0	0
	Seriolella	Spotted warehou	*Seriolella punctata* ^{P}	0	0	0.1	0	0	0	0	0	0	0
	Serranidae	Perch	Serranidae^{B}	0	0	0	0	0	0	0	0	0	0
	Sillaginidae	Eastern school whiting	*Sillago flindersi* ^{B}	0.3	0	0.4	0.1	0	0	0	0	0	0
		Western school whiting	*Sillago bassensis* ^{B}	0	0.2	0.2	0.3	0	0	0	0	0	0
		King George whiting	*Sillaginodes puncata* ^{B}	0	0	0	0	0	0	0	0	0	0
	Sparidae	Snapper	*Chrysophrys auratus* ^{B}	0	0	0	0	0	0	0	0	0	0
	Sygnathidae	Pipefish	Sygnathidae^{B}	0	0	0	0	0	0	0	0	0	0
		Sea horse	Sygnathidae^{B}	0	0	0	0	0	0	0	0	0	0
	Terapontidae	Western striped trumpeter	*Pelates octolineatus* ^{B}	0	0	0	0	0	0	0	0	0	0
	Tetraodontidae	Toads	Tetraodontidae^{B}	0	0	0	0	0	0	0	0	0	0
	Trachichthyidae	Roughy	Trachichthyidae^{P}	0	0	0	0	0.8	0	0	0	0	0
	Unknown fish eggs	Unknown fish eggs	Unknown fish eggs^{P}	0	0	0	0	0	0	0	0	0	0
	Uranoscopidae	Stargazer	Uranoscopidae^{B}	0.7	0.1	0	0.1	0	0	0	0	0	0

Table 3.2. (cont.)

				Predator guild and taxa									
				2	3	4	4	5	5	5	5	5	5
Prey type	Prey family/group	Prey common name	Prey taxa	*A. pusillus*	*A. forsteri* -JUV	*A. forsteri* -AF	*A. forsteri* -AM	*N. gouldi*	*S. australis*	*S. lalandi*	*T. cauta*	*P. tenuirostris* -LT	*P. tenuirostris* -ST
Cephalopod	Argonautidae	Southern argonaut	*Argonauta nodosa* ^{P}	0	0	0	0	0	0	0	0.4	0	0
	Histioteuthidae	Jewel squid	*Histioteuthis* sp.^{P}	0.2	0	0.1	0	0	0	0	0	0	0
	Loliginidae	Calamary squid	*Sepioteuthis australis* ^{P}	0	0.3	0.4	0.2	1.1	0	0	0	0	0
	Octopodidae	Octopus - other	Octopodidae^{B}	0.5	0	0	0.1	0	2.1	0	0	0	0
		Maori octopus	*Octopus maorum* ^{B}	0	0	0	0.1	0	0	0	0	0	0
		Southern keeled octopus	*Octopus berrima* ^{B}	0	0	0	0	0	0	0	0	0	0
	Ommastrephidae	Gould's squid	*Nototodarus gouldi* ^{P}	0.3	14	15.9	11.6	21.4	0	26.9	8.1	11.6	6.2
		Arrow squids	Ommastrephidae^{P}	4.3	22.7	12.2	10.9	0	0	0	0	1.7	0.6
		Southern Ocean arrow squid	*Todarodes filippovae* ^{P}	0	2.3	4.1	1.5	0	0	0	0	0	0
	Sepiidae	Giant cuttlefish	*Sepia apama* ^{B}	0	0	0	0.1	0	0	0	26.7	0	0
		Cuttlefish - other	*Sepia* sp.^{B}	0.2	0	0	0	0.9	0	4.6	0	0	0
	Sepiolidae	Southern dumpling squid	*Euprymna tasmanica* ^{B}	0	0	0	0	0	0	0	0	0	0
Crustacean	Amphipoda	Amphipod	Amphipoda^{P}	0	0	0	0	3.7	0	0	0	0.3	12.9
	Crustacean - other	Crustacean - other	Crustacean^{P}	0	0	0	0	0.4	2.4	0	0	1.8	4.7
	Decapoda	Decapod	Decapoda^{P}	0	0	0	0	0	2.4	0	0	0	0
	Euphausiidae	Krill	*Nyctiphanes australis* ^{P}	0	0	0	0	25.8	0	15.6	0.1	11	39.2
	Isopoda	Isopod	Isopoda^{P}	0	0	0	0	0	0	0	0	1.2	0
	Maxillopoda	Copepod	Maxillopoda^{P}	0	0	0	0	0	0	0	0	0.1	0.3
	Ostracoda	Ostracod	Ostracoda^{P}	0	0	0	0	0	0	0	0	0	0
	Palinuridae	Southern rock lobster	*Jasus edwardsii* ^{B}	0	0	0	0	0	0	0	0	0	0
	Pandalidae	Pandalid prawn	Pandalidae^{B}	0	0	0	0	0.4	0	0	0	0	0
	Penaeidae	Western king prawn	*Melicertus latisulcatus* ^{B}	0	0	0	0	0	3.6	0	0	0	0
		Southern velvet shrimp	*Metapenaeopsis palmensis* ^{B}	0	0	0.1	0	0	0	0	0	0	0
Bird	Procellariidae	Shearwater	*Puffinus* sp.^{P}	0	0.3	3.6	2.4	0	0	0	0	0	0
	Seabird - other	Seabird - other	Seabird - other^{P}	0	0	0	0.1	0	0	0	0	0	0
	Spheniscidae	Little penguin	*Eudyptula minor* ^{P}	0	11.8	13.7	30.1	0	0	0	0	0	0
Other	Donacidae	Cockle	*Plebidonax deltoides* ^{B}	0	0	0	0	0	0	0	0	0	0
	Insecta	Insect	Insecta^{T}	0	0	0	0	0	0	0	0	0	0.3
	Mollusca	Mullusc	Mollusca^{B}	0	0	0	0	0.1	0	0	0	0	0
	Nereididae	Nereid Worm	Nereididae^{P}	0	0	0	0	3.9	0	0	0	0	0
	Plant - marine	Plant - marine	Plant - marine^{B}	0	0	0	0	0	0	0	0	0	0
	Plant - terrestrial	Plant - terrestrial	Plant - terrestrial^{T}	0	0	0	0	0	0	0	0	0	0
	Polychaeta	Worm	Polychaeta^{P}	0	0	0	0	0	0	0	0	0	0
	Salpidae	Salp	Salpidae^{P}	0	0	0	0	0	0	0	0.4	0	0

Table 3.2. (cont.)

				Predator guild and taxa						
				6	6	6	6	6	6	7
Prey type	Prey family/group	Prey common name	Prey taxa	*N. cinerea*	*P. glauca*	*C. brachyurus*	*phyrna* sp.	*I. oxyrinchus*	*T. aduncus*	*C. novaehollandiae*
Fish	Apogonidae	Cardinal fish	*Vincentia* sp^{B}	0.1	0	0	0	0	20.7	0
		Wood's siphon fish	*Siphaemia cephalotes*^{B}	0	0	0	0	0	0	0
	Arripidae	Herring and salmon	*Arripis georgianus* & *A. truttacea*^{P}	0.2	0	0	0	0	0.1	0.1
	Atherinidae	Silversides	Atherinidae^{P}	0	0	0	0	0	0	0
	Berycidae	Swallowtail	*C. lineatus*^{B}	1.6	0	0	0.2	0	0.1	0
	Callorhinchidae	Elephant fish	*Callorhinchus milii*^{B}	0	0	0	0	0	0	0
	Carangidae	Jack and yellowtail mackerel	*Trachurus* sp.^{P}	0.8	0	0	1.9	0.1	1.5	0
		Trevally	*Pseudocaranx* sp.^{P}	0	0	0	0	0	0.9	0
	Clupeidae	Sardine	*Sardinops sagax*^{P}	0.1	0	35.4	0	0	2.1	2.8
		Sprats	Clupeidae^{P}	0	0	0	0	0	0	0.1
		Maray	*Etrumeus teres*^{P}	0	0	0	0	0	0	0
	Cyttidae	Silver dory	*Cyttus australis*^{P}	0	0	0	0	0	0	0
	Dasyatidae	Stingray	Dasyatidae^{B}	0.4	0	1.1	0	0	0	0
	Dinolestidae	Long-finned pike	*Dinolestes lewini*^{P}	0	0	0	0	0	0	0
	Diodontidae	Porcupine fish	Diodontidae^{B}	0	0	0	0	0	0	0
	Elasmobranchii	Shark - other	Elasmobranchii^{B}	0.2	0	2.2	0	0	0	0
	Emmelichthyidae	Redbait	*Emmelichthys nitidus*^{P}	0	0	0	0	0	0	0
	Engraulidae	Anchovy	*Engraulis australis*^{P}	0	0	0.3	0.3	0	0	0
	Gempylidae	Barracouta	*Thyrsites atun*^{P}	0.1	0	0	0	0	0	0
		Western gemfish	*Rexea solandri*^{P}	0	0	0	0	0	0	0
	Gerreidae	Silverbelly	*Parequula melbournensis*^{B}	1.6	0	0	0	0	1.4	1
	Gobiidae	Goby	Gobiidae^{B}	0	0	0	0	0	1.5	0
	Gonorynchidae	Beaked salmon	*Gonorynchus greyi*^{P}	0	0	0	0	0	0	0
	Hemiramphidae	Southern sea garfish	*Hyporhamphus melanochir*^{B}	0	0	0	0	0	0	0.3
	Labridae	Wrasse - other	Labridae^{B}	0.8	0	0.3	0	0	0	14.7
		Blue throat wrasse	*Notolabrus tetricus*^{B}	3.8	0	0	0	0	0	0
	Latridae	Trumpeter	*Latris lineata*^{B}	0	0	0	0	0	0.2	0
	Microstomatidae	Microstomatid	*Nansenia macrolepis*^{P}	0	0	0	0	0	0	0
	Monocanthidae	Leatherjacket - other	Monocanthidae^{B}	2.4	0	0	0	0	0	0
		Degan's leatherjacket	*Thamnaconus degeni*^{B}	0	0	0.1	0	0	0	29.1
		Velvet Leatherjacket	*Meuschenia scaber*^{B}	3.7	0	0	0	0	0	0
		Bridled leatherjacket	*Acanthaluteres spilomelanurus*^{B}	0	0	0	0	0	0	0
	Moridae	Red rock cod	*Pseudophycis bachus*^{B}	0.8	0	0	0	0	0	0
	Mugilidae	Yellow-eye mullet	*Aldrichetta forsteri*^{P}	0	0	0	0	0	0	0
	Mullidae	Goatfish	*Upeneichthys* sp.^{B}	1.5	0	0	0	0	0	1.2

Table 3.2. (cont.)

				Predator guild and taxa						
				6	6	6	6	6	6	7
Prey type	Prey family/group	Prey common name	Prey taxa	*N. cinerea*	*P. glauca*	*C. brachyurus*	*ohyrna* sp.	*I. oxyrinchus*	*T. aduncus*	*C. novaehollandiae*
Fish (cont.)	Myctophidae	Lanternfish	*Symbolophorus* sp.^{P}	0	0	0	0	0	0	0
		Lanternfish	Myctophidae^{P}	0	0	0	0	0	0	0
		Belted lanternfish	*Electrona paucirastra* ^{P}	0	0	0	0	0	0	0
		Jensen's lanternfish	*Diaphus jenseni* ^{P}	0	0	0	0	0	0	0
		Carlsberg's lanternfish,	*Electrona carlsbergi* ^{P}	0	0	0	0	0	0	0
		Lanternfish	*Krefftichthys anderssoni* ^{P}	0	0	0	0	0	0	0
		Lanternfish	*Gymnoscopelus robustus* ^{P}	0	0	0	0	0	0	0
	Neosebastidae	Gurnard perch	Neosebastidae^{B}	2.1	0	0	0	0	0	0
	Notocheiridae	Surf sardine	*Iso rhothophilus* ^{P}	0	0	0	0	0	0	0
	Odacidae	Weed whiting	Odacidae^{B}	0.1	0	0	0	0	0.2	0
	Oreosomatidae	Smooth oreo	*Pseudocyttus maculatus* ^{P}	0	0	0	0	0	0	0
	Pempherididae	Bullseye	*Pempheris* sp.^{B}	0.2	0	0	0	0	1.2	0.2
	Platycephalidae	Flathead	*Neoplatycephalus* sp^{B}	0.8	0	0	0	0	0.2	1.3
		Tiger flathead	*Neoplatycephalus richardsoni* ^{B}	2.7	0	0	0	0	0	0
	Scomberesocidae	Saury	*Scomberesox saurus* ^{P}	0	0	0	0	0	0	0
	Scombridae	Blue Mackerel	*Scomber australasicus* ^{P}	0	28.4	0.1	0	0	0	0
		Southern bluefin tuna	*Thunnus maccoyii* ^{P}	0	0	0	7.8	74.1	0	0
	Scorpaenidae	Scorpionfish	Scorpaenidae^{B}	0	0	0	0	0	0.3	0
	Scorpididae	Sweep	*Scorpis aequipinnis* ^{P}	0	0	0	0	0	0.7	0
	Seriolella	Spotted warehou	*Seriolella punctata* ^{P}	0	0	0	0	0	0	0
	Serranidae	Perch	Serranidae^{B}	1.5	0	0	0	0	1.9	0
	Sillaginidae	Eastern school whiting	*Sillago flindersi* ^{B}	0.4	0	0	0	0	0	0
		Western school whiting	*Sillago bassensis* ^{B}	0.2	0	0	0	0	0	0
		King George whiting	*Sillaginodes puncata* ^{B}	0.1	0	0	0	0	0.7	0
	Sparidae	Snapper	*Chrysophrys auratus* ^{B}	0	0	0	0	0	0	0
	Sygnathidae	Pipefish	Sygnathidae^{B}	0	0	0	0	0	0	0
		Sea horse	Sygnathidae^{B}	0	0	0	0.1	0	0.2	0
	Terapontidae	Western striped trumpeter	*Pelates octolineatus* ^{B}	0	0	0	0	0	4.1	0
	Tetraodontidae	Toads	Tetraodontidae^{B}	0	0	0	0	0	0	0
	Trachichthyidae	Roughy	Trachichthyidae^{P}	0	0	0	0	0	0	0
	Unknown fish eggs	Unknown fish eggs	Unknown fish eggs^{P}	0	0	0	0	0	0	0
	Uranoscopidae	Stargazer	Uranoscopidae^{B}	0	0	0	0	0	0	0

Table 3.2. (cont.)

				Predator guild and taxa						
				6	6	6	6	6	6	7
Prey type	Prey family/group	Prey common name	Prey taxa	*N. cinerea*	*P. glauca*	*C. brachyurus*	*phyrna* sp.	*I. oxyrinchus*	*T. aduncus*	*C. novaehollandiae*
Cephalopod	Argonautidae	Southern argonaut	*Argonauta nodosa* ^{P}	0	0	0	0	0	0	0
	Histioteuthidae	Jewel squid	*Histioteuthis* sp.^{P}	0	0	0	0	0	0	0
	Loliginidae	Calamary squid	*Sepioteuthis australis* ^{P}	14.8	0	29.5	39.5	0	13.4	0.1
	Octopodidae	Octopus - other	Octopodidae^B	23.4	0	8.1	0	0	28.5	0.5
		Maori octopus	*Octopus maorum* ^{B}	4.3	11.9	0	0	0	0	0
		Southern keeled octopus	*Octopus berrima* ^{B}	2.5	0	0	0	0	0	0
	Ommastrephidae	Gould's squid	*Nototodarus gouldi* ^{P}	1.2	29.9	9.4	24.6	15.9	0	0.2
		Arrow squids	Ommastrephidae^P	3.5	0	0	0	0	0	0
		Southern Ocean arrow squid	*Todarodes filippovae* ^{P}	0	17.9	5.4	3.5	9.9	0	0
	Sepiidae	Giant cuttlefish	*Sepia apama* ^{B}	15.3	11.9	0	12.7	0	3.6	0
		Cuttlefish - other	*Sepia* sp.^{B}	3.2	0	8.1	8.5	0	16.5	2.3
	Sepiolidae	Southern dumpling squid	*Euprymna tasmanica* ^{B}	0.6	0	0	0	0	0	0
Crustacean	Amphipoda	Amphipod	Amphipoda^P	0	0	0	0	0	0	0
	Crustacean - other	Crustacean - other	Crustacean^P	2.2	0	0	0	0	0	5
	Decapoda	Decapod	Decapoda^P	0	0	0	0.9	0	0	0
	Euphausiidae	Krill	*Nyctiphanes australis* ^{P}	0	0	0	0	0	0	0
	Isopoda	Isopod	Isopoda^P	0	0	0	0	0	0	0
	Maxillopoda	Copepod	Maxillopoda^P	0	0	0	0	0	0	0
	Ostracoda	Ostracod	Ostracoda^P	0	0	0	0	0	0	0
	Palinuridae	Southern rock lobster	*Jasus edwardsii* ^{B}	2.1	0	0	0	0	0	0
	Pandalidae	Pandalid prawn	Pandalidae^B	0	0	0	0	0	0	0
	Penaeidae	Western king prawn	*Melicertus latisulcatus* ^{B}	0.4	0	0	0	0	0	0
		Southern velvet shrimp	*Metapenaeopsis palmensis* ^{B}	0	0	0	0	0	0	0
Bird	Procellariidae	Shearwater	*Puffinus* sp.^{P}	0	0	0	0	0	0	0
	Seabird - other	Seabird - other	Seabird - other^P	0	0	0	0	0	0	1.3
	Spheniscidae	Little penguin	*Eudyptula minor* ^{P}	0.3	0	0	0	0	0	0
Other	Donacidae	Cockle	*Plebidonax deltoides* ^{B}	0	0	0	0	0	0	1.7
	Insecta	Insect	Insecta^T	0	0	0	0	0	0	5.2
	Mollusca	Mullusc	Mollusca^B	0	0	0	0	0	0	3.5
	Nereididae	Nereid Worm	Nereididae^P	0	0	0	0	0	0	0
	Plant - marine	Plant - marine	Plant - marine^B	0	0	0	0	0	0	9.4
	Plant - terrestrial	Plant - terrestrial	Plant - terrestrial^T	0	0	0	0	0	0	20
	Polychaeta	Worm	Polychaeta^P	0	0	0	0	0	0	0
	Salpidae	Salp	Salpidae^P	0	0	0	0	0	0	0

Table 3.2. (cont.)

				Predator guild and taxa								
				8	8	8	8	8	8	8	8	8
Prey type	Prey family/group	Prey common name	Prey taxa	*E. australis*	*S. australasicus*	*B. musculus*	*T. declivis*	*E. teres*	*E. nitidus*	*S. sagax*	*S. robustus*	*C. lineatus*
Fish	Apogonidae	Cardinal fish	*Vincentia* sp^{B}	0	0	0	0	0	0	0	0	0
		Wood's siphon fish	*Siphaemia cephalotes*^{B}	0	0	0	0	0	0	0	0	0
	Arripidae	Herring and salmon	*Arripis georgianus* & *A. truttacea*^{P}	0	0	0	0	0	0	0	0	0
	Atherinidae	Silversides	Atherinidae^{P}	0	0	0	0	0	0	0	0	0
	Berycidae	Swallowtail	*C. lineatus*^{B}	0	0	0	0	0	0	0	0	0
	Callorhinchidae	Elephant fish	*Callorhinchus milii*^{B}	0	0	0	0	0	0	0	0	0
	Carangidae	Jack and yellowtail mackerel	*Trachurus* sp.^{P}	0	0	0	0	0	0	0	0	0
		Trevally	*Pseudocaranx* sp.^{P}	0	0	0	0	0	0	0	0	0
	Clupeidae	Sardine	*Sardinops sagax*^{P}	0	0	0	0	0	0	0	0	0
		Sprats	Clupeidae^{P}	0	0	0	0	0	0	0	0	0
		Maray	*Etrumeus teres*^{P}	0	0	0	0	0	0	0	0	0
	Cyttidae	Silver dory	*Cyttus australis*^{P}	0	0	0	0	0	0	0	0	0
	Dasyatidae	Stingray	Dasyatidae^{B}	0	0	0	0	0	0	0	0	0
	Dinolestidae	Long-finned pike	*Dinolestes lewini*^{P}	0	0	0	0	0	0	0	0	0
	Diodontidae	Porcupine fish	Diodontidae^{B}	0	0	0	0	0	0	0	0	0
	Elasmobranchii	Shark - other	Elasmobranchii^{B}	0	0	0	0	0	0	0	0	0
	Emmelichthyidae	Redbait	*Emmelichthys nitidus*^{P}	0	0	0	0	0	0	0	0	0
	Engraulidae	Anchovy	*Engraulis australis*^{P}	0	0	0	0	0	0	0	0	0
	Gempylidae	Barracouta	*Thyrsites atun*^{P}	0	0	0	0	0	0	0	0	0
		Western gemfish	*Rexea solandri*^{P}	0	0	0	0	0	0	0	0	0
	Gerreidae	Silverbelly	*Parequula melbournensis*^{B}	0	0	0	0	0	0	0	0	0
	Gobiidae	Goby	Gobiidae^{B}	0	0	0	0	0	0	0	0	0
	Gonorynchidae	Beaked salmon	*Gonorynchus greyi*^{P}	0	0	0	0	0	0	0	0	0
	Hemiramphidae	Southern sea garfish	*Hyporhamphus melanochir*^{B}	0	0	0	0	0	0	0	0	0
	Labridae	Wrasse - other	Labridae^{B}	0	0	0	0	0	0	0	0	0
		Blue throat wrasse	*Notolabrus tetricus*^{B}	0	0	0	0	0	0	0	0	0
	Latridae	Trumpeter	*Latris lineata*^{B}	0	0	0	0	0	0	0	0	0
	Microstomatidae	Microstomatid	*Nansenia macrolepis*^{P}	0	0	0	0	0	0	0	0	0
	Monocanthidae	Leatherjacket - other	Monocanthidae^{B}	0	0	0	0	0	0	0	0	0
		Degan's leatherjacket	*Thamnaconus degeni*^{B}	0	0	0	0	0	0	0	0	0
		Velvet Leatherjacket	*Meuschenia scaber*^{B}	0	0	0	0	0	0	0	0	0
		Bridled leatherjacket	*Acanthaluteres spilomelanurus*^{B}	0	0	0	0	0	0	0	0	0
	Moridae	Red rock cod	*Pseudophycis bachus*^{B}	0	0	0	0	0	0	0	0	0
	Mugilidae	Yellow-eye mullet	*Aldrichetta forsteri*^{P}	0	0	0	0	0	0	0	0	0
	Mullidae	Goatfish	*Upeneichthys* sp.^{B}	0	0	0	0	0	0	0	0	0

Table 3.2. (cont.)

				Predator guild and taxa								
				8	8	8	8	8	8	8	8	8
Prey type	Prey family/group	Prey common name	Prey taxa	*E. australis*	*S. australasicus*	*B. musculus*	*T. declivis*	*E. teres*	*E. nitidus*	*S. sagax*	*S. robustus*	*C. lineatus*
Fish (cont.)	Myctophidae	Lanternfish	*Symbolophorus* sp.^{P}	0	0	0	0	0	0	0	0	0
		Lanternfish	Myctophidae^{P}	0	0	0	0	0	0	0	0	0
		Belted lanternfish	*Electrona paucirastra* ^{P}	0	0	0	0	0	0	0	0	0
		Jensen's lanternfish	*Diaphus jenseni* ^{P}	0	0	0	0	0	0	0	0	0
		Carlsberg's lanternfish,	*Electrona carlsbergi* ^{P}	0	0	0	0	0	0	0	0	0
		Lanternfish	*Krefftichthys anderssoni* ^{P}	0	0	0	0	0	0	0	0	0
		Lanternfish	*Gymnoscopelus robustus* ^{P}	0	0	0	0	0	0	0	0	0
	Neosebastidae	Gurnard perch	Neosebastidae^{B}	0	0	0	0	0	0	0	0	0
	Notocheiridae	Surf sardine	*Iso rhothophilus* ^{P}	0	0	0	0	0	0	0	0	0
	Odacidae	Weed whiting	Odacidae^{B}	0	0	0	0	0	0	0	0	0
	Oreosomatidae	Smooth oreo	*Pseudocyttus maculatus* ^{P}	0	0	0	0	0	0	0	0	0
	Pempherididae	Bullseye	*Pempheris* sp.^{B}	0	0	0	0	0	0	0	0	0
	Platycephalidae	Flathead	*Neoplatycephalus* sp^{B}	0	0	0	0	0	0	0	0	0
		Tiger flathead	*Neoplatycephalus richardsoni* ^{B}	0	0	0	0	0	0	0	0	0
	Scomberesocidae	Saury	*Scomberesox saurus* ^{P}	0	0	0	0	0	0	0	0	0
	Scombridae	Blue Mackerel	*Scomber australasicus* ^{P}	0	0	0	0	0	0	0	0	0
		Southern bluefin tuna	*Thunnus maccoyii* ^{P}	0	0	0	0	0	0	0	0	0
	Scorpaenidae	Scorpionfish	Scorpaenidae^{B}	0	0	0	0	0	0	0	0	0
	Scorpididae	Sweep	*Scorpis aequipinnis* ^{P}	0	0	0	0	0	0	0	0	0
	Seriolella	Spotted warehou	*Seriolella punctata* ^{P}	0	0	0	0	0	0	0	0	0
	Serranidae	Perch	Serranidae^{B}	0	0	0	0	0	0	0	0	0
	Sillaginidae	Eastern school whiting	*Sillago flindersi* ^{B}	0	0	0	0	0	0	0	0	0
		Western school whiting	*Sillago bassensis* ^{B}	0	0	0	0	0	0	0	0	0
		King George whiting	*Sillaginodes puncata* ^{B}	0	0	0	0	0	0	0	0	0
	Sparidae	Snapper	*Chrysophrys auratus* ^{B}	0	0	0	0	0	0	0	0	0
	Sygnathidae	Pipefish	Sygnathidae^{B}	0	0	0	0	0	0	0	0	0
		Sea horse	Sygnathidae^{B}	0	0	0	0	0	0	0	0	0
	Terapontidae	Western striped trumpeter	*Pelates octolineatus* ^{B}	0	0	0	0	0	0	0	0	0
	Tetraodontidae	Toads	Tetraodontidae^{B}	0	0	0	0	0	0	0	0	0
	Trachichthyidae	Roughy	Trachichthyidae^{P}	0	0	0	0	0	0	0	0	0
	Unknown fish eggs	Unknown fish eggs	Unknown fish eggs^{P}	6.8	18.8	0	2.4	1.9	0.4	1.6	0	0
	Uranoscopidae	Stargazer	Uranoscopidae^{B}	0	0	0	0	0	0	0	0	0

Table 3.2. (cont.)

				Predator guild and taxa								
				8	8	8	8	8	8	8	8	8
Prey type	Prey family/group	Prey common name	Prey taxa	*E. australis*	*S. australasicus*	*B. musculus*	*T. declivis*	*E. teres*	*E. nitidus*	*S. sagax*	*S. robustus*	*C. lineatus*
Cephalopod	Argonautidae	Southern argonaut	*Argonauta nodosa* ^{P}	0	0	0	0	0	0	0	0	0
	Histioteuthidae	Jewel squid	*Histioteuthis* sp.^{P}	0	0	0	0	0	0	0	0	0
	Loliginidae	Calamary squid	*Sepioteuthis australis* ^{P}	0	0	0	0	0	0	0	0	0
	Octopodidae	Octopus - other	Octopodidae^{B}	0	0	0	0	0	0	0	0	0
		Maori octopus	*Octopus maorum* ^{B}	0	0	0	0	0	0	0	0	0
		Southern keeled octopus	*Octopus berrima* ^{B}	0	0	0	0	0	0	0	0	0
	Ommastrephidae	Gould's squid	*Nototodarus gouldi* ^{P}	0	0	0	0	0	0	0	0	0
		Arrow squids	Ommastrephidae^{P}	0	0	0	0	0	0	0	0	0
		Southern Ocean arrow squid	*Todarodes filippovae* ^{P}	0	0	0	0	0	0	0	0	0
	Sepiidae	Giant cuttlefish	*Sepia apama* ^{B}	0	0	0	0	0	0	0	0	0
		Cuttlefish - other	*Sepia* sp.^{B}	0	0	0	0	0	0	0	0	0
	Sepiolidae	Southern dumpling squid	*Euprymna tasmanica* ^{B}	0	0	0	0	0	0	0	0	0
Crustacean	Amphipoda	Amphipod	Amphipoda^{P}	0	0	0	0	0	0	0	0	0
	Crustacean - other	Crustacean - other	Crustacean^{P}	47.2	14.9	0	11.7	5.5	20.2	28.4	86.5	0
	Decapoda	Decapod	Decapoda^{P}	0	0	0	0	0	0.1	0	0	0
	Euphausiidae	Krill	*Nyctiphanes australis* ^{P}	46.0	61.7	100	72.6	92.6	55.8	65.3	0	91.3
	Isopoda	Isopod	Isopoda^{P}	0	0	0	0	0	0	0	0	8.7
	Maxillopoda	Copepod	Maxillopoda^{P}	0	0	0	0	0	0.6	0	0	0
	Ostracoda	Ostracod	Ostracoda^{P}	0	0	0	0	0	0	1.1	0	0
	Palinuridae	Southern rock lobster	*Jasus edwardsii* ^{B}	0	0	0	0	0	0	0	0	0
	Pandalidae	Pandalid prawn	Pandalidae^{B}	0	0	0	0	0	0	0	0	0
	Penaeidae	Western king prawn	*Melicertus latisulcatus* ^{B}	0	0	0	0	0	0	0	0	0
		Southern velvet shrimp	*Metapenaeopsis palmensis* ^{B}	0	0	0	0	0	0	0	0	0
Bird	Procellariidae	Shearwater	*Puffinus* sp.^{P}	0	0	0	0	0	0	0	0	0
	Seabird - other	Seabird - other	Seabird - other^{P}	0	0	0	0	0	0	0	0	0
	Spheniscidae	Little penguin	*Eudyptula minor* ^{P}	0	0	0	0	0	0	0	0	0
Other	Donacidae	Cockle	*Plebidonax deltoides* ^{B}	0	0	0	0	0	0	0	0	0
	Insecta	Insect	Insecta^{T}	0	0	0	0	0	0	0	0	0
	Mollusca	Mullusc	Mollusca^{B}	0	2.4	0	0.6	0	19.3	0.5	0	0
	Nereididae	Nereid Worm	Nereididae^{P}	0	0	0	0	0	0	0	0	0
	Plant - marine	Plant - marine	Plant - marine^{B}	0	1.6	0	0	0	1.7	0.2	13.5	0
	Plant - terrestrial	Plant - terrestrial	Plant - terrestrial^{T}	0	0	0	0	0	0	0	0	0
	Polychaeta	Worm	Polychaeta^{P}	0	0	0	11.4	0	0.8	0	0	0
	Salpidae	Salp	Salpidae^{P}	0	0.6	0	1.3	0	1.1	2.9	0	0

Guilds based on diet of predator groups

SIMPROF detected 8 significantly different guilds among the 37 predator groups, based on similarities in their diets ($P < 0.05$, Figure 3.1). Because the diet data are weighted by the number and size of prey consumed, these guilds reflect differences and similarities in the relative prey mass taken by each group. The average number of predator groups in each guild was 4.6 ± 4.0, with a maximum of 11. Three guilds contained a single predator group, indicating that their diets were significantly different from all other predator groups in the study (*A. p. doriferus* adult males, *A. forsteri* juveniles and *C. novaehollandiae*, Figure 3.1). The average similarity level between different guilds was 20% ± 18%. Five of the guilds were discerned with diet similarities of less than 25%, indicating that 75% of their diets comprised either different prey species or different proportions of the same prey. The 3 remaining guilds were discerned with diet similarities between 30%-54%, which were predator groups of *A. p. doriferus* and the age/sex groups of *A. forsteri*. The average of within-guild similarity could be calculated for guilds 1, 4, 5, 6 and 8, because they had more than one predator group, and it was 33.8%, 73.5%, 34.5%, 24.3% and 57.7% respectively (average 44.8% ± 20.2%).

SIMPER determined how much each prey taxon accounted for the differences between the guilds (Table 3.3). Prey that accounted for these differences do not indicate that a guild was its sole consumer, but rather that such prey were relatively important for one guild compared to the others. Overall, the prey species that accounted for the differences between the guilds (Table 3.3) comprised an average of 71.8% ± 13.8% of each guild's total consumption. Guild 1 (11 predator groups) was largely discerned by its consumption of *S. sagax*, *E. australis* and *Pempheris* sp. Guild 2 (*A. p. doriferus* adult males) was unique in its high consumption of *E. nitidus* and Monocanthidae. Guild 3 (*A. forsteri* juveniles) was distinguished by consumption of Ommastrephidae, *Pempheris* sp. and *N. gouldi*. Guild 4 (Adult male and female *A. forsteri*) was distinguished from other predator groups by the consumption of *E. minor*, *E. nitidus* and *N. gouldi*. Guild 5 (6 predator groups) was distinguished by its consumption of *Trachurus* sp., *N. australis*, *N. gouldi* and *E. australis*. Guild 6 (6 predator groups) used *S. australis*, *T. maccoyii*, *N. gouldi* and *E. australis*. Guild 7 (*C. novaehollandiae*) was unique in its consumption of *T. degeni*, terrestrial plant material and Labridae. Guild 8 (9 predator groups) was largely discerned by its consumption of *N. australis* and other Crustaceans.

Figure 3.1. Bray-Curtis similarity dendrogram of the diets of the 37 predator groups. Significantly different guilds are numbered 1-8 at the bottom of the dendrogram (abbreviations: LT: long trip, ST: short trip, JUV: juvenile, AF: adult female, AM: adult male).

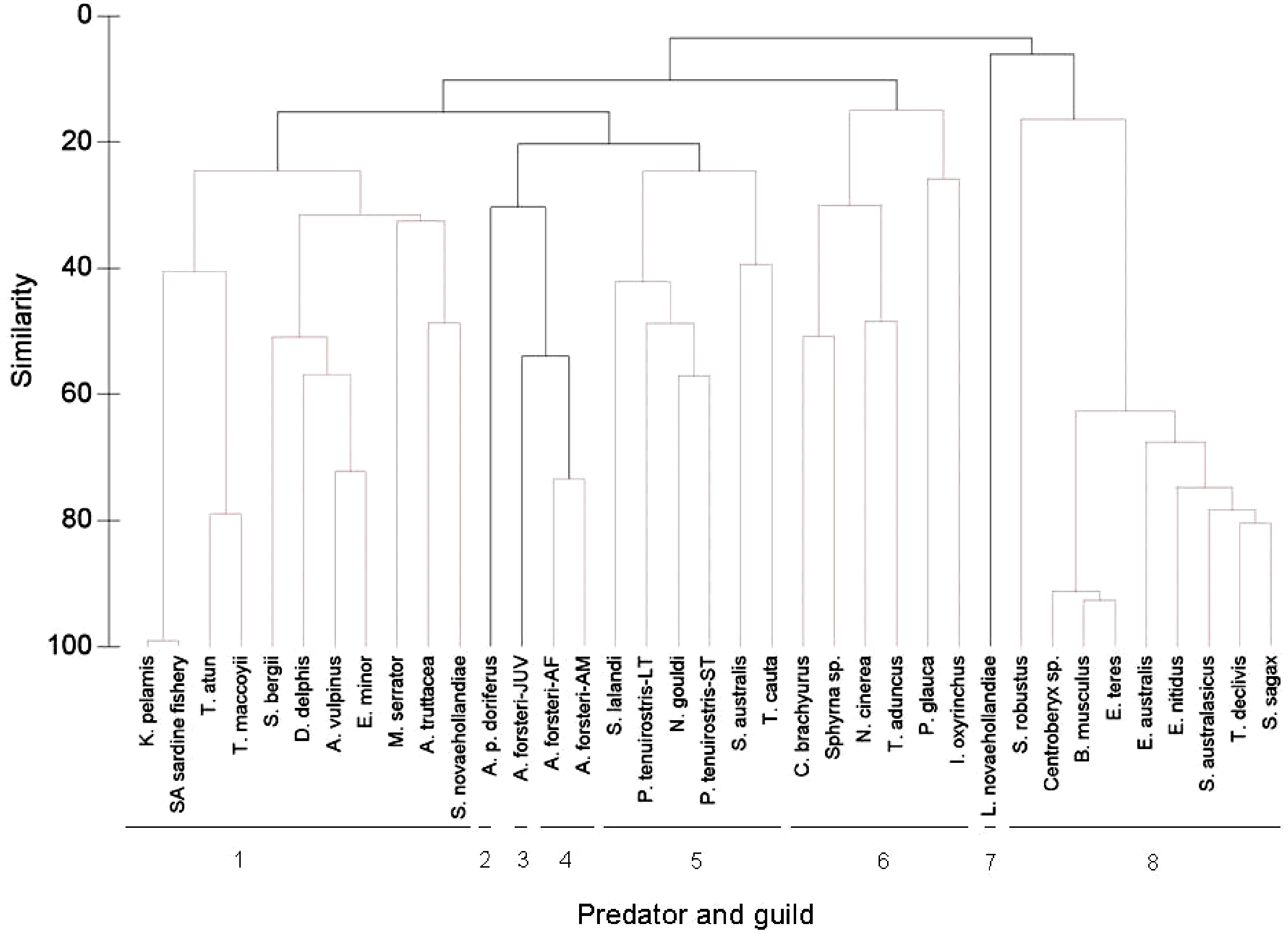

Table 3.3. SIMPER results for prey taxa that contributed to the differences between guilds (reported as SIMPER percentage). Only prey taxa that were consumed by a guild are reported for that guild (prey taxa that were not consumed by a guild can contribute to differences between guilds if the other guild used the taxa).

Guild	Prey	SIMPER (%)	Average proportion of diet
1	*Sardinops sagax*	48	34.7
1	*Engraulis australis*	35	25.8
1	*Pempheris* sp.	10	7.6
1	Atherinidae	3	4.5
1	*Nototodarus gouldi*	2	4.8
1	*Scomber australasicus*	1	4.6
1	*Thyrsites atun*	1	4.0
	Total	**100**	**86.0**
2	*Emmelichthys nitidus*	51	44.9
2	Monocanthidae	49	36.7
	Total	**100**	**81.6**
3	*Pempheris* sp.	48	16.7
3	Ommastrephidae	38	22.5
3	*Nototodarus gouldi*	14	14.0
	Total	**100**	**53.2**
4	*Eudyptula minor*	46	21.9
4	*Emmelichthys nitidus*	20	17.8
4	*Nototodarus gouldi*	19	13.7
4	Ommastrephidae	10	11.5
4	Monocanthidae	5	8.1
	Total	**100**	**73.1**
5	*Trachurus* sp.	44	23.5
5	*Nyctiphanes australis*	27	15.2
5	*Nototodarus gouldi*	16	12.4
5	*Engraulis australis*	11	8.0
5	*Sepia apama*	2	4.5
	Total	**100**	**63.5**
6	*Sepioteuthis australis*	32	16.2
6	*Thunnus maccoyii*	27	13.6
6	*Nototodarus gouldi*	19	13.5
6	Octopodidae	17	10.0
6	*Sepia apama*	5	7.2
	Total	**100**	**60.5**
7	*Thamnaconus degeni*	57	29.1
7	Plant - terrestrial	25	20.0
7	Labridae	18	14.7
	Total	**100**	**63.8**
8	*Nyctiphanes australis*	72	65.0
8	Crustacean	26	23.8
8	Fish eggs	2	3.5
	Total	**100**	**92.3**
	Grand mean ± SD		**71.8 ± 13.8**

To determine the habitats that were typically used by the above guilds, we considered the habitats used by their prey. Most prey were from PELAGIC habitats, with guild 7 (*C. novaehollandiae*) being the only BENTHIC guild (65.2% of prey from benthic habitats). In total, 33 predator groups were classified as PELAGIC, 4 were classified as BENTHIC and none were classified as TERRESTRIAL (Table 3.4). As expected, the prey taxa used by the 33 PELAGIC predators were significantly different to the prey taxa used by the 4 BENTHIC predators ($R = 0.65$, $P = 0.026$). Predator groups that were classified as PELAGIC (based on proportional contribution of prey) were present in all guilds except guild 7 and BENTHIC predator groups were present in guilds 2, 6 and 7. Guilds 1, 3, 4, 5 and 8 used more than 80% PELAGIC prey, and guilds 2 (55.1% PELAGIC) and 6 (62.7%) were also classified as PELAGIC, but used relatively more BENTHIC prey (Table 3.4).

Table 3.4. Habitat classification of each predator group, based on the habitat of the majority of their prey and sorted by guilds.

Guild	Predator	Pelagic	Benthic	Terrestrial	Habitat classification of predator
1	*A. vulpinus*	100	0	0	Pelagic
1	Sardine fishery	100	0	0	Pelagic
1	*Sarda australis*	100	0	0	Pelagic
1	*T. atun*	100	0	0	Pelagic
1	*T. maccoyii*	99.8	0.2	0	Pelagic
1	*D. delphis*	91.7	8.3	0	Pelagic
1	*E. minor*	87.3	12.7	0	Pelagic
1	*M. serrator*	82	18	0	Pelagic
1	*S. bergii*	72.4	27.6	0	Pelagic
1	*A. truttacea*	68.6	31.4	0	Pelagic
1	*S. novaehollandiae*	41.5	58.5	0	Benthic
	Average	**85.8 (18.7)**	**14.2 (18.7)**	**0**	
2	*A. pusillus doriferus*	55.1	44.9	0	Pelagic
3	*A. forsteri* -JUV	80.1	19.9	0	Pelagic
4	*A. forsteri* -AM	85	15	0	Pelagic
4	*A. forsteri* -AF	82.7	17.3	0	Pelagic
	Average	**83.9 (1.6)**	**16.1 (1.6)**	**0**	
5	*P. tenuirostris* -LT	99.8	0.2	0	Pelagic
5	*P. tenuirostris* -ST	99.6	0.1	0.3	Pelagic
5	*N. gouldi*	96.9	3.1	0	Pelagic
5	*S. lalandi*	95.3	4.7	0	Pelagic
5	*T. cauta*	66.7	33.3	0	Pelagic
5	*S. australis*	66	34	0	Pelagic
	Average	**87.4 (16.4)**	**12.6 (16.4)**	**0**	
6	*I. oxyrinchus*	100	0	0	Pelagic
6	*C. brachyurus*	80.2	19.8	0	Pelagic
6	*Sphyrna* sp.	78.5	21.5	0	Pelagic
6	*P. glauca*	76.1	23.9	0	Pelagic
6	*N. cinerea*	23.2	76.8	0	Benthic
6	*T. aduncus*	18.1	81.9	0	Benthic
	Average	**62.7 (33.7)**	**37.3 (33.7)**	**0**	
7	*C. novaehollandiae*	9.6	65.2	25.2	Benthic
8	*B. musculus*	100	0	0	Pelagic
8	*C. lineatus*	100	0	0	Pelagic
8	*E. australis*	100	0	0	Pelagic
8	*E. teres*	100	0	0	Pelagic
8	*T. declivis*	99.4	0.6	0	Pelagic
8	*S. sagax*	99.3	0.7	0	Pelagic
8	*S. australasicus*	96.1	3.9	0	Pelagic
8	*S. robustus*	86.5	13.5	0	Pelagic
8	*E. nitidus*	79	21	0	Pelagic
	Average	**95.6 (7.6)**	**4.4 (7.6)**	**0**	

Discussion

Our study focussed on the diets of pelagic predators and our sampling was influenced by their availability and the availability of datasets on the diet of other predators. Our study did not include some ecologically-important species that occur in the region, because data on their diets were not available. Four shark species that occur in the region, but are not part of this study, are *Carcharodon carcharias*, *Galeorhinus galeus*, *Carcharhinus obscurus* and *Mustelus antarcticus*, all of which are likely to compete with some of the predators in this study (Simpfendorfer *et al.* 2001, Stevens 2005, Bruce *et al.* 2006). Seabirds such as *Pelagodroma marina*, *Thalassarche* sp. and *Puffinus carneipes* (Copley 1996) and toothed whales (Odontoceti) are also abundant in this region and studies from other regions indicate that they are pelagic predators (e.g. Evans and Hindell 2004, Brooke 2004, Colabuono and Vooren 2007). The prevalence of Myctophidae and *Todarodes filippovae* in the diets of several predators indicates that they are ecologically important species. Nevertheless, they were not included in this study because their diet data were not available.

Dietary differences among demographic groups of a single species are often profound, indicating that some demographic groups should be separated in studies of ecosystem processes (Polis 1984, reviewed in Andersson 1994, Isaac 2005). Our study indicated that diets were not significantly different in 2 of the 3 age/sex classes of *A. forsteri* nor between the two behavioural patterns analysed for *P. tenuirostris*. The diet of juvenile *A. forsteri* was in a different guild to the adult age classes of that species, and Page *et al.* (2005) found significant differences among all 3 of its age/sex classes. Our data indicate that in the broad context of this study, diet differences between these age/sex classes are not as important as inter-species differences. Despite this, our study would undoubtedly have benefited from the inclusion of information about additional predator species and from splitting the different age/sex classes of predator groups to identify ontogenetic diet shifts. Had we assessed dietary differences among demographic groups of each species, it is likely that we would have identified additional guilds, and so the results we have presented are likely to be over-simplified. Nonetheless, we describe differences between the diets of 37 predator groups that are ecologically and/or economically important in the region.

The patterns of resource partitioning identified in this study indicate that, based on their diets, the 37 predator groups could be categorised into 8 guilds. We found that the guilds could be discerned by 2-6 prey taxa and in all guilds these prey taxa comprised more than half of their diets (Table 3.3). The guilds were split at similarity levels that averaged 20% ± 18%, which is at the lower end of dietary overlap values reported in other studies of resource competition in the marine environment (20%-70%, summarised in Garrison and Link 2000). The average similarity level within guilds was 44.8% ± 20.2%, which is lower than has been reported elsewhere (50%-80%, summarised in Garrison and Link 2000). Together, these low similarity levels indicate that some of the predator groups are specialists, with some groups using <5 prey taxa. This is an accurate reflection of the SA sardine fishery, but is unlikely to be true for *Sarda australis*, *A. vulpinus*, *B. musculus* and *I. oxyrinchus*, which have been reported to have broader diets elsewhere (e.g. Stillwell and Kohler 1982, Collette and Nauen 1983, Preti *et al.* 2001, Sears 2002). Rather, the narrow range of prey reported here is likely to be a result of the limited availability of diet samples, and has most likely resulted in the lower than expected similarity levels. The apparent specialist nature of *E. australis*, *E. teres*, *S. robustus* and *C. lineatus*, largely resulted from their use of *N. australis*, but the similarity levels within their guild (Guild 8) may have been inflated by the broad taxonomic groups used to classify their other prey (Daly 2007, Table 3.2). The difficulty in identifying the prey of predators that use small crustaceans is a common problem in dietary studies (e.g. Braley 2010). Despite these shortcomings, our analyses provide a practical picture of this complex and productive ecosystem.

Nyctiphanes australis and other pelagic crustaceans are the key links between primary production and higher trophic levels in this pelagic ecosystem, as demonstrated by their importance in the diets of the members of guilds 5 and 8. The small pelagic fishes from guild 8 and *N. gouldi* (guild 5) were, in turn, important prey for the predators in guilds 1, 2, 3, 4, 5 and 6, indicating that much of the energy flow in this ecosystem involves just 3 trophic levels before it is available to top predators (2 trophic levels for some predator groups, including the SA sardine fishery). It is likely that populations of *N. australis* and other pelagic crustaceans in this region are regulated by the timing and strength of seasonal upwelling, which enhances its primary productivity (Middleton *et al.* 2007). Bottom-up control of this pelagic ecosystem and the importance of Euphausidae are consistent with findings from other highly productive systems, such as the Arctic seas (Falk-Petersen *et al.* 1990) the Antarctic and along the upwelling regions of the California, Humboldt, Benguela and Canarias current systems (Mauchline and Fisher 1969). Such efficient energy flows enhance tertiary production and typically support relatively dense aggregations of predators and commercial fisheries (Mauchline and Fisher 1969, Ward *et al.* 2006). However, the reliance on upwelling also highlights the vulnerability of the top predators and commercial fisheries in this region to climate changes that may affect the timing or strength of upwelling and populations of Euphausidae. Interestingly, oceanographic modelling of this region indicates that, based on predicted climate changes, seasonal upwelling may intensify in this region, which may further enhance productivity (Middleton *et al.* 2007).

Most of the predator groups in this study used multiple prey taxa, indicating that their consumption patterns are more generalist than specialist. Higher diversity of diets typically reduces competition among predators because there is less reliance on single prey taxa by any predator group (reviewed in Helfman *et al.* 1997). The prevalence of generalist predators in this study is similar to findings from similar studies of the marine environment, which typically indicate at least slight differences in the diets of predator groups (reviewed in Helfman *et al.* 1997). Even among the members of Guild 8, which used large proportions of *N. australis*, there were 10-20% differences in their diets. Differences in the phenotype (particularly body size) of predators are often highlighted to explain apparent dietary differences between groups of predators. The body size of a predator constrains the size of prey that can be captured and killed, and this is undoubtedly an important determinant of the diet differences among many predators (e.g. Ashmole 1968, Hulsman 1987). Our study included such a broad spectrum of different sized predators that it is not surprising that clear differences were apparent between guilds and also that many of the predator groups were recorded as the prey of other predator groups. Many studies indicate that competition for prey is partly responsible for the evolution of life history differences between predators, but competition for a single prey taxon does not appear to be an important factor that is driving the structure and function of the pelagic ecosystem in this region.

We showed that several competitors were present in most guilds, which is a common finding in marine ecosystems (Garrison and Link 2000). The maintenance of biodiversity in a marine ecosystem provides an ecological buffer against change and highlights the importance of an ecosystem approach to fisheries management. Such buffers protect ecosystems from perturbations and provide scope for adaptive management, which is beneficial because management decisions are typically based on an incomplete understanding of ecosystem processes. Three guilds in our study had only one member, which stems from our focus on pelagic predators. Juvenile *A. forsteri* forage in oceanic waters, up to 1000 km south of Australia (Page *et al.* 2005), well beyond the focal region of this study, so it is not surprising that they were shown to use some unique resources. Likewise, *C. novaehollandiae* used some terrestrial feeding grounds. Populations of *A. p. doriferus* are recovering in the region of this study, following extensive harvesting ~200 years ago. We showed that male *A. p. doriferus* had a relatively narrow prey base, with two prey comprising more than 80% of their diet, but their large proportion of benthic prey indicates that benthic competitors may have been filling that void over the past ~200 years.

Theories of ecosystem resilience indicate that the risk of collapse is reduced if ecologically important guilds are comprised of many species (MacArthur 1955, Ehrlich and Ehrlich 1981). Empirical evidence of dramatic changes in the stock of *Sardinops* sp., in this and other regions, highlight its ecologically similar competitors (e.g. Murphy 1967). In South Australia in 1995 and 1998, two mass mortalities of *S. sagax* resulted from a herpes virus, which killed more than 75% and 70% of the spawning stock respectively (Ward *et al.* 2001a,b). Following the mortality events, a competitor with similar ecological requirements, *E. australis*, increased in range and abundance, undoubtedly exploiting the resources once used by *S. sagax* (Ward *et al.* 2001a,b). Following both mortalities, the stock of *S. sagax* recovered quickly (Ward *et al.* 2001 a,b). A similar pattern was observed in California, following a collapse in the stock of *Sardinops* sp., but it took longer to recover (Murphy 1967). In a resilient ecosystem, predators would exhibit reduced consumption of *S. sagax* and increased consumption of *E. australis*. We are not aware of any studies that recorded this switch in the diet of a predator, but there is evidence that predator populations were adversely affected by the *S. sagax* mortality, at least in the short term. Before the mortality, the diet of *M. serrator* included up to 50% *S. sagax*, and reductions in the size of the *S. sagax* stock were mirrored in the diet of *M. serrator* (Bunce 2004). In addition, the survival of *S. bergii* and *S. albifrons sinensis* (little tern) were negatively affected by the mass mortalities (Taylor and Roe 2004, McLeay *et al.* 2009b). Similar results, and even predator population crashes, have been recorded in other regions, following reductions in the availability of *Sardinops* sp. (Beamish *et al.* 1999, Crawford *et al.* 2008), highlighting their ecological importance.

The SA sardine fishery was included in a guild with 10 other predator groups, which are the most likely competitors for *S. sagax* in this region. Given the impact of the depletion of *S. sagax* on *M. serrator* and *S. bergii*, it is worth discussing the role of other potential competitors for *S. sagax*. Before the SASF commenced, populations of *T. maccoyii* were severely depleted by fishing, and are thought to still be <15% of their virgin biomass (Basson *et al.* 2004). There are no ecological studies from the era when *T. maccoyii* were most abundant in this region, but it is likely that those larger populations consumed more *S. sagax*. Trophic modelling will facilitate analyses of the historical roles of predators such as *T. maccoyii*. Populations of *D. delphis* are known to directly interact with the SA sardine fishery (Hamer *et al.* 2008), confirming the importance of *S. sagax* to *D. delphis*, but no studies have quantified the role of this apparent competition for resources. The importance of *S. sagax* to the predators in this guild (including *T. atun*, *Sarda australis*, *A. vulpinus*, *E. minor*, *A. truttacea* and *S. novaehollandiae*) and to other predators (e.g. *C. brachyurus*, *N. gouldi,* and *S. lalandi*) highlights the need for the ongoing monitoring of ecosystem processes in the region used by the SA sardine fishery. Improved understanding of the diets of these predators will provide fishery managers with complementary, fishery-independent measures of *S. sagax* availability, which would help to ensure that natural predators of *S. sagax* are not adversely impacted by the fishery.

Fisheries managers, researchers and the general public all recognise that determining sustainable fishery yields requires ongoing data on both the population dynamics of exploited species and the ecosystem processes that influence the population (Fletcher 2005, 2006, Smith *et al.* 2007). Because data on ecosystem processes are expensive to collect and difficult to incorporate into fishery models, management typically continues with a single-species focus, aimed at ensuring that fish stocks provide the optimal yield in the long term (e.g. Smith *et al.* 2007). This approach is used effectively in many fisheries, including the SA sardine fishery, but there is increasing recognition at the policy level that improved knowledge of ecosystem processes will reduce the risk to populations of predators that use the same stocks.

To provide a snapshot of the pelagic ecosystem used by the SA sardine fishery, we summarised the diets of several pelagic predators and the degrees of similarity between their diets. We also discussed how resilient the ecosystem is to changes in prey abundance, noting that some of the predator populations have been impacted by reductions in the abundance of *S. sagax*. The construction of guilds of predators has helped to simplify the complex ecosystem processes that support the SA sardine fishery and identify groups of predators that can be monitored to improve fisheries management. These data will serve as assess managers to assess the potential benefits of establishing ecological performance indicators for the SASF.

4 SPATIAL DISTRIBUTION OF CONSUMPTION EFFORT OF KEY APEX PREDATORS AND THEIR OVERLAP WITH THE SA SARDINE FISHERY

SD Goldsworthy, B Page, A Wiebkin, L Einoder, A Baylis, L McLeay, N Bool, J McKenzie, D Hamer

Introduction

The spatial scales at which ecological performance indicators (EPI), such as diet, reproductive success and population change, integrate information on prey availability, for land-breeding marine predators (seabirds and seals), will vary considerably among species, depending on their reproductive and foraging strategies. As such it is critical in the assessments of the merit of individual performance indicators that spatial and temporal discontinuities between the areas of potential anthropogenic impact and other areas are fully understood. In particular, it is vital to ascertain whether EPI are measuring anthropogenic factors associated with fishing or natural environmental variation. The only way to assess the at-sea movements of seals and seabirds, and thus assess their foraging areas, is through tracking studies. Recent developments in satellite telemetry enable even small marine species such as seabirds, to be instrumented with either satellite transmitters (platform transmitting terminals: PTT) or GPS tags (McLeay *et al.* 2010). Satellite tracking is the most cost-effective and reliable method, and has become a standardised component of most ecological studies of tractable marine species in recent years (reviewed in Hart and Hyrenbach 2009).

We used state of the art satellite tracking methods to obtain data on the at-sea foraging distributions of a range of land-breeding apex predators that may provide valuable EPI for the South Australian (SA) sardine fishery. We ask the key questions: to what extent do the distributions of foraging and consumption effort of these predators in the eastern Great Australian Bight (EGAB) ecosystem, overlap with the core area of the sardine fishery; and can this knowledge provide the basis to identify those species mostly likely to elicit a measurable response to changes in sardine biomass in their foraging areas?

The aims of this study were to estimate the spatial distribution of foraging effort of key land-breeding marine apex predators and, based on dietary assessments and estimates of consumption, to determine the extent to which the spatial distribution of consumption effort of key prey taxa, including sardines, overlaps with the spatial distribution of catch in the SA sardine fishery.

Materials and methods

Satellite telemetry data

Satellite telemetry data were obtained from five land-breeding apex predators: New Zealand fur seals (NZFS*; Arctocephalus forsteri*); Australian sea lions (ASL, *Neophoca cinerea*); Short-tailed shearwaters (STSW, *Puffinus tenuirostris*), little penguins (*Eudyptula minor*) and crested terns (*Sterna bergii*). For most species, telemetry data were obtained from ARGOS PTTs, with some of the most recent data obtained from fully archival or archival/ARGOS linked GPS tags. For Australian sea lions, data on the spatial distribution of foraging effort were obtained from Goldsworthy *et al.* (2010).

Filtering and analysis of time spent in areas

PTT satellite location data were obtained through CLS ARGOS (Toulouse, France). The location-class Z positions were omitted due to the magnitude of their error (Sterling and Ream 2004), leaving location classes B, A, 0, 1, 2, 3 for subsequent analyses. For GPS telemetry units, GPS positions were calculated either using the LocSolve (Wildlife Computers, Redmond, Washington USA) or Sirtrack (Havelock North, New Zealand) software packages. The R statistical software (version 2.8.1, R Development Core Team, R Foundation for Statistical Computing, Vienna) and the Trip package (M. D. Sumner, University of Tasmania, Hobart) were used to apply a speed filter as described by McConnell *et al.* (1992) to remove erroneous positions. The maximum horizontal speed considered possible was 7.2 km/h for NZFS (Page *et al.* 2006); 11 km/h for ASL and little penguins (Goldsworthy *et al.* 2010; A Wiebkin unpublished data); and 60 km/h for STSW and crested terns (McLeay *et al.* 2010, Einoder 2010). In order to remove all time on land and restrict subsequent analyses to data on foraging trips only, the departure and arrival times, and locations of successive foraging trips were calculated following the methods detailed by Goldsworthy (2009).

To determine key areas used during foraging trips, a grid of cells of 5 x 5 km was developed using the Trip package, and the amount of time that each individual seal or seabird spent within each cell area (25 km^2) was calculated assuming a constant horizontal speed based on the distance and duration between successive filtered locations and interpolated new positions for each 15 minutes (of time) along the satellite track. The number of original and interpolated positions located within these cells were summed and assigned to the central node. To ensure the different deployment durations recorded for each individual did not bias comparisons, the amount of time spent in each cell was converted to a proportion of the total time spent at sea for each individual.

Model development

Statistical models using data distributions were used to estimate the spatial distribution of foraging effort of the five tracked species throughout SA, following the methods outlined in Goldsworthy *et al.* (2010). Continental shelf and slope waters in SA were overlaid with the 5 x 5 km grid, and the distance from each breeding colony for each species and location tracked to each node in the array was calculated. The depth at each cell node was also calculated using bathymetric data from GeoScience Australia. For each subpopulation, the time spent at distance and depth from the subpopulation site was examined using density plots created within R. The fits of observations to the normal probability

$$f(x:\mu,\sigma)=\frac{1}{\sqrt{2\pi\sigma}}e^{-\left(\frac{(x-\mu)^2}{2\sigma^2}\right)},$$

and to the gamma probability density functions,

$$f(x:k,\theta)=x^{k-1}\frac{e^{\frac{-x}{\theta}}}{\theta^k\Gamma(k)},$$

were examined using the MASS package, with the best density function fits selected for distance and depth. The means (*μ*) and standard deviations (*σ*) were calculated for normal probability distributions, while the shape (*k*) and scale (*θ*) functions were determined for gamma distributions. The *x* variable represents either depth (m) or distance (km).

The probability of an animal from a given species and colony foraging in a particular grid node was modelled as a continuous variable on the range [0,1]. The probabilities were calculated as the joint probability (i.e., product) of distance and depth (assuming both are independent), based on the distance of the node from the subpopulation and its depth, after standardising each onto the range [0,1]. For NZFS and ASL, in which spatial distribution of foraging vary markedly between males and females, sex-specific models were developed

(Goldsworthy *et al.* 2010). For juvenile NZ fur seals which forage exclusively in oceanic pelagic waters, all data were excluded (Page *et al.* 2006).

Each foraging model was constrained by the upper limits of the observed distance and depth in the data on which it was based. For subpopulations in the lower Eyre Peninsula, Spencer Gulf and Kangaroo Island regions, distances from subpopulations were corrected to follow the minimum coastal route where the shortest distance was over land.

The distribution of consumption effort (by prey type) was based on diet summaries and population and bio-energetic models developed for each species for *Ecopath with Ecosim* models (see Chapter 5). Consumption was apportioned for each colony within each species based on its proportional contribution to the total estimate of its population within the EGAB region (Table 4.1). For STSW and NZFS that only spend part of the year foraging within the EGAB and/ or where a portion of total consumption occurs outside the EGAB region, only consumption within the EGAB was considered in spatial models (i.e. total consumption minus estimated import, see Chapter 5). For species such as little penguins and crested terns where the tracking data was only available for the breeding and chick rearing periods, spatial foraging and consumption models were restricted to March to November (0.75 years) and November to February (0.33 years), respectively.

Data on the spatial distribution (latitude and longitude positions) of sardine catch was available for the 1999 to 2010 fishing seasons. In some instances where the lat/long position of the catch was not recorded it was applied proportionally to other catches where lat/long was reported, from the same year. Plots of the spatial distribution of consumption effort and sardine fishery catch were visualised in MapInfo™ (Version 9, MapInfo Corporation, New York, USA) and then interpolated (triangular irregular network interpolation with 5th order polynomial) and plotted using VerticalMapper™ (Version 3.0, Northwood Geosciences Ltd, Nepean, Ontario, Canada).

Results

Satellite telemetry data from the five key land-breeding apex predators were derived from a total of 403 deployments and 3,601 individual foraging trips (Table 4.2). Little penguin data were derived from 85 deployments (85 foraging trips) across seven locations (Granite Is, Kingscote, Olive Is, Pearson Is, Reevesby Is, Troubridge Is and West Is); crested tern data were derived from 22 deployments (25 foraging trips) from one site (Troubridge Is); STSW data were derived from 22 deployments (22 foraging trips) from one site (Althorpe Is); NZFS data were derived from 64 deployments (137 foraging trips) from four sites (Cape

Gantheaume and Cape du Couedic [Kangaroo Is], North Neptune Is, Liguanea Is); and ASL data were derived from 210 deployments (3,332 foraging trips) from 17 sites (see Goldsworthy *et al.* 2010) (Table 4.2).

Results from the analyses of the time spent at depth and distance from colony are presented in Table 4.2. They indicate marked variability in the mean and maximum depth and distance across species and sex classes (seals). The best model fits (normal or gamma) for depth and distance variables, and the mean and standard deviation and shape and scale parameters for the models are also presented (Table 4.2).

NZFS concentrated their foraging and consumption effort over shelf waters between lower Eyre Peninsular and south-east of Kangaroo Island, including the lower portions of Spencer Gulf (Figure 4.1). In contrast, ASL foraging and consumption effort is more concentrated in inner shelf (near coast) and lower gulf waters (Figure 4.1). For crested terns and little penguins that were only tracked over the breeding and chick rearing periods, mean foraging ranges were small (11.4 km and 14.2 km and, respectively) and mean depths shallow (22.0 m and 35.6 m, respectively) (Table 4.2). Consumption effort for both these species is concentrated in the lower gulfs and inshore regions of the western Eyre Peninsula (Figure 4.1). STSW travelled a mean of 39.2 km over mean depths of 79.9 m on their short foraging trips (Table 4.2). Consumption effort on these trips is concentrated in shelf waters off western Kangaroo Island and south and west of the lower Eyre Peninsula (Figure 4.1).

Combined plots of the estimated spatial distribution of consumption by the five predators, based on the foraging distribution models developed for each species, are presented in Figures 4.1 F, highlighting the importance of lower gulf and shelf waters between the western Eyre Peninsula and Kangaroo Island to these predator populations. The estimated spatial distribution of consumption of fish, cephalopods, crustacean and sardines by the five predators in the EGAB region are presented in Figures 4.2 A-D. The plot for sardine consumption again highlights the importance of lower gulf and shelf waters off the lower Eyre Peninsula and Kangaroo Island to predator populations. The spatial distribution of the SA sardine fishery catch and the spatial overlap in sardine catch and consumption by the five predators is presented in Figures 4.2 E-F. These figures demonstrate the potential for competition is greatest in the region of the fishery and adjacent waters off Kangaroo Island and western and lower Eyre Peninsula (Figure 4.2). The mean annual catch of sardines between 1999 and 2010 was 21,202 t (sd = 12,592).

A summary of the estimated total annual consumption by the five key predators, their consumption within the EGAB region and a breakdown of their consumption of small pelagic fish, sardines, total bird, fish, cephalopod and crustacean consumption is presented in Table

4.3. Sardine consumption is estimated to make up only 1.2% of the total annual consumption within the EGAB region by the five key land-breeding marine predators; and only 1.6%, 0.1%, 3.5%, and 0.2% of the prey consumed by NZFS, ASL, little penguins, and petrels, respectively (Table 4.3). Crested terns were the only species for which sardine consumption represented a significant component of prey consumed (22.7%, Table 4.2). Small pelagic fish consumption was important for all species (NZFS 28.0%, little penguin 75.1%, STSW 32.9%, crested terns 63.6%) except ASL (0.9%) (mean of 27.7% across the five species), but for most predators the contribution of sardines to small pelagic fish consumption was relatively minor (4.2% on average). Crested terns were the exception because sardines comprised 35.7% of their small pelagic fish consumption (Table 4.3).

Discussion

Spatial analyses of the distribution of foraging and consumption effort by the five key land-breeding apex predators studied indicates that biogeographically, shelf waters in the EGAB region off the western and lower Eyre Peninsula, south and south-east of Kangaroo Island and in the lower portions of Spencer Gulf and Gulf Saint Vincent, provide critical foraging habitat for these species (Figure 4.1 and 4.2). Prey consumption over shelf and lower gulf waters by NZFS is concentrated near their key breeding colonies south of the Eyre Peninsula and on the southern coastline of Kangaroo Island, with an estimated 30% of prey consumption occurring outside the EGAB region in oceanic waters south of their breeding colonies (Figure 4.1, Table 4.3). In contrast, consumption by ASL is restricted to continental shelf waters of the EGAB, with greater levels of consumption in inner-shelf waters closer to their breeding colonies (Figure 4.1). Little penguins are also estimated to restrict their foraging effort to shelf waters, and over at least nine months of the year, focus their foraging and consumption effort in inner shelf and lower gulf waters near their breeding colonies, especially off the western Eyre Peninsula and lower Spencer Gulf (Figure 4.1). STSW are highly migratory species, and during their spring/summer breeding season, undertake short (1-2 day) foraging trips over shelf waters, between longer oceanic trips to Southern Ocean waters. Approximately 30% of prey consumed during the chick rearing period is caught on short trips in shelf waters (Einoder 2010), accounting for about 18% of their total estimated annual consumption (Table 4.3). Shelf foraging and consumption effort is concentrated in the mid-outer shelf waters off lower Eyre Peninsula and south of Kangaroo Island (Figure 4.1). During their four month breeding season, crested terns have highly restricted foraging ranges, with most consumption focussed in close proximity to their breeding colonies, particularly in mid-lower Spencer Gulf and in Gulf Saint Vincent (Figure 4.1). For all the species investigated, with the exception of STSW, the core area of the SA sardine fishery in lower Spencer Gulf represented important foraging and prey consumption grounds (Figure 4.1 and 4.2).

Most of the consumption by the five key land-breeding apex predators consisted of fish (53%), squid (39%) and crustaceans (7%). Small pelagic fish accounted for 28% of the total consumption and 52% of the total fish consumed in the EGAB (Table 4.3). Overall, sardines only made up about 1% of the total prey biomass consumed by the five apex predators, and only 2% of the total fish biomass consumed (Table 4.3). The total estimated consumption of sardines by these predators (753 t/y), is very small (3%) relative to the current annual TACC (30,000 t) of the SA sardine fishery. As such, the catch of sardines by the fishery exceeds the consumption by the five apex predators wherever fishing effort occurs, but there are also large areas where consumption of sardines by the five apex predators exceeded that of the fishery (Figure 4.2 F).

Based on these analyses, crested terns were the only species identified for which sardine consumption represented a substantial component of total prey consumed (22.7%), and where this consumption overlapped with the core region of the fishery in southern Spencer Gulf and Investigator Strait (Figure 4.1 and 4.2). Therefore, of all the species investigated, crested terns are the most likely to provide potential ecological performance indicators for the SA sardine fishery, as spatial and consumption analyses suggest they may integrate information on the availability of sardines into their foraging, reproductive and population ecology.

For NZFS and ASL, foraging distribution models developed were based on considerable satellite tracking effort to account for both between individual, sex, inter-colony and seasonal differences foraging. Although tracking effort was also considerable for little penguins (n=85), and representative across multiple colonies across the EGAB region, tracking was confined to adult birds during the breeding and chick rearing period (approximately 9 months). How representative these data are of the non-breeding period or for juvenile birds is unclear, and more tracking works is required. For STSW and for crested terns tracking was confined to the chick rearing period (about 4 months) and from only 22 individuals (both species) from a single site. Clearly for these species additional tracking data from more individuals across multiple sites and throughout the year would be advantageous and would improve model estimates of the spatial distribution of foraging and consumption effort.

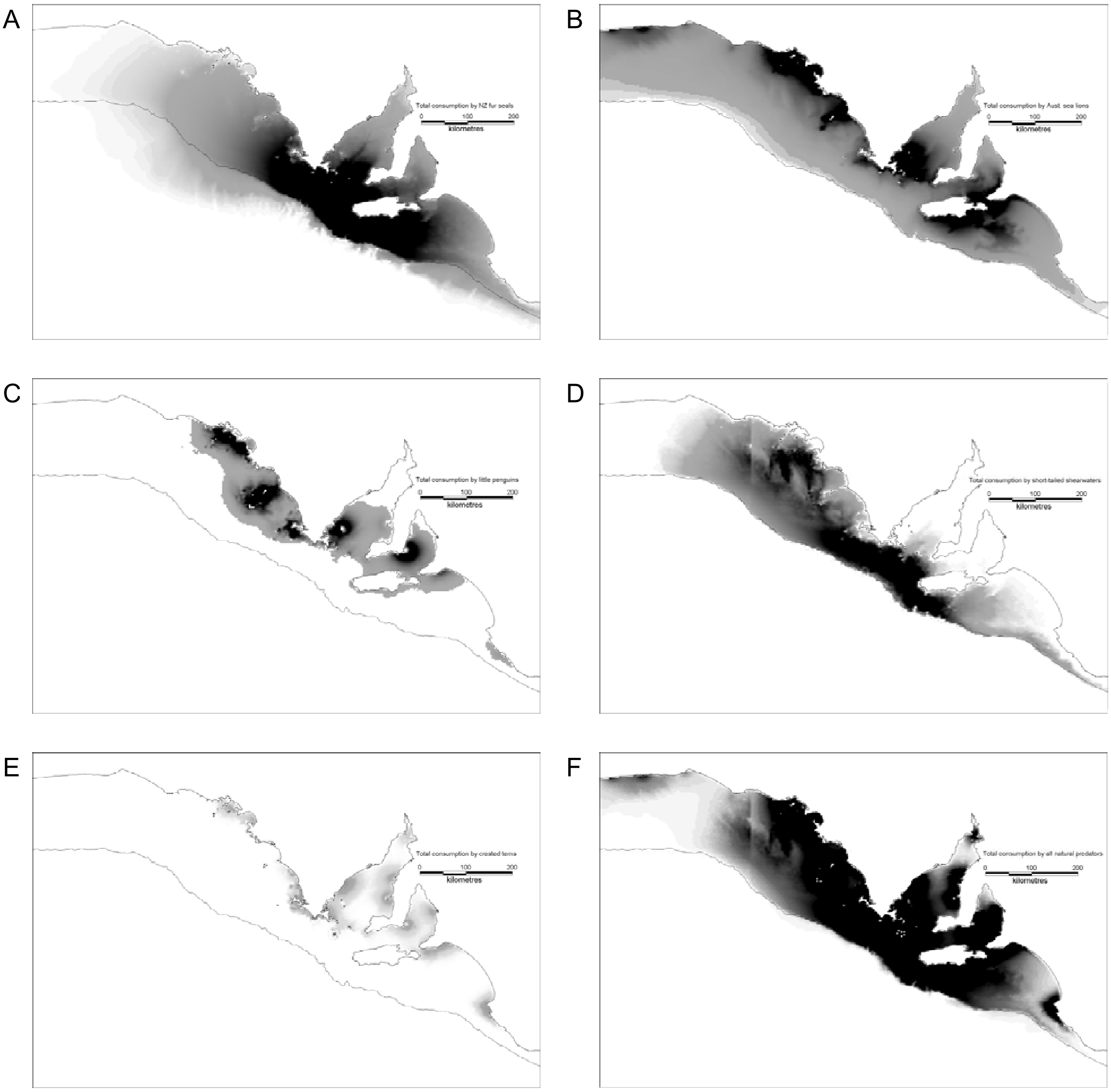

Figure 4.1. Estimated spatial distribution of foraging and consumption (t/y) effort by five key land-breeding apex predators: A NZFS, B ASL, C little penguins, D STSW, E crested tern and F all species combined. Foraging and consumption by species outside of shelf waters has been excluded. Data for crested terns and little penguins are restricted to the chick rearing period (4 and 9 months, respectively). For all other species, plots represent total annual consumption. Data for ASL are from Goldsworthy *et al.* (2010).

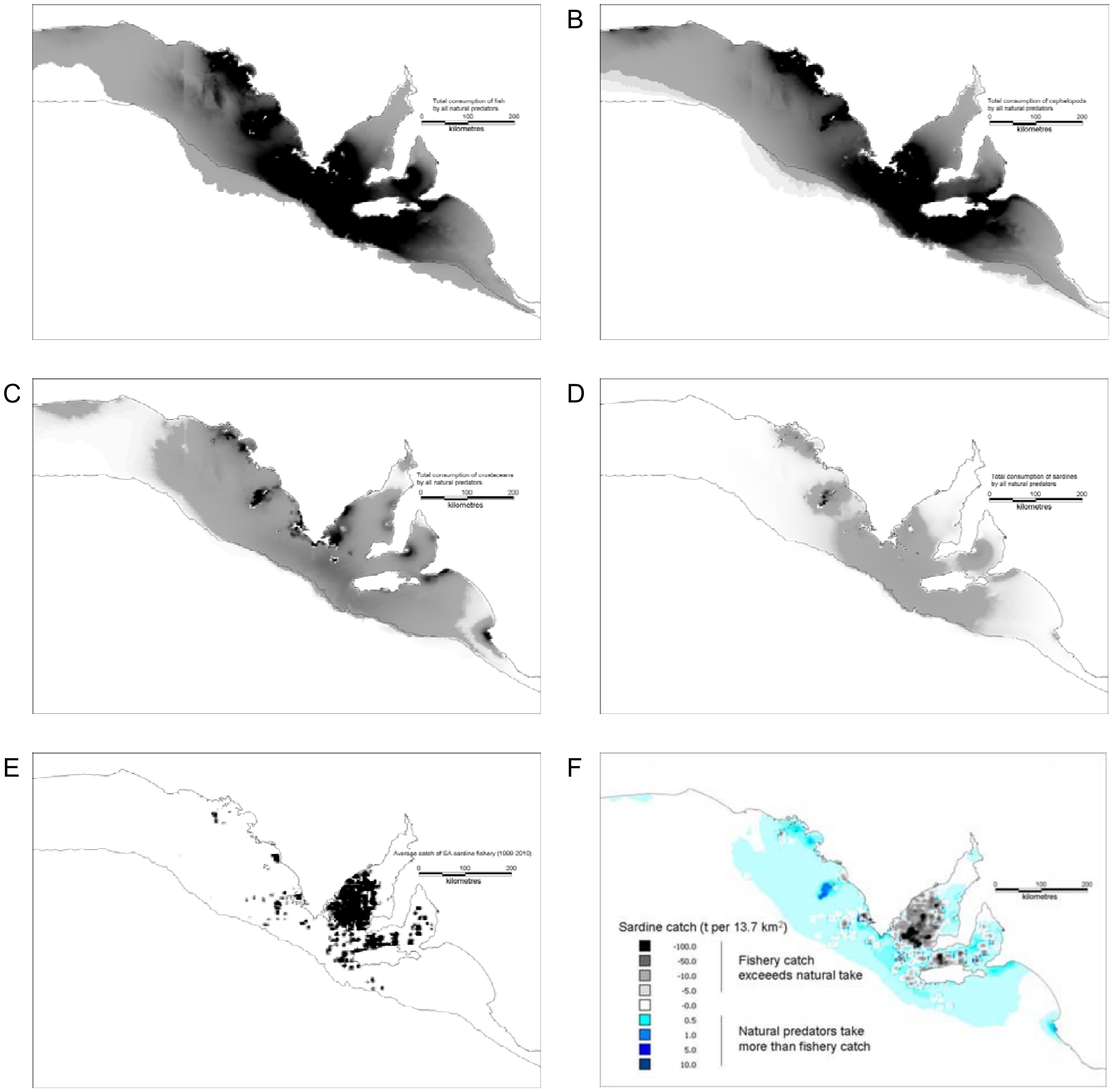

Figure 4.2. Estimated spatial distribution of consumption of fish (A), cephalopods (B), crustaceans (C) and sardines (D) (t/y) by the five key land-breeding apex predators combined; spatial distribution of the mean annual catch of sardines (1999-2010) by the SA sardine fishery (E) and a plot demonstrating the overlap in fishery catch and apex predator consumption of sardines (F).

Table 4.1. Location of breeding colonies of five land-breeding apex predators in the EGAB ecosystem, and the estimated percentage contribution that each colony represents relative to the total EGAB population of that species.

Predator	Site	lat	long	Percentage
New Zealand fur seal	Admirals Arch (Kangaroo Island)	-36.063	136.705	0.08
	Baudin Rocks	-37.086	139.724	0.02
	Berris Point (Kangaroo Island)	-36.052	137.506	6.80
	Cape Bouguer (Kangaroo Island)	-36.042	136.909	0.10
	Cape Gantheaume (Kangaroo Island)	-36.075	137.460	20.44
	Cape Gantheaume, (West of Beach)	-36.063	137.450	0.08
	Cape Hart (Kangaroo Island)	-35.900	138.048	0.01
	Cape Linois (Kangaroo Island)	-36.020	137.586	0.02
	Cave Point (Kangaroo Island)	-36.026	136.957	0.18
	Dorothee Island	-33.997	134.249	0.01
	Fenelon Island	-32.581	133.282	0.01
	Four Hummocks Island	-34.769	135.031	0.35
	Greenly Island	-34.642	134.772	0.04
	Horseshoe Bay (Kangaroo Island)	-36.038	136.928	0.02
	Knife & Steel Point (Kangaroo Island)	-36.047	136.723	0.91
	Ladders North (Kangaroo Island)	-36.058	136.707	1.57
	Ladders South (Kangaroo Island)	-36.059	136.707	0.13
	Libke (Kangaroo Island)	-36.050	136.707	4.49
	Liguanea Island	-34.998	135.620	10.88
	Little Hummock Is	-34.750	135.083	0.04
	Little Weirs & nearby (Kangaroo Island)	-36.063	137.481	0.04
	Nautilus North (Kangaroo Island)	-36.054	136.700	2.40
	Nautilus Rock (Kangaroo Island)	-36.055	136.697	0.72
	Nicolas Baudin Island	-33.016	134.133	0.01
	North Casuarina Island	-36.068	136.703	1.35
	North Neptune (East) Island	-35.230	136.068	1.00
	North Neptune (West) Island	-35.233	136.067	21.90
	Nuyts Reef (middle)	-32.139	132.141	0.01
	Olive Island	-32.719	133.970	0.01
	Pearson Island	-33.949	134.261	0.10
	Rocky Island (South)	-34.817	134.700	0.31
	South Neptune (Lighthouse) Island	-35.338	136.110	0.37
	South Neptune (Main) Island	-35.322	136.112	20.36
	South Neptune (Middle) Island	-35.335	136.112	0.76
	Spooks North (Kangaroo Island)	-36.045	136.706	2.49
	Ward Island	-33.741	134.285	0.39
	Weirs Cove North (Kangaroo Island)	-36.054	136.718	1.05
	Weirs Cove South (Kangaroo Island)	-36.055	136.719	0.52
	West Waldegrave Island	-33.596	134.762	0.01
	Xenolith Point (Kangaroo Island)	-36.024	136.977	0.01

Table 4.1. cont.

Predator	Site	lat	long	Percentage
Australian sea lion	Albatross Island	-35.069	136.181	0.48
	Black Point (Kangaroo Island)	-36.038	137.406	0.03
	Blefuscu Island	-32.462	133.639	2.70
	Breakwater Island	-32.322	133.561	0.55
	Bunda Cliffs B1	-31.518	131.061	0.48
	Bunda Cliffs B2	-31.586	130.581	0.16
	Bunda Cliffs B3	-31.582	130.126	1.00
	Bunda Cliffs B4	-31.586	130.061	0.06
	Bunda Cliffs B5	-31.585	130.031	1.38
	Bunda Cliffs B6	-31.609	129.762	0.39
	Bunda Cliffs B7	-31.625	129.511	0.10
	Bunda Cliffs B8	-31.640	129.381	1.22
	Bunda Cliffs B9	-31.647	129.311	0.55
	Cape Bouguer (Kangaroo Island)	-36.042	136.909	0.10
	Cave Point (Kangaroo Island)	-36.026	136.957	0.10
	Dangerous Reef	-34.817	136.217	22.82
	Dorothee Island	-33.997	134.249	0.03
	English Island	-34.638	136.196	0.87
	Fenelon Island	-32.581	133.282	1.29
	Four Hummocks (North) Island	-34.758	135.042	0.39
	Gliddon Reef	-32.320	133.560	0.23
	Jones Island	-33.185	134.367	0.48
	Lewis Island	-34.957	136.032	4.22
	Liguanea Island	-34.998	135.620	1.38
	Lilliput Island	-32.449	133.669	2.16
	Lounds Island	-32.273	133.366	1.09
	Nicolas Baudin Island	-33.016	134.133	3.15
	North Casuarina Island	-36.068	136.703	0.10
	North Islet	-35.117	136.470	0.90
	North Neptune (East) Island	-35.230	136.068	0.45
	North Pages Island	-35.759	138.301	8.30
	Nuyts Reef (middle)	-32.139	132.141	0.10
	Nuyts Reef (west)	-32.119	132.131	0.39
	Olive Island	-32.719	133.970	6.63
	Peaked Rocks	-35.187	136.483	0.77
	Pearson Island	-33.949	134.261	1.13
	Point Fowler (Camel-foot Bay)	-32.011	132.438	0.03
	Point Labatt	-33.152	134.261	0.19
	Price Island	-34.708	135.290	0.80
	Purdie Island	-32.270	133.228	4.25
	Rocky Island (North)	-34.259	135.261	0.51
	Seal Bay (Kangaroo Island)	-35.997	137.327	8.37
	Seal Slide (Kangaroo Island)	-36.026	137.536	0.51
	South Neptune (Main) Island	-35.322	136.112	0.19
	South Pages Island	-35.777	138.292	10.65
	Ward Island	-33.741	134.285	1.45
	West Island (Nuyts)	-32.511	133.251	1.80
	West Waldegrave Island	-33.596	134.762	5.05

Table 4.1. cont.

Predator	Site	lat	long	Percentage
Little penguin	Albatross Island	-35.069	136.181	0.12
	Althorpe Island	-35.373	136.872	0.25
	Blythe Island	-34.568	136.292	0.12
	Boston Island	-34.696	135.928	0.50
	Cape Gantheaume (Kangaroo Island)	-36.075	137.460	0.05
	Dog Island	-32.484	133.335	0.12
	Dorothee Island	-33.997	134.249	1.25
	Egg Island	-32.473	133.315	0.12
	Emu Bay (Kangaroo Island)	-35.595	137.512	0.04
	English Island	-34.638	136.196	0.12
	Evans Island	-32.376	133.484	0.50
	Flinders Island	-33.693	134.359	0.12
	Four Hummocks Island	-34.769	135.031	0.50
	Franklin Island	-32.448	133.665	5.00
	Goat Island	-32.311	133.515	0.12
	Golden Island	-34.700	135.307	0.25
	Goose Island	-34.456	137.364	0.25
	Granite Island	-35.564	138.629	0.50
	Greenly Island	-34.642	134.772	5.00
	Hareby Island	-34.582	136.295	2.50
	Harveys Return	-35.748	136.594	0.05
	Hopkins Island	-34.965	136.060	0.12
	Kingscote (Kangaroo Is)	-35.654	137.642	0.12
	Kirkby Island	-34.550	136.213	0.12
	Lacy Island	-32.397	133.372	0.25
	Lewis Island	-34.957	136.032	0.50
	Lipson Island	-34.264	136.266	0.25
	Lounds Island	-32.273	133.366	0.12
	Lusby Island	-34.542	136.260	0.12
	Masilon Island	-32.559	133.289	0.50
	North Islet	-35.117	136.470	0.12
	Olive Island	-32.719	133.970	8.98
	Outer Harbour	-34.774	138.484	0.02
	Pearson Island	-33.949	134.261	47.06
	Penguin Island	-37.498	140.014	0.15
	Penneshaw (Kangaroo Island)	-35.717	137.939	0.25
	Price Island	-34.708	135.290	0.50
	Pt Ellen (Kangaroo Island)	-35.997	137.187	0.05
	Purdie Island	-32.270	133.228	0.12
	Rabbit Island	-34.607	135.983	0.12
	Ravine des Casours	-35.792	136.600	0.04
	Reevesby Island	-34.543	136.282	4.64
	Sibsey Island	-34.645	136.182	0.25
	Smith Island	-34.984	136.029	0.12
	Smoothe Island	-32.485	133.310	0.12
	St Francis Island	-32.508	133.296	0.25
	St Peter Island	-32.285	133.582	2.50
	Thistle Island	-34.989	136.146	0.25

Table 4.1. cont.

Predator	Site	lat	long	Percentage
Little penguin	Troubridge Island	-35.118	137.828	11.76
	Waldegrave Island	-33.597	134.793	0.25
	Ward Island	-33.741	134.285	0.25
	Wardang Island	-34.488	137.343	0.25
	Wedge Island	-35.162	136.478	0.25
	West Island (Nuyts)	-32.511	133.251	0.50
	West Island (Victor Harbor)	-35.583	138.608	0.94
	West Waldegrave Island	-33.596	134.762	0.25
	Williams Island	-35.030	135.973	0.25
	Winceby Island	-34.489	136.283	0.12
Crested tern	Althorpe Island	-35.373	136.872	11.63
	Baudin Rocks	-37.086	139.724	1.94
	Bird Island	-34.130	136.349	1.94
	Bird Islands	-34.008	137.510	1.94
	Brothers Island	-34.595	135.375	7.75
	Cap Island	-33.945	135.118	3.88
	Donington Rock	-34.721	136.001	1.94
	Goose Island	-34.456	137.364	3.68
	Liguanea Island	-34.998	135.620	4.26
	Lilliput Island	-32.449	133.669	3.88
	Lipson Island	-34.264	136.266	7.75
	North Pages Island	-35.759	138.301	7.75
	Outer Harbour	-34.774	138.484	1.55
	Pigface Island	-32.695	134.278	1.94
	Rocky Island (North)	-34.259	135.261	0.39
	Rocky Island (Spencer)	-34.485	137.425	5.23
	South Neptune (Main) Island	-35.322	136.112	7.75
	South Pages Island	-35.777	138.292	1.94
	Troubridge Island	-35.118	137.828	13.57
	Ward Spit	-33.017	137.921	1.16
	West Island (Victor Harbor)	-35.583	138.608	0.39
	Wright Is	-35.583	138.608	7.75
Short-tailed shearwater	Althorpe Island	-35.373	136.872	1.82
	Curta Rocks	-34.945	135.871	0.38
	Dog Island	-32.484	133.335	0.96
	Dorothee Island	-33.997	134.249	0.15
	Egg Island	-32.473	133.315	0.03
	Evans Island	-32.376	133.484	2.39
	Four Hummocks (Central) Island	-34.769	135.031	0.15
	Four Hummocks (North) Island	-34.758	135.042	0.19
	Four Hummocks (South) Island	-34.778	135.032	0.04
	Franklin Island	-32.448	133.665	8.26
	Freeling Island	-32.481	133.345	0.01
	Goat Island	-32.311	133.515	7.67
	Golden Island	-34.700	135.307	0.27
	Greenly Island	-34.642	134.772	0.04
	Hopkins Island	-34.965	136.060	5.65
	Lacy Island	-32.397	133.372	0.38
	Liguanea Island	-34.998	135.620	1.73
	Masillon Island	-32.559	133.289	3.20
	North Islet	-35.117	136.470	0.55
	North Neptune (West) Island	-35.233	136.067	1.53
	Penguin Island	-37.498	140.014	0.06
	Perforated Island	-34.727	135.158	1.15
	Price Island	-34.708	135.290	0.69
	Smith Island	-34.984	136.029	0.08
	South Neptune (Lighthouse) Island	-35.338	136.110	1.07

Table 4.1. cont.

Predator	Site	lat	long	Percentage
Short-tailed shearwater	South Neptune (Main) Island	-35.322	136.112	0.30
	St Francis Island	-32.508	133.296	22.10
	St Peter Island	-32.285	133.582	27.10
	Topgallant Island	-33.714	134.612	0.04
	Waldegrave Island	-33.597	134.793	7.18
	Ward Island	-33.741	134.285	0.04
	Ward Island	-33.741	134.285	0.12
	Williams Island	-35.030	135.973	4.68

Table 4.2. Details on the numbers of individual apex predators instrumented with satellite transmitters and/or GPS tags, and the number of foraging trips recorded. The best model fit (normal or gamma distribution) for depth and distance parameters for each taxa group are provided. The maximum depth and distance from colony, and the mean and standard deviation (SD) and the shape and slope parameters of pooled depth and distance distributions for each taxa group based on the pertinent function are also presented. Details of the model used for ASL are presented in Goldsworthy *et al.* (2010).

	Total	Total	Model used	Depth					Distance				
Predator group	no. indiv.	no. trips	(depth-distance)	Max. (m)	Mean (m)	SD	Gamma shape	Gamma scale	Max. (km)	Mean (km)	SD	Gamma shape	Reference
Little penguin	85	85	Gamma-Gamma	92	35.6	15.2	5.468	6.502	84	14.2	13.9	1.217	Wiebkin et al. unpublished data
Crested tern	22	25	Gamma-Gamma	41	22.0	11.6	3.010	7.301	35	11.4	9.3	1.225	McLeay et al. 2010
Short-tailed shearwater	22	22	Normal-Gamma	279	79.9	27.9	-	-	188	39.2	27.0	1.370	Einoder 2010
NZ fur seal - adult female	44	101	Normal-Gamma	288	81.0	33.1	-	-	279	53.0	51.4	1.203	Page et al. 2006, Baylis et al. 2008
NZ fur seal - adult male: <200 m	20	36	Gamma-Gamma	200	96.5	34.4	7.855	12.290	471	111.6	73.4	1.875	Page et al. 2006
NZ fur seal - adult male: 200-275 m			Normal-Gamma	272	233.7	12.6	-	-	407	144.4	33.2	25.417	Page et al. 2006
NZ fur seal - adult male: >275 m			Gamma-Gamma	2841	735.6	502.7	2.141	343.560	489	181.0	70.9	2.141	Page et al. 2006
Australian sea lion - adult female	157	2340	full details in ref.	132					189				Goldsworthy et al. 2010
Australian sea lion - adult male	31	571	full details in ref.	997					339				Goldsworthy et al. 2010
Australian sea lion - juvenile	22	421	Gamma-Gamma	91			1.466	18.499	118			0.732	Goldsworthy et al. 2010

Table 4.3. Estimated total annual consumption and the total annual consumption in the EGAB ecosystem, including a breakdown of prey into the following categories: small pelagic fish, sardines, all fish, cephalopods and crustaceans. All estimates in t/y are derived from diet, population biomass and consumption estimates from Chapter 5.

	Consumption t/y							
Land breeding apex predator	Total consumption of predators from SA colonies	Total consumption in EGAB	Small pelagic fish	Sardine	Bird	Fish	Cephalopod	Crustacean
NZ fur seal	33163	23284	6527	361	153	14059	9059	13
Aust sea lion	19150	19150	179	19	8	5069	13177	897
Little penguin	9205	9205	6917	325	0	8903	251	51
Short-tailed shearwater	69434	12775	4207	28	0	6387	2631	3756
Crested terns	88	88	56	20	0	87	1	0
Total	131041	64502	17886	753	160	34506	25120	4717

5 TROPHODYNAMICS OF THE EASTERN GREAT AUSTRALIAN BIGHT PELAGIC ECOSYSTEM: IMPLICATIONS FOR ASSESSING THE ECOLOGICAL SUSTAINABILITY OF AUSTRALIA'S LARGEST FISHERY

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Introduction

The Great Australian Bight (GAB) region between Cape Leeuwin (Western Australia) and Portland (Victoria) lies along the world's longest south-facing, mid-latitude shelf and is host to the only northern boundary current system – the Flinders Current (Middleton & Cirano 2002, Ward *et al.* 2006, Middleton & Bye 2007) (Figure 5.1). The dominant oceanographic feature of the Flinders Current is coastal upwelling that occurs in summer/autumn (November-April) in the eastern GAB (EGAB), especially off the Bonney Coast, Kangaroo Island and southern and western Eyre Peninsula (McClatchie *et al.* 2006, Ward *et al.* 2006, Middleton & Bye 2007). The Flinders Current has oceanographic, biological and ecological similarities to eastern boundary currents that underpin the Benguela, Humbolt, California and Canary Current upwelling systems (Mann & Lazier 1996, Middleton & Platov 2003, Ward *et al.* 2006).

During winter and spring (May-October), westerly winds favour downwelling ,and in association with coastal cooling, water generally becomes well mixed over the shelf and the influence of the Leeuwin Current from the west ensures a continuous eastward current over the continental shelf (Kämpf *et al.* 2004, Middleton & Bye 2007, van Ruth *et al.* 2010). Downwelling and mixing events, coupled with low irradiances and short day-length, result in low productivity (~300-550 mg C m^{-2} d^{-1}) (van Ruth *et al.* 2010). However, during summer and autumn (November-April), south-easterly winds favour upwelling and assist the movement of water from the Flinders Current (~250m depth) onto and across the shelf (Kämpf *et al.* 2004, Middleton & Bye 2007). Upwelling originates to the south and south-east of Kangaroo Island, is largely sub-surface and is directed along the 100 m isobath to the north and north-west (Middleton & Bye 2007). Stratification of the water column with cold water upwelling beneath a shallow surface mixed layer, promotes high rates of primary productivity (up to 2,958 mg C m^{-2} d^{-1}) (McClatchie *et al.* 2006, van Ruth *et al.* 2010). The high degree of seasonality in meteorological and oceanographic conditions that drive variations in mixing and stratification result in a highly dynamic, temporally variable marine ecosystem (van Ruth *et al.* 2010).

The seasonal upwelling boosts primary, secondary and fish production, making the EGAB Australia's richest pelagic ecosystem (Ward *et al.* 2006), and an ecological 'hot-spot' of international significance. The region supports the highest densities of small pelagic fishes in Australian waters, the dominant species being sardine (*Sardinops sagax*) and anchovy (*Engraulis australis*) (Ward *et al.* 2006). Other important small pelagic fish species include blue mackerel (*Scomber australasicus*), jack mackerel (*Trachurus declivis*), redbait (*Emmelichthys nitidus*), maray (*Etrumeus teres*) and saury (*Scomberesox saurus*). These rich pelagic resources also underpin arguably the greatest density and biomass of apex predators to be found in Australian coastal waters. These include marine mammals such as pygmy blue whales (*Balaenoptera musculus brevicauda*), and >80% of Australia's populations of New Zealand fur seals (*Arctocephalus forsteri*) and Australian sea lions (*Neophoca cinerea*) (Goldsworthy *et al.* 2003, Branch *et al.* 2007, Goldsworthy *et al.* 2009), and a recently established breeding population of the Australian fur seal (*A. pusillus doriferus*) (Shaughnessy *et al.* 2010). All seal species were subjected to early colonial sealing, with recovery of fur seal populations commencing in the 1970s and 1980s and still continuing (Ling 1999) . Other key apex predators include seabirds, such as short-tailed shearwaters (*Puffinus tenuirostris*) (~1.3 million pairs breed in the EGAB), little penguins (*Eudyptula minor*) and crested terns (*Sterna bergii*); pelagic sharks including bronze and dusky whalers (*Carcharhinus brachyurus, C. obscurus*), great white (*Carcharodon carcharias*) and shortfin mako (*Isurus oxyrinchus*); and predatory fishes such as southern bluefin tuna (SBT, *Thunnus maccoyii*) (Copley 1996, Goldsworthy *et al.* 2003, Ward *et al.* 2006, Goldsworthy & Page 2007).

The variable nature of upwelling and seasonal production in the EGAB presents significant challenges for apex predators and there appears to be a range of foraging strategies that species adopt in response to the variable states of the ecosystem (Chapters 3 and 4). Species such as pygmy blue whales, short-tailed shearwaters and southern bluefin tuna make conspicuous migrations into the region during the upwelling season. Resident species such as New Zealand fur seals form large breeding colonies in close proximity to areas of enhanced seasonal productivity, foraging over shelf waters during the productive upwelling season, then shifting their foraging effort to oceanic waters of the subtropical front during winter and spring (downwelling season) (Page *et al.* 2006, Baylis *et al.* 2008 a,b, Chapter 4).

The EGAB supports some of Australia's most valuable fisheries, including four main Commonwealth and five main South Australian (State) managed fisheries (Wilson *et al.* 2009, Knight & Tsolos 2010). The main Commonwealth fisheries that operate are the GAB Trawl (GABT), South East Trawl (SET), southern bluefin tuna (SBT) and shark gillnet component of the Gill Hook and Trap fishery (GHAT) (Wilson *et al.* 2009). The main South Australian fisheries that operate in the region are the sardine, southern rock lobster, abalone, western king prawn and marine scalefish (MSF) fisheries (Knight & Tsolos 2010). By weight, the South Australian sardine fishery (SASF) is Australia's largest fishery. It was established in 1991 to provide feed for the SBT mariculture industry in Port Lincoln (Ward *et al.* 2005) (Figure 5.2). Sardines are taken by purse seine; the fishery is centred on southern Spencer Gulf, Investigator Strait and the western Eyre Peninsula (Ward *et al.* 2005). Spawning biomass of sardine in SA has been estimated using the daily egg production method (DEPM) since 1995, when the spawning biomass was estimated to be ~165,000 t (Ward *et al.* 2009b). However, it declined by over 70% to ~37,000 t in 1996 following an unprecedented mass mortality event, recovered to ~146,000 t in 1998 and then declined by over 70% again to ~36,000 t in early 1999 following a second mass mortality event (Ward *et al.* 2001 a,b). Between 1994 and 2001, fishery catches remained between 2,500 and 6,500 t each year, then steadily increased to ~39,000 t in 2005. Since then, the harvest strategy has been to maintain a baseline total allowable catch (TAC) of 30,000 t, while estimates of the spawning biomass using the DEPM remain between 150,000 and 300,000 t, corresponding to an exploitation rate of between 20% and 10%, respectively (Ward *et al.* 2009b).

Provisions of Australia's *Environment Protection and Biodiversity Conservation Act* (*EPBC Act*), require strategic assessment and, if necessary, mitigation of the ecological effects of fishing, including trophic impacts. Strategic assessment of the SA sardine fishery identified the need to measure and minimize the impacts of the fishery on the broader ecosystem and to review the current ecological management objectives, management strategies and performance indicators. In recognition of the economic importance of the SA sardine fishery, the important role of sardines and other small pelagic fishes in underpinning ecological processes in the EGAB ecosystem, and the high socio-economic and conservation significance of the region's marine predators, the fishing industry and fishery managers have identified the need to establish world's best practices for managing the potential ecological impacts of the fishery. In response to this need, a large-scale ecosystem project was initiated to assess the role of sardines in the EGAB ecosystem and to develop ecological performance indicators and reference points for the fishery. This included an understanding of the diets and trophic guild-structure of key marine predator and prey species (see Chapter 3) in order to assess the significance of sardines in the ecosystem and identify potentially dependent species. Land-breeding apex predators such as seals and seabirds that prey on the mid-trophic levels integrate information about prey through changes in their foraging behaviour and diet, reproductive performance and ultimately over longer times-scales through changes in population size and distribution. The suitability of these accessible marine predators as ecological performance indicators for the fishery was assessed by determining if their foraging ranges and prey consumption overlapped significantly with the sardine fishery (see Chapter 4), and if measures of their diet, foraging and/or reproductive success were sensitive to changes in the biomass and catch of sardines (see Chapter 6).

The aims of this study were to: 1) develop a food web model for the EGAB and describe its temporal dynamics over the period since the SA sardine fishery established in 1991 using time-series data of fishing activity and environmental drivers; 2) identify the trophic interactions and functional groups most sensitive to variability in sardine biomass; 3) determine in particular, the sensitivity of land breeding apex predators to changes in sardine biomass and fishery catch, and assess their appropriateness as ecological performance indicators for the sardine fishery; 4) examine ecosystem change through time using ecosystem indicators and assess their potential as ecological performance indicators of ecosystem health; 5) examine the trophodynamic changes occurring in response to changes in apex predator biomasses, including recovering fur seal populations and declining SBT populations and 6) examine how the recovery of these groups is likely to impact on the EGAB ecosystem and the SA sardine fishery.

Currently, there is no scientific framework to assess whether the management arrangements that have been established for the SA sardine fishery are sufficiently conservative to ensure that the fishery is managed according to the principles of ecological sustainable development (ESD) (i.e. that fishing does not significantly affect the status of other components of the ecosystem, Fletcher *et al.* 2002). The principal aim of this study is to provide an ecosystem perspective of the SA sardine fishery, by placing its establishment and growth in the context of other dynamic changes in the ecosystem, including changes to other fisheries, changes to apex predator populations, and meteorological and oceanographic changes. Such information is essential to enhance the management and sustainability of Australia’s largest pelagic fishery by incorporating scientifically-based approaches for assessing and, if necessary, mitigating the fishery's trophic impacts.

Materials and methods

Ecopath and mass balance approach

We used the *Ecopath with Ecosim* software (www.Ecopath.org) to develop a trophic mass-balance model of the EGAB ecosystem. *Ecopath* was developed by Polovina (1984), based on a simple steady-state trophic box model, and further developed by Christensen and Pauly (1992) and Walters *et al.* (1997). Whereas *Ecopath* enables description of the static state energy flow of an ecosystem at a particular point in time, *Ecosim* enables dynamic simulations based on *Ecopath* parameters that allow the forecasting of ecosystem response to environmental perturbations. The *Ecopath with Ecosim* software has now been used to describe a diverse range of aquatic ecosystems world-wide, and details of the ecological theory and mathematical equations that underpin its key functions have been extensively detailed elsewhere (e.g. Christensen & Walters 2004, Shannon *et al.* 2008, Griffiths *et al.* 2010, Piroddi *et al.* 2010).

Model area and structure

This study describes the EGAB pelagic ecosystem, a region off the South Australian coast that includes continental shelf waters to 200m depth between 132° and 139.7° longitude. It includes the Investigator Strait and the lower portions of Gulf St Vincent and Spencer Gulf; a region of about 154,084 km^2 (Figure 5.1). The modelled area was selected based on knowledge of the distribution and abundance of sardine and other small pelagic fish species derived from Small Pelagic Fish ichthyoplankton surveys between 1995 and 2010 (Ward *et al.* 2007, Ward *et al.* 2009b).

As the aims of the model were to investigate the potential impacts of the sardine fishery on high tropic level predators, especially land-breeding seals and seabirds, these groups were disaggregated into single species where data on diet and population biomass permitted. As we were also interested in ecological effects of fishing on the key target species (sardines), the functional roles and importance of mid-trophic small pelagic species, such as sardines, anchovies, jack mackerel, redbait, arrow squid and calamary, were also of interest, and these species were also examined as single-species functional groups where data permitted.

A total of 40 functional groups were developed in the EGAB ecosystem model, based on species similarity in terms of diet, habitat, foraging behaviour, size, consumption and rates of production (Appendices 3 and 4). Intrinsic to *Ecopath* model development, each trophic group operates as a single biomass, despite groups often being composed of several species. The aggregation of species into trophic groups will therefore impact on model dynamics in some instances, however, by matching species for diet, consumption, and production rates we attempted to constrain the errors and uncertainty of aggregating.

An *Ecopath* with *Ecosim* model (software version 6.1) was constructed for the EGAB ecosystem for 1991, when the South Australian sardine fishery commenced. Model simulations were run over the following 18 years until (1991 - 2008). For each trophic group, four key parameters were estimated: diet, biomass, production per unit of biomass (P/B) and consumption per unit of biomass (Q/B). A dietary matrix was constructed using empirical data derived within the EGAB ecosystem where possible. For marine mammals, seabirds, large and small pelagic fishes, arrow squid and calamary, dietary data summarised by Page *et al.* (Chapter 3) was used (Table 5.1). For most other fish species in the region, data from Currie and Sorokin (2010) was used. Where possible, the biomasses (t km^{-2}) of functional groups were estimated either from field surveys or stock assessments. A detailed description of the functional groups and how estimates of biomass, P/B and Q/B were derived is presented in Appendices 3 and 4.

Fishery data on landings, discards and effort were obtained and broken down into eleven fisheries (fleets) operating within the EGAB ecosystem (Table 5.2, 5.3). Six South Australian managed fisheries: the South Australian (SA) sardine, SA Marine Scalefish line, SA Marine Scalefish net, and three prawn fisheries (Spencer Gulf; Gulf St Vincent; West Coast); and five fisheries managed by the Australian Government: SBT purse seine, SBT pole and bait, South East trawl, GAB trawl, and the gillnet demersal shark fishery (part of the Gillnet Hook and Trap Fishery). Fleet data from the SA abalone and southern rock lobster fisheries were not included because the EGAB ecosystem model was primarily developed to be a pelagic model. Annual fishery landings and effort data were obtained for all fleets between 1991 and 2008 (logbook data obtained from the Australian Fisheries Management Authority, CSIRO Marine and Atmospheric Research and SARDI Aquatic Sciences). Retained and discarded catch data were typically only available for between 1 and 3 years for each fishery, and were estimated for 1991 based on their proportion to landed catch (Currie *et al.* 2009, Fowler *et al.* 2009, Roberts & Steer 2010). All landed and discarded species were assigned their functional group, and biomasses summed at the functional group level (t km^{-2}) (Tables 5.2 and 5.3, respectively). Time series of annual catch and catch per unit-effort (CPUE) could be estimated for 18 and 16 functional groups, respectively, and fishing mortality (F) was used for sardines instead of CPUE because marked changes in vessel size in the fleet between 1991 and 2008 confounded CPUE estimates.

Model fitting

Dynamic simulations were run in *Ecosim* using the time-series estimates (1991-2008) of biomass and fishing mortality (F) (sardines only), catch and CPUE for functional groups with available data. Several *Ecosim* scenarios were explored through adjustment of predator prey vulnerability using the 'fit to time series' procedure. Different numbers of predator-prey interactions within the dietary matrix were selected (10-90) within this procedure to identify the most sensitive and optimal number of predator-prey interactions, and their vulnerability values that would minimise the model sum of squares (SS) and produce the best fit to the time series data. Some of the default *Ecosim* parameters were then adjusted to further decrease the model SS. This included adjusting the maximum relative feeding time of marine mammals and seabirds from 2.0 (default) to 10.0, and their feeding time adjustment rates to 0.5 (0 for all other groups), to account for modifications to their search feeding times in response to changes in prey availability (Christensen *et al.* 2008). Similarly, we adjusted density-dependent predator-prey switching power of the dolphin and seal groups from 0 to 2.0, to account for their capacity to opportunistically adjust their diet in response to changes in prey availability (Piroddi *et al.* 2010). We also explored improvements to model fits by adjusting values of density-dependent changes in catchability for pelagic schooling fish such as sardines and tuna (Christensen *et al.* 2008, Piroddi *et al.* 2010), but these did not produce improvements to the model fits.

The final step of the model fitting procedure was to link the best combination of vulnerability values and the time-series data to an estimated trend in primary productivity (PP) within the EGAB ecosystem. The importance of seasonal upwelling in the EGAB to enhancing primary, secondary and fish production has been established (Ward *et al.* 2006). As such, time-series of mean monthly upwelling anomalies, calculated from the alongshore component of wind stress at South Neptune Island (Middleton *et al.* 2007), were considered as potential PP forcing functions (FF) in the EGAB ecosystem. Wind stress values between 1991 and 2008 were scaled to positive values, and then converted to values relative to that of January 1991. Using the anomaly search function within *Ecosim*, we set the PP variance to 50 to enable the model to capture changes in the magnitude of 'spikes'. The number of spline points was set to 16, the number of time-series to be fitted in the model. Model estimated FFs were compared to the raw upwelling index time series to assess if they reduced the model SS.

Ecosystem indicators

After the model fitting procedure in *Ecosim*, we examined four ecosystem indictors that can be used to evaluate changes in the marine ecosystem. 1) Total catch; 2) Kempton's index of biodiversity (*Q*), which expresses biomass species diversity of functional groups with a trophic level (TL) of 3 or higher (Kempton & Taylor 1976, Ainsworth & Pitcher 2006); 3) the mean trophic level of the catch (mTLC) which is calculated as the weighted average of the TL of fishery targeted species (Pauly *et al.* 1998); and 4) the Fishing in Balance Index (FIB index), which assesses whether catch rates are in balance with ecosystem trophic production due to catch at a given TL being related to the assimilation efficiency of the ecosystem (Coll *et al.* 2009). The FIB index will remain constant if a decline in mTLc is matched by an ecologically appropriate increase in catch, and conversely for increasing trophic level (Pauly & Palomares 2005). In general, the index increases if the underlying fishery expands beyond its traditional fishing area or ecosystem, and decreases if the geographic area contracts, or if the underlying food web is collapsing (Pauly & Palomares 2005).

Model scenarios

We used the fitted *Ecosim* EGAB model without PP forcing to explore the ecosystem response to reductions in sardine biomass. Simulated reductions in biomass were achieved by adjusting the time series to year 2040, and maintaining the fishing effort of all fleets at 2008 levels, with the exception of the sardine fishery where changes in effort were used to drive reductions in sardine biomass. The time series of sardine fishing mortality was deselected in these procedures. In these scenarios, both NZ and Australian fur seal biomass was forced to increase between 2008 and 2040 (in line with observed increases between 1991 and 2008), so that population abundances in the EGAB were about 1.5 and 7.0 times present levels (2008), respectively. This is equivalent to an increase from ~73,000 to ~111,000 for NZ fur seals and ~2,800 to ~20,000 for Australian fur seals between 2008 and 2040. In addition, biomass increases in SBT were simulated by reducing effort in the purse seine fishery by adjusting time series values between 2008 and 2040. For all scenarios, relative change in biomass between 2008 and 2040 was examined by dividing the end biomass by the starting biomass.

Results

Trophic structure and flow

The basic parameters used to inform the 40 functional groups within the *Ecopath* model are presented in Table 5.4, those in bold represent parameters estimated by *Ecopath*. The balancing procedure required adjustment to the diets of some groups where ecotrophic efficiencies (EE) were initially >1. EE is the proportion of production that is either harvested or predated upon by higher trophic levels and cannot exceed 1. The main dietary adjustments included reduction in the contribution of little penguins and petrels in the diets of NZ fur seals, and of petrels in the diets of Australian sea lions; reduction in the contribution of large bentho-pelagic fish in the diet of arrow squid; reduction in the contribution of medium demersal invertebrate feeders in the diets of demersal sharks and octopus; and reductions in the contribution of small demersal omnivores in the diets of salmon and ruffs and of medium demersal piscivorous fish. The trophic flows between the functional groups in the EGAB ecosystem estimated by *Ecopath*, are summarised in Figure 5.3.

The trophic level of the functional groups ranged from 1 to 5.02, with the highest values for seals, dolphins, seabirds, pelagic sharks, SBT, other tuna and kingfish, large bentho-pelagic fish, salmon and ruffs, arrow squid, calamary, octopus and medium demersal invertebrate feeders (TL ≥ 4) (Table 5.1, Figure 5.3). Demersal sharks, rays and skates, and medium demersal piscivorous fish had trophic levels ranging between 3.47 and 3.92. The small pelagic fish group (sardines, anchovy, blue mackerel, jack mackerel, redbait and inshore small planktivores) had trophic levels ranging between 3.23 and 3.93. All other fish groups, other cephalopods, and baleen whales had trophic levels ranging between 2.66 and 3.77 (Table 5.1, Figure 5.3).

The mixed trophic impacts routine in the network analysis tools within *Ecopath* was used to evaluate critical trophic interactions between groups in the ecosystem (Figure 5.4). The routine is based on the method developed by Ulanowicz and Puccia (1990), and allows the computation of direct and indirect impacts which a change in biomass of a predator group will have on other groups in the system, assuming that the diet matrix remains unchanged, and may thus be viewed as a tool for sensitivity analysis. Some key results from the routine indicate the positive effect of primary production on small and large zooplankton, the positive effects of these groups on small pelagic fishes which in turn have a positive effect on dolphin, seal and seabird groups (Figure 5.4). Groups containing commercially targeted species influenced their respective fishing fleets positively, and most groups affected themselves negatively (Figure 5.4). Cephalopod groups affected small demersal invertebrate feeders and meso-pelagic fish negatively; salmon and ruffs affected medium and small demersal piscivorous fish and large negatively; and bentho-pelagic piscivorous fish affected small pelagic fish and octopus groups negatively (Figure 5.4). SBT affected other tunas and kingfish negatively; demersal sharks affected medium demersal invertebrate feeders, pelagic sharks and sea lions negatively; and sea lions affected rays and skates and little penguins negatively; NZ fur seals affected little penguins and petrels negatively (Figure 5.4). Of the fisheries, the SA marine scalefish net fishery affected pelagic and demersal sharks negatively; the SBT purse seine fishery affected SBT, other tuna and kingfish negatively; the demersal shark fishery affected sea lions and demersal sharks negatively, and the Spencer Gulf prawn fishery affected rays and skates negatively (Figure 5.4). Based on this analysis, the sardine fishery has negligible impacts on other groups. However, as a steady-state analysis, all these assessments do not take into account the changing abundances or diets of groups. Such dynamics are explored in *Ecosim* scenarios below.

The proportional breakdown of the consumption by predators of sardines and of all combined small pelagic fish groups is presented in Figure 5.5. The key predators of sardines in ranked order were large bentho-pelagic fish, arrow squid, salmon and ruffs, SBT, other tuna and kingfish, common dolphins and pelagic sharks. For all small pelagic fish, the key predators in ranked order were salmon and ruffs, arrow squid, SBT, calamary, other tuna and kingfish, common dolphins, little penguins, NZ fur seals, petrels and pelagic sharks (Figure 5.5). The total consumption of sardines estimated by the *Ecosim* model was 123,369 t y^{-1}, representing about a third (31%) of the total combined consumption of small pelagic fish (396,129 t y^{-1}) in the EGAB ecosystem.

Time-series fitting

Ecosim model fits to fishery time series and model-derived wind-stress anomaly forcing functions (Figure 5.6) are presented in Figure 5.7 There was a high degree of variability in the degree to which *Ecosim* reproduced the CPUE (biomass) and yield (catch) trends of various functional groups (Figure 5.7). Trends in modelled biomass fitted observed trends reasonably well for pelagic sharks, demersal sharks, large bentho-pelagic piscivorous fish, sardine, medium demersal piscivorous fish, and calamary (Figure 5.7). Trends in modelled catch fitted observed trends reasonably well for pelagic sharks, demersal sharks, SBT, large bentho-pelagic piscivorous fish, sardine, salmon and ruffs, medium demersal piscivorous fish, calamary and benthic grazers (Figure 5.7). In many instances the fit of modelled trends to observed trends was substantially better for one of biomass or catch over the other. For example for SBT, sardine, salmon and ruffs, and arrow squid, the fit to catch was much better than the fit to biomass; whereas for pelagic sharks, rays and skates, medium demersal piscivorous fish and calamary, the fit to biomass was much better than the fit to catch (Figure 5.7). The latter instances were predominantly for functional groups caught across multiple fishing fleets, and because *Ecosim* uses only one fleet's CPUE time series to estimate biomass (but uses all fleet effort data to estimate catch), such variability in fits is not surprising. For SBT, the capacity for *Ecosim* to fit to changes in biomass was confounded by the transition from pole and bait as the main fishing method to purse seining, with the pole and bait fishery ending in 2000. CPUE from the purse seine fishery was used to drive biomass trends for this group. The congruence between modelled and observed trends in biomass and catch was best for functional groups composed of species targeted by fisheries, such as demersal sharks (gummy and school shark), SBT, large bentho-pelagic fish (snapper), sardine, medium demersal piscivorous fish (King George whiting), calamary and benthic grazers (western king prawn); and poorer for non-targeted (byproduct) species such as other tuna and kingfish, blue and jack mackerel and small demersal piscivorous fish (Figure 5.7).

A summary of the estimated changes in the biomass of the EGAB ecosystem between 1991 and 2008 summarised into nine basic groups, with and without PP forcing in the *Ecosim* model, is presented Table 5.5. With no change in PP, the observed increase in high trophic predators (fur seals) over the period occurs to the detriment of large fish (SBT and other tuna and kingfish mainly due to over fishing) and of medium and small fish. However, the CPUE/biomass trends for many functional groups (pelagic shark, demersal shark, large bentho-pelagic fish, sardine, salmon and ruffs, medium demersal piscivorous fish, arrow squid, calamary and benthic grazers, Figure 5.7) suggest that changes in fishing effort alone are not enough to capture the observed biomass trends in these groups, and that the best model fits with minimum SS resulted from PP forcing, suggesting that an overall increase in PP is also required to support the observed biomass increases in many of the groups between 1991 and 2008 ($P<0.0001$, Table 5.5, Figure 5.7).

Temporal changes in group biomass

Trends in landings of the eight main fisheries between 1991 and 2008 are presented in Figure 5.2. It demonstrates the significant growth of the sardine fishery, a 313,141% increase between 1991 and 2008 (linear regression, $P < 0.001$), with the major growth period occurring since 2000. Landings of SBT increased by 75% between 1991 and 2008 (linear regression, $P = 0.007$), while landings from the marine scalefish net and demersal shark fisheries declined by 88% and 35%, respectively (linear regression, $P < 0.001$ and $P = 0.0015$, respectively). There has been little change in the landings from the marine scalefish line fisheries (+7%, linear regression, $P = 0.1750$) and western king prawn (-1%, linear regression, $P = 0.741$, three regions combined); although GAB trawl fishery landings declined by 35%, the decrease was not significant (linear regression, $P = 0.085$).

The trends in apex predators and small pelagic species biomass between 1991 and 2008, as estimated by *Ecosim,* are presented in Figure 5.8. These results suggest significant increases in population sizes of NZ fur seals, Australian fur seals, Australian sea lions, bottlenose dolphins and common dolphins (linear regression, $P<0.0001$), and no significant change in the biomass of little penguins and petrels between 1991 and 2008 (Figure 5.8a). In addition, pelagic and demersal shark showed significant increases in biomass (linear regression, $P<0.0001$) in response to reductions in fishing mortality as a result of reduction in effort in the marine scalefish net fishery and the demersal shark fishery between 1991 and 2008 (Figure 5.8b). Rays and skates were also estimated to increase but predicted trends are at odds with CPUE data, especially between 2002 and 2008 (Figure 5.7). Large bentho-pelagic piscivorous fish are also estimated to have increased in biomass (linear regression, $P<0.0001$), while the biomass of SBT and other tuna and kingfish groups declined (linear regression, $P<0.0001$), principally in response to high fishing mortality. All the small pelagic fish groups (blue mackerel, jack mackerel, redbait, anchovy, sardine, inshore small planktivores, salmon and ruffs) and squids (arrow squid, calamary) were estimated to increase in biomass between 1991 and 2008 (linear regression, $P<0.0001$) (Figure 5.9).

Ecosystem indicators

The ecosystem indicators identified significant changes in the EGAB ecosystem between 1991 and 2008 (Figure 5.10 a-d). Total catch of the combined fisheries showed a 4.6 fold increase, attributed to growth in the sardine fishery (Figure 5.10a). The total catch of all other fisheries combined showed a slight (0.89) but non-significant reduction over the same period (Figure 5.10a). Kempton’s Q biodiversity index usually increases with growing biomass of high trophic level species, and decreases with increased fishing impacts. Results for the EGAB ecosystem indicated that the Kempton’s Q index increased significantly between 1991 and 2008 (Figure 5.10b), commensurate with increasing biomasses of marine mammals, sharks, and some large piscivorous fish and squid (Figures 5.7 and 5.8). The mTLC decreased significantly between 1991 and 2008 (linear regression, $P < 0.001$) (Figure 5.10c). Typically, reductions in mTLC are attributed to increased fishery impacts as a consequence of a reduction in high trophic level predators relative to lower trophic level organisms. In the EGAB ecosystem between 1991 and 2008, reductions in the mTLC are attributed to the significant growth in the small pelagic fishery for sardines, and not to a reduction in high trophic level biomass relative to the lower trophic levels. There was a general increasing trend in the fishing in balance (FIB) index between 1991 and 2008 (but not significant) and all values were >3.5 (Figure 5.10d). An FIB index < 0 occurs where fishing impacts are so high that the ecosystem function is impaired, or where discarding occurs and it is not considered in the analysis (Christensen 2000). An FIB index = 0 indicates high production at lower trophic levels with fishing in balance; and an FIB index > 0 indicates an expansion of fishing and/or where bottom-up effects are occurring, resulting in more catch than expected (Coll *et al.* 2009).

Model scenario results

Results of the *Ecosim* scenarios out to 2040 with the biomass of sardines reduced to 0.75, 0.50 and 0.25 of the 2008 biomass, are presented in Figure 5.11. In these scenarios the model was initialised with the biomass of NZ fur seal and Australian fur seal set to increase each year, in line with observed rates of increase between 1991 and 2008. Functional groups that were negatively impacted either directly or indirectly by reductions in sardine biomass, where biomass fell below 2008 levels included: common dolphins, little penguins, gannets, terns, SBT, other tunas and kingfish, small demersal invertebrate feeders, mesopelagics, and arrow squid (Figure 5.11). For some declining groups, biomass was positively affected by reductions in sardine biomass (e.g. petrels and jack mackerel) (Figure 5.11). For some groups with increasing biomass, biomass was either negatively (e.g. NZ fur seals, Australian sea lions, pelagic sharks, and salmon and ruffs);,or positively (e.g. bottlenose dolphins, Australian fur seals, redbait, anchovy, small demersal omnivores, other squids and octopus) affected by reductions in sardine biomass (Figure 5.11).

Of the land-breeding marine predators, those most directly impacted by reductions in sardine biomass were terns, which reduced in biomass by 9%, 20% and 35% in response to 25%, 50% and 75% reductions in sardine biomass, respectively; and gannets which reduced in biomass by 6%, 17%, 27% (Figure 5.11). Little penguin decline was less pronounced with declines of 12.0%, 12.5% and 12.9% in response to 25%, 50% and 75% reductions in sardine biomass, respectively. Rates of increase in NZ fur seals (65%, 61%, and 56%) and Australian sea lions (28%, 20% and 14%), were lower, respectively in response to 25%, 50% and 75% reductions in sardine biomass, due mainly to indirect interactions reducing the biomass of arrow squid and large bentho-pelagic piscivorous fish, respectively (Figure 5.11).

Ecosim simulations suggest that maintenance of current (2008) fishing effort on SBT will result in a further 21% decline in biomass between 2008 and 2040. This is on top of the estimated 51% decline in biomass between 1991 and 2008. *Ecosim* scenarios to rebuild SBT biomass were explored by reducing current (2008) levels of fishing effort by 50%, 75% and 100% (Figure 5.12). These scenarios resulted in increases in SBT biomass of 45%, 148% and 615% between 2008 and 2040, respectively (Figure 5.12). They also resulted in marked increases in the biomass of the other tuna and kingfish group (58%, 199% and 1159%, respectively). Recovery of SBT appears to enhance the biomass of pelagic sharks, but does not have major positive or negative effects on any other groups (Figure 5.12). Although the consequences are not examined here, the global quota of southern bluefin tuna was coincidentally cut by 20 per cent for the 2010 and 2011 fishing seasons while this report was being written.

A summary of the estimated changes between 2008 and 2040 in the biomass of the EGAB ecosystem summarised into nine basic groups, with and without fishing effort on SBT, and with and without PP forcing is presented Table 5.5. As in the above scenarios, both NZ and Australian fur seal biomass was forced to increase to 1.5 and 7.0 times present (2008) levels. These simulations indicated that the recovery of apex predators (marine mammals, birds and pelagic sharks) and of large pelagic fish such as SBT is supported through reductions in the biomass of small pelagic fish and cephalopods (Table 5.5). The main small pelagic fish group impacted is jack mackerel, with negligible change in other small pelagic species, while the most negatively impacted cephalopods include arrow squid and calamary (Figure 5.13). The potential role of reduced competition for small pelagic fish by a reduction in biomass of arrow squid was explored by gradually increasing fishing mortality in the time series between 2008 and 2040, while excluding SBT fishing effort and enabling fur seals to recover as above.

Results from scenarios where arrow squid biomass was reduced by 50% and 75% are presented in Figure 5.13, and relative changes in the biomass of the EGAB ecosystem summarised into nine basic groups between 2008 and 2040 are presented in Table 5.5. Results indicate that for some high trophic-level predator groups which increased in biomass over the 2008-2040 period and which predate significantly on arrow squid (NZ fur seals, pelagic sharks, SBT, and other tunas and kingfish), rates of increase were reduced commensurate with reductions in arrow squid biomass. However, most other high trophic-level predators responded positively to reductions in arrow squid biomass, including common dolphins, Australian fur seals, Australian sea lions, little penguins, petrels, gannets, terns, and demersal sharks (Figure 5.13). All the small pelagic fish species (with the exception of inshore small planktivores), large bentho-pelagic piscivorous fish and salmon and ruffs also responded positively to reductions in arrow squid biomass (Figure 5.13). Comparison between the scenarios of increasing fur seal biomass with removal of SBT fishing effort, and the same scenario where arrow squid biomass was reduced by 50% (last two columns of Table 5.5) highlight the significant predation pressure that cephalopods (especially arrow squid) likely place on small pelagic fish, and how with reduced arrow squid biomass, greater production of small pelagic fish can be directed into higher trophic levels (in this instance, the increased production of small pelagic fish was directed into large piscivorous fish and other apex predator species (Table 5.5)).

Discussion

This study presents the first trophic model of the EGAB ecosystem. Elements of the ecosystem include an abundance of high trophic level predators and small pelagic fish, the latter dominated by sardine and anchovies. This ecosystem has many similarities with other sardine/anchovy dominated eastern boundary current upwelling ecosystems which are situated in the eastern Atlantic and Pacific Oceans, but differs in being underpinned by a unique northern boundary current system, the Flinders Current (Middleton & Cirano 2002, Middleton & Bye 2007). The region is economically significant for the fishing and aquaculture industry, including Australia’s largest volume fishery for sardines, which underpins the SBT grow-out (aquaculture) industry based at Port Lincoln. During the growth of the SBT aquaculture industry and the sardine fishery, there have been other significant ecological and fishery management changes. Most notable are the ongoing recovery of the NZ fur seal population in the region, which grew from about 30,000 to 80,000 animals between 1991 and 2008; a 75% increase in the landings of SBT and ongoing concerns about the status and sustainability of this fishery (Wilson *et al.* 2009); and reductions in landings from the South Australian marine scalefish net fishery and the Commonwealth managed demersal gillnet shark fishery. There have also been seasonal and annual perturbations in the extent and timing of upwelling which underpins primary production (van Ruth *et al.* 2010). How the ecosystem responds to such changes is still uncertain (Ward *et al.* 2006, Middleton *et al.* 2007). The key aim of this study was to provide an ecosystem perspective of the SA sardine fishery, by placing its establishment and growth in the context of these dynamic changes and assessing if the current level of fishing pressure on sardines is likely to adversely impact the EGAB ecosystem.

Ecopath with *Ecosim* was used to resolve the trophodynamics of the EGAB ecosystem, to improve understanding of the trophic interactions of the major functional groups and to asses the impacts of temporal changes in fishing activity and changes in apex predator biomass between 1991 and 2008. The model results suggest that most apex predator populations have increased over this period, including baleen whales, dolphins, fur seals, pelagic sharks, large bentho-pelagic piscivorous fish, small pelagic fish and cephalopods. Fur seal recovery is likely to have been driven by a demographic response to reduced mortality from the 1970s onwards that enabled remnant populations to overcome population reductions from colonial sealing that occurred 140-170 years earlier (Ling 1999). In contrast, the recovery of pelagic and demersal sharks, large bentho-pelagic piscivorous fish and some other (non-tuna) fishes indicated by the model, appears primarily to be driven by reductions in fishing mortality. However, *Ecosim* models suggest that a positive PP forcing over the period improves model fits, and is needed to underpin uniform increases in small pelagic fish and cephalopod biomass that have contributed to the broad scale increase in these higher trophic levels (Figure 5.9). A study by Brown *et al.* (2010) simulated positive increases in PP to examine the ecosystem responses to climate change scenarios in a range of marine ecosystems in Australia. As with the EGAB model, they found that such increases in PP also generally led to a positive increase to most functional groups and to increased biomass of high trophic species (Brown *et al.* 2010).

In contrast, the projected decline in SBT and other tunas and kingfish biomass in the EGAB ecosystem appear to be driven by increases in fishing effort. SBT are a highly migratory species and the purse seine fishery in the EGAB targets juvenile fish (Wilson *et al.* 2009). The extent to which modelled changes in the EGAB reflect changes in the broader population or are compounded by impacts in other jurisdictions and fisheries (high seas long-line fisheries) is unclear, but should be investigated further.

The growth of the sardine fishery in the EGAB region since its establishment in 1991 has been rapid, and its catch now exceeds that of all other fisheries by a factor of three. Despite this, sensitivity analyses based on mixed trophic impacts detected negligible impacts on other predators. However, this steady-state analysis was based on the establishment years of the fishery, when catches were very low, and does not take into account the substantial increases in catch and the changing abundances or diets of groups. In contrast, dynamic assessment using *Ecosim* indicated that many groups were sensitive to changes in sardine biomass. Of the land breeding marine predators, crested terns demonstrated the greatest sensitivity to reduction in sardine biomass both in direction (negative) and magnitude, followed by Australasian gannets. The latter species only breeds at one site in the EGAB region, in its south east, on a disused light-house platform off Cape Jaffa, distant from the centre of the sardine fishery. In contrast, there are many breeding colonies of crested terns situated adjacent to the sardine fishery in southern Spencer Gulf and Investigator Strait (Chapter 4). Demographic studies of the species indicate that birds were smaller and had lower survival rates in years following the two mass mortality events of sardines, in 1995 and 1998 (McLeay *et al.* 2009b). In contrast, Page *et al.* (Chapter 6) found support for the reverse relationship, that the morphology of crested terns was negatively related to sardine spawning biomass in the previous year. Their dependency on small pelagic fish, including sardines which they take from waters near colonies during a short breeding season (McLeay *et al.* 2009a, McLeay *et al.* 2010), gives this species the greatest potential to provide ecological performance indices for the sardine fishery, through foraging, reproductive and population response variables; and further research to assess their suitability is warranted.

Little penguins also demonstrated a slight reduction in biomass in response to reduced sardine biomass, although the magnitude of this response was minor. In Victoria, the reproductive success of little penguins decreased coincident with a reduction in the contribution of sardines in their diet due to a sardine mortality event (Dann *et al.* 2000, Chiaradia *et al.* 2003). Page *et al.* (Chapter 6) identified a number of correlations between reproductive performance indices and sardine spawning biomass, including breeding success (eggs to hatching), chick morphology and foraging trip duration at colonies adjacent to the fishery which may also serve as ecological performance indices for the fishery. The *Ecosim* model also identified that the rate of increase in the biomass of New Zealand fur seals declined in response to reduced sardine biomass. Page *et al.* (Chapter 6) identified that the morphology of pups was negatively related to the spawning biomass in the current and previous years and to the catch in the current and previous years, and that the growth rate of pups was positively related to the catch in the previous and current years, also indicating the species' potential as an ecological performance indicator for the fishery. The *Ecosim* model also identified common dolphins, pelagic sharks, SBT, and other tuna and kingfish as sensitive to biomass reductions in sardines, but because they are entirely marine and some are subject to their own fishery related mortality, they are less suitable as indicator taxa compared to the land breeding marine predators.

Ecosystem indicators identified significant changes in the EGAB ecosystem between 1991 and 2008, the most significant of which was the 4.6-fold increase in total catch for the EGAB, which is entirely attributable to growth in the sardine fishery (Figure 5.10 a-d). Not surprisingly, this precipitated a significant reduction in the mTLc between 1991 and 2008. Usually such reductions demonstrate increased ecosystem impact as a consequence of a reduction in high trophic level predators relative to lower trophic level organisms (Coll *et al.* 2009). However, significant increases in Kempton's Q biodiversity index over the period demonstrating increased growth in high trophic level species, indicate that reduction in the mTLc is attributable to growth in the sardine fishery, and not a reduction in high trophic level biomass. The high FIB index values (all well above zero) and its increasing trend over the study period reflect expansion of fishing effort and/or more catch than expected in the system where there are bottom-up effects (Coll *et al.* 2009). Values above zero indicate fishing in balance, whereas values below zero suggest high fishing impacts that are impairing ecosystem function (Christensen 2000). These indices are useful because they integrate many aspects of fishing impacts and trophodynamic change to provide indices of ecosystem health. For the EGAB ecosystem, all these indices suggest that despite the large scale expansion of the sardine fishery between 1991 and 2008 and reduction in the mTLc, most functional groups from cephalopods to small pelagic fish, and high trophic level predators have been increasing in biomass over this period. This suggests that the current fishery management strategy is sufficiently conservative to ensure that the fishery is being managed according to the principles of ecological sustainable development (ESD) (Fletcher *et al.* 2002). However, the extent to which this positive assessment reflects management of the sardine fishery is uncertain, given that much of the positive changes estimated by the model reflect other changes in the ecosystem, including reductions in fishing effort and mortality in some fleets (SA marine scalefish net and Commonwealth demersal gillnet shark fishery), and a positive trend in PP over the study period. How the current or alternate management strategies of the sardine fishery would perform under alternate oceanographic conditions (such as more variable upwelling), and/or in response to management changes in other fisheries in the EGAB region, may be useful to explore.

One of the key dynamics in the EGAB ecosystem is the variable trends in biomass of some of the high trophic level species. Most notable is the recovery of fur seal populations over the last 25 years to numbers not seen for more than 150 years. This recovery is continuing and it is unclear at what level populations will stabilise. Fishing pressure on populations of SBT in the GAB and Southern Ocean south of Australia over the last 30 years has seen these stocks reduced to a fraction of their pre-exploited levels, and they are subject to continued over fishing (Wilson *et al.* 2009). Other important pelagic predators, such as pelagic sharks have also been targeted by some fisheries and are subject to bycatch in others, and have seen historic declines in the biomass of their stocks. How such depletions in populations of key high trophic level predators over the last 200 years (a ‘predator-gap’) has affected the EGAB ecosystem is unclear, but this study has provided some interesting insights, in particular hypotheses about the role of cephalopods in regulating the flow of small pelagic fish production to high trophic levels.

Ecosim scenarios that explored the ecosystem response to recovering fur seal and SBT populations out to 2040, indicated that such recoveries were supported through reductions in the biomass of small pelagic fish (mainly jack mackerel) and cephalopods (especially arrow squid). Scenarios investigating the response to reduced arrow squid biomass as a consequence of predation and competition pressure exerted by apex predators suggest this may enable greater production of small pelagic fish to be directed into higher trophic levels, in this case to large piscivorous fish and apex predator species. This hypothesis suggests that ‘predator gaps’ that may have resulted from reduced fur seal, SBT and shark biomass, have been quickly filled by cephalopods which can build up biomass quickly in response to increased small pelagic fish availability and reduced predation pressure. A build-up in cephalopods may reduce trophic flows to high trophic levels, with their short generation times and reduced predation pressure resulting in much of their biomass being returned to detritus. A study of the role of small pelagic fish in southern Australian ecosystems identified that these ecosystems were largely bottom-up forced, but that different parts of the these food webs could exert both bottom-up and top-down control (Bulman *et al.* 2010). Furthermore, switching between these states can occur in response to fishing and climate change pressures (Bulman *et al.* 2010). In this respect it is possible that cephalopods may be able to exert some degree of ‘wasp-waist’ control (Cury *et al.* 2000), by restricting energy flow to competitors at similar or higher trophic levels, although typically wasp-waist species are highly mobile low trophic level species such as small plankton feeding pelagic fish (Cury *et al.* 2000, Freon *et al.* 2009, Bulman *et al.* 2010).

Results from this study of the EGAB ecosystem highlight the importance of small pelagic fish to the higher trophic levels, the trophic changes that result from loss and recovery of apex predator populations, and the potential pivotal role of changing cephalopod biomass in regulating trophic flows. The use of trophodynamic models as a tool to provide context to the potential impacts and management strategies of a single fishery relative to the temporal changes of a complex dynamic ecosystem subject to dynamic impacts from multiple fishing fleets and climate change, is highlighted by this study. Although there are clearly some limitations of the model due to data deficiencies, and the number and composition of functional groups, the model provides a basis from which future improvements in model design and data inputs will enable more complex management and ecological questions to be examined. The ecosystem performance indicators produced by the *Ecosim* model also provide a means to assess the potential impacts of the sardine fishery relative to those from other fisheries and environmental change. This ability to resolve and attribute potential impacts from multiple fishing fleets and environmental changes is also critical for the development and utility of ecological performance indicators for assessing ESD targets, and will not be possible without further trophodynamic modelling.

Table 5.1. Functional groups and their diet matrix showing proportional prey contribution used in the balanced EGAB *Ecopath* model.

Group name	Group	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19
Baleen whales	1																			
Bottlenose dolphin	2																			
Common dolphin	3																			
NZ fur seal	4																			
Aust fur seal	5																			
Aust sea lion	6																			
Little penguin	7				0.0015		0.0004													
Petrels	8				0.0010															
Gannets	9																			
Terns	10																			
Pelagic sharks	11																			
Demersal sharks	12						0.0025				0.0009	0.0133								
Rays and skates	13						0.0042					0.0064								
SBT	14											0.0818								
Other tunas-kingfish	15																			
Large bentho-pelagic piscs.	16		0.0406	0.0862	0.0712	0.0640	0.0954	0.0006	0.0153	0.3850	0.0586	0.0002				0.0777				
Blue mackerel	17		0.0004	0.0032	0.0011			0.0161	0.0002	0.1100	0.0043	0.0288			0.0606	0.0447	0.0303			
Jack mackerel	18		0.0153	0.1137	0.0611	0.0536	0.0083		0.0509		0.0016	0.0019			0.0720	0.0356	0.0360			
Redbait	19				0.1186	0.4489		0.0441	0.0045		0.0141				0.1818	0.0433	0.0909			
Anchovy	20			0.4328	0.0055			0.6152	0.0044	0.0900	0.3072	0.0781			0.0750		0.0375			
Sardine	21		0.0209	0.2082	0.0109		0.0010	0.0353	0.0004	0.1600	0.2294	0.2298			0.4288	0.5267	0.2164			
Inshore small planktivores	22			0.0063	0.0003			0.0406	0.0002	0.0075	0.0783									
Salmons & ruffs	23		0.0013	0.0170	0.0145		0.0025	0.0885		0.0200	0.0249	0.0072				0.0370				
Medium demersal piscs	24		0.0087		0.0630	0.3674	0.1097	0.0053		0.0375	0.0567	0.0021								
Small demersal piscs	25		0.2326	0.0208	0.0375	0.0029	0.0075	0.0075			0.1612	0.0004								
Medium demersal invert feeders	26				0.0021		0.0022				0.0007		0.1000							
Small demersal invert feeders	27		0.0140	0.0019	0.0043	0.0027	0.0161	0.0001			0.0007						0.0003			
Mesopelagics	28				0.0301	0.0012			0.0161											
Small demersal omnivore	29		0.0467	0.0095	0.0050	0.0048	0.0151	0.1137		0.1500	0.0527	0.0001								
Arrow squid	30		0.0007	0.0124	0.2697	0.0460	0.0475	0.0273	0.0104	0.0200	0.0021	0.1904			0.1818	0.1341	0.0909			
Calamary	31		0.1337	0.0602	0.0022		0.1472			0.0200	0.0037	0.2166	0.1187							
Other squids	32		0.2004	0.0248	0.0008	0.0034	0.1905		0.0271		0.0002	0.0814				0.0232	0.0502			
Octopus	33		0.2846	0.0029	0.0013	0.0051	0.3029		0.0004			0.0603					0.3771			
Large zooplankton (carnivores)	34	0.0250							0.0098				0.0964	0.0045			0.0147	0.0076	0.0137	0.0163
Small zooplankton (herbivores)	35	0.4750							0.0361				0.0237	0.0009		0.0777	0.0108	0.7604	0.7438	0.5799
Benthic grazer (megabenthos)	36				0.0004		0.0468	0.0055	0.0082		0.0029	0.0012	0.2790	0.2219			0.0234	0.1834	0.1197	0.1960
Detritivore (infauna - macrobenthos)	37												0.0524	0.7727			0.0092		0.1164	0.0073
Filter feeders	38												0.2032				0.0118	0.0292	0.0064	0.1841
Primary production	39												0.1264				0.0002	0.0194		0.0163
Detritus	40																			
Import	41	0.5000			0.2979				0.8161											

Table 5.1 cont.

Group name	Group	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38
Baleen whales	1																			
Bottlenose dolphin	2																			
Common dolphin	3																			
NZ fur seal	4																			
Aust fur seal	5																			
Aust sea lion	6																			
Little penguin	7																			
Petrels	8																			
Gannets	9																			
Terns	10																			
Pelagic sharks	11																			
Demersal sharks	12																			
Rays and skates	13																			
SBT	14																			
Other tunas-kingfish	15																			
Large bentho-pelagic piscs	16											0.0400	0.0990							
Blue mackerel	17											0.0016								
Jack mackerel	18											0.1032	0.3940							
Redbait	19				0.0420							0.0744								
Anchovy	20				0.1920							0.0371	0.1120							
Sardine	21				0.0980							0.1213								
Inshore small planktivores	22				0.3251							0.0075								
Salmons & ruffs	23				0.0245							0.0088	0.1050							
Medium demersal piscs	24				0.0442	0.0035		0.0300					0.0140							
Small demersal piscs	25				0.2205								0.1430		0.0807					
Medium demersal invert feeders	26														0.0050					
Small demersal invert feeders	27				0.0093							0.0031		0.0500	0.0538					
Mesopelagics	28									0.0600				0.0600						
Small demersal omnivore	29				0.0200	0.0150	0.0033	0.0427			0.0001	0.0109	0.0270							
Arrow squid	30				0.0245							0.2202								
Calamary	31					0.0091						0.0112								
Other squids	32									0.0020		0.0095	0.0050	0.0020	0.0538					
Octopus	33					0.0115	0.0003						0.0160							
Large zooplankton (carnivores)	34		0.0296			0.0693	0.1129	0.1310	0.0377	0.0080	0.1684	0.0381		0.0080					0.1000	
Small zooplankton (herbivores)	35	0.4938	0.6748			0.0056	0.0347	0.0040		0.9300	0.0024	0.2626		0.8800		0.2000		0.4000	0.4000	0.8000
Benthic grazer (megabenthos)	36	0.5062	0.2883	0.8652		0.0825	0.1036	0.6292	0.0118		0.6602	0.0086	0.0850		0.6992					
Detritivore (infauna - macrobenthos)	37					0.1864	0.0815	0.1574	0.9163		0.0479	0.0407						0.5500		
Filter feeders	38		0.0049			0.4419	0.0710	0.0018	0.0342		0.0106	0.0012			0.1076					
Primary production	39		0.0023	0.1348		0.1752	0.5926	0.0039			0.1104					0.8000	1.0000			0.2000
Detritus	40																			
Import	41																			

Table 5.2. Summary of fleet landings (catch t km^{-2}) by functional group used in the balanced EGAB *Ecopath* model.

Group name	SA Sardine Fishery	SAMS-Line Fisheries	SAMS-Net Fisheries	SBT Purse Seine	SBT Pole & Bait	SE Trawl	GAB Trawl	Demersal shark	SG Prawn Fishery	GSV Prawn Fishery	WC Prawn Fishery	Total
Pelagic sharks	0	9.13×10^{-5}	8.11×10^{-3}	0	0	0	0	1.54×10^{-5}	0	0	0	8.22×10^{-3}
Demersal sharks	0	1.66×10^{-3}	8.21×10^{-3}	0	0	2.16×10^{-5}	2.26×10^{-5}	6.28×10^{-3}	0	0	0	1.62×10^{-2}
Rays and skates	0	2.29×10^{-4}	6.00×10^{-5}	0	0	0	2.23×10^{-6}	0	0	0	0	2.92×10^{-4}
SBT	0	0	0	2.14×10^{-2}	9.23×10^{-6}	0	0	2.42×10^{-4}	0	0	0	2.16×10^{-2}
Other tunas-kingfish	0	0	0	1.35×10^{-2}	9.17×10^{-5}	1.38×10^{-5}	1.95×10^{-7}	2.51×10^{-4}	0	0	0	1.38×10^{-2}
Large bentho-pelagic piscs	0	1.37×10^{-3}	4.47×10^{-4}	0	0	1.27×10^{-3}	3.36×10^{-4}	1.00×10^{-7}	0	0	0	3.42×10^{-3}
Blue mackerel	0	0	0	0	0	0	1.95×10^{-7}	0	0	0	0	1.95×10^{-7}
Jack mackerel	0	0	0	0	1.03×10^{-6}	0	6.49×10^{-8}	0	0	0	0	1.09×10^{-6}
Sardine	5.72×10^{-5}	0	0	0	0	0	0	0	0	0	0	5.72×10^{-5}
Salmons & ruffs	0	9.05×10^{-5}	4.69×10^{-3}	0	0	0	0	5.00×10^{-7}	0	0	0	4.78×10^{-3}
Medium demersal piscs	0	2.65×10^{-3}	1.73×10^{-3}	0	0	5.95×10^{-4}	1.05×10^{-3}	1.83×10^{-4}	0	0	0	6.21×10^{-3}
Small demersal piscs	0	0	0	0	0	8.47×10^{-6}	0	0	0	0	0	8.47×10^{-6}
Medium demersal invert feeders	0	0	0	0	0	8.47×10^{-6}	0	3.30×10^{-6}	0	0	0	1.18×10^{-5}
Small demersal invert feeders	0	0	0	0	0	0	0	6.00×10^{-7}	0	0	0	6.00×10^{-7}
Arrow squid	0	0	0	0	0	5.18×10^{-5}	1.09×10^{-5}	0	0	0	0	6.27×10^{-5}
Calamary	0	9.66×10^{-4}	7.49×10^{-4}	0	0	0	3.25×10^{-8}	0	1.51×10^{-4}	1.26×10^{-7}	1.48×10^{-10}	1.87×10^{-3}
Other squids	0	0	0	0	0	0	3.25×10^{-8}	0	0	0	0	3.25×10^{-8}
Octopus	0	0	0	0	0	0	0	1.00×10^{-7}	0	0	0	1.00×10^{-7}
Benthic grazer (megabenthos)	0	0	0	0	0	0	1.45×10^{-6}	3.00×10^{-7}	2.15×10^{-3}	1.80×10^{-6}	2.11×10^{-9}	2.16×10^{-3}
Sum	5.72×10^{-5}	7.06×10^{-3}	2.40×10^{-2}	3.48×10^{-2}	1.02×10^{-4}	1.97×10^{-3}	1.42×10^{-3}	6.98×10^{-3}	2.30×10^{-3}	1.93×10^{-6}	2.26×10^{-9}	7.87×10^{-2}

Table 5.3. Summary of fleet discards (catch t km^{-2}) by functional group used in the balanced EGAB *Ecopath* model.

Group name	SA Sardine Fishery	SAMS-Line Fisheries	SAMS-Net Fisheries	SBT Purse Seine	SBT Pole & Bait	SE Trawl	GAB Trawl	Demersal shark	SG Prawn Fishery	GSV Prawn Fishery	WC Prawn Fishery	Total
Aust sea lion	0	0	0	0	0	0	0	2.2×10^{-5}	0	0	0	2.17×10^{-5}
Pelagic sharks	0	0	0	3.00×10^{-5}	0	1.52×10^{-5}	0	0	0	0	0	4.49×10^{-5}
Demersal sharks	0	5.33×10^{-5}	4.63×10^{-6}	0	0	0	8.79×10^{-6}	4.4×10^{-5}	6.46×10^{-4}	5.40×10^{-7}	6.33×10^{-10}	7.57×10^{-4}
Rays and skates	0	1.35×10^{-4}	2.33×10^{-7}	0	0	0	2.09×10^{-5}	0	9.22×10^{-4}	7.72×10^{-7}	9.04×10^{-10}	1.08×10^{-3}
SBT	0	0	0	0	0	0	0	8.4×10^{-6}	0	0	0	8.40×10^{-6}
Other tunas-kingfish	0	0	0	0	0	1.40×10^{-6}	3.70×10^{-9}	1.4×10^{-6}	0	0	0	2.80×10^{-6}
Large bentho-pelagic piscs	0	1.34×10^{-3}	5.28×10^{-5}	0	0	7.84×10^{-5}	2.23×10^{-5}	0	1.07×10^{-3}	8.96×10^{-7}	1.05×10^{-9}	2.56×10^{-3}
Blue mackerel	0	0	0	0	0	0	2.22×10^{-7}	0	0	0	0	2.22×10^{-7}
Jack mackerel	0	8.20×10^{-6}	0	0	0	5.83×10^{-5}	3.73×10^{-6}	0	1.88×10^{-4}	1.57×10^{-7}	1.84×10^{-10}	2.58×10^{-4}
Anchovy	0	0	0	0	0	0	0	0	2.88×10^{-7}	2.41×10^{-10}	2.82×10^{-13}	2.88×10^{-7}
Sardine	0	0	0	0	0	0	0	0	3.79×10^{-7}	3.17×10^{-10}	3.71×10^{-13}	3.79×10^{-7}
Inshore small planktivores	0	0	0	0	0	0	0	0	3.09×10^{-7}	2.58×10^{-10}	3.02×10^{-13}	3.09×10^{-7}
Salmons & ruffs	0	0	4.84×10^{-5}	0	0	0	0	0	6.64×10^{-8}	5.56×10^{-11}	6.51×10^{-14}	4.85×10^{-5}
Medium demersal piscs	0	2.05×10^{-4}	1.11×10^{-3}	0	0	0	6.39×10^{-5}	4.0×10^{-7}	7.08×10^{-4}	5.93×10^{-7}	6.94×10^{-10}	2.08×10^{-3}
Small demersal piscs	0	0	1.14×10^{-5}	0	0	0	0	0	3.10×10^{-3}	2.60×10^{-6}	3.04×10^{-9}	3.12×10^{-3}
Medium demersal invert feeders	0	1.23×10^{-5}	1.01×10^{-6}	0	0	6.04×10^{-6}	0	1.0×10^{-7}	1.74×10^{-3}	1.461×10^{-7}	1.71×10^{-10}	1.94×10^{-5}
Small demersal invert feeders	0	0	0	0	0	0	0	0	1.99×10^{-4}	1.66×10^{-7}	1.95×10^{-10}	1.99×10^{-4}
Small demersal omnivore	0	0	1.61×10^{-3}	0	0	0	5.20×10^{-6}	0	3.64×10^{-3}	3.05×10^{-7}	3.57×10^{-10}	1.98×10^{-3}
Arrow squid	0	0	0	0	0	0	4.10×10^{-7}	0	0	0	0	4.10×10^{-7}
Calamary	0	0	0	0	0	0	0	0	4.68×10^{-4}	3.92×10^{-7}	4.59×10^{-10}	4.69×10^{-4}
Benthic grazer (megabenthos)	0	0	1.24×10^{-4}	0	0	0	0	0	2.08×10^{-3}	1.74×10^{-6}	2.04×10^{-9}	2.20×10^{-3}
Filter feeders	0	0	0	0	0	0	1.13×10^{-5}	0	1.00×10^{-4}	8.37×10^{-8}	9.80×10^{-11}	1.11×10^{-4}
Sum	0	1.75×10^{-3}	2.96×10^{-3}	3.00×10^{-5}	0	1.59×10^{-4}	1.37×10^{-4}	7.6×10^{-5}	1.00×10^{-2}	8.39×10^{-6}	9.82×10^{-9}	1.51×10^{-2}

Table 5.4. Biological parameters by functional group of the balanced EGAB *Ecopath* model. Parameters in bold were estimated by the model. P/B = production/biomass; Q/B = consumption/biomass; EE = ecotrophic efficiency.

	Group Name	Trophic level	Biomass (t km^{-2})	P/B (year^{-1})	Q/B (year^{-1})	EE
1	Baleen whales	**3.01**	0.0389	0.020	5.097	**0.000**
2	Bottlenose dolphin	**4.61**	0.00611	0.080	16.566	**0.000**
3	Common dolphin	**4.66**	0.0039	0.090	20.511	**0.000**
4	NZ fur seal	**4.80**	0.00453	1.184	47.526	**0.944**
5	Aust fur seal	**4.53**	0.00047	1.157	28.819	**0.983**
6	Aust sea lion	**4.91**	0.00422	0.792	29.445	**0.150**
7	Little penguin	**4.71**	0.000698	1.250	85.600	**0.994**
8	Petrels	**4.09**	0.00306	1.000	147.100	**0.070**
9	Gannets	**5.01**	0.0000308	1.000	138.300	**0.000**
10	Terns	**4.52**	0.00000635	1.000	89.900	**0.000**
11	Pelagic sharks	**4.92**	**0.0459**	0.200	1.200	0.900
12	Demersal sharks	**3.92**	0.307	0.180	1.800	**0.326**
13	Rays and skates	**3.68**	0.459	0.350	2.700	**0.014**
14	SBT	**4.52**	**0.145**	0.200	1.600	0.900
15	Other tunas-kingfish	**4.50**	**0.0769**	0.200	1.200	0.900
16	Large bentho-pelagic piscs	**4.68**	0.452	0.338	3.315	**0.959**
17	Blue mackerel	**3.23**	0.219	0.370	3.500	**0.857**
18	Jack mackerel	**3.22**	**0.919**	0.470	3.300	0.900
19	Redbait	**3.38**	**0.561**	0.740	2.800	0.900
20	Anchovy	**3.63**	1.272	0.700	5.040	**0.555**
21	Sardine	**3.36**	1.517	1.600	5.040	**0.330**
22	Inshore small planktivores	**3.93**	**0.489**	1.010	7.300	0.900
23	Salmons & ruffs	**4.51**	**0.246**	0.440	5.400	0.900
24	Medium demersal piscs	**3.47**	0.302	0.485	5.400	**0.844**
25	Small demersal piscs	**2.66**	1.467	0.853	5.367	**0.412**
26	Medium demersal invert feeders	**4.00**	0.0786	0.860	5.400	**0.960**
27	Small demersal invert feeders	**3.53**	**0.149**	1.090	5.500	0.900
28	Mesopelagics	**3.07**	**0.106**	1.005	6.673	0.900
29	Small demersal omnivore	**3.77**	0.170	0.840	16.000	**0.957**
30	Arrow squid	**4.18**	**0.341**	1.950	3.900	0.900
31	Calamary	**4.50**	**0.0837**	1.950	3.900	0.900
32	Other squids	**3.14**	**0.111**	2.500	5.850	0.900
33	Octopus	**4.16**	**0.294**	2.500	5.850	0.900
34	Large zooplankton (carnivores)	**2.20**	1.287	20.000	70.000	0.800
35	Small zooplankton (herbivores)	**2.00**	**35.119**	5.000	32.000	0.800
36	Benthic grazer (megabenthos)	**3.24**	**11.013**	1.600	6.000	0.800
37	Detritivore (infauna - macrobenthos)	**2.52**	**30.983**	1.600	6.000	0.800
38	Filter feeders	**2.80**	**1.581**	1.600	6.000	0.800
39	Primary production	**1.00**	14.900	745.000	0.000	**0.108**
40	Detritus	**1.00**	10.0			**0.009**

Table 5.5. Change of estimated relative biomass of key taxa groups based on *Ecosim* simulations of the EGAB ecosystem between 1991 and 2008, without and with primary production (PP) forcing; and between 2008 and 2040 with no PP forcing, with SBT fishery catch removed and with a 50% reduction in arrow squid biomass.

Summary groups	Relative biomass change of groups				
	1991-2008	1991-2008	2008-2040	2008-2040	2008-2040
	No PP forcing	PP forcing	No PP forcing	No PP forcing No SBT fishery	No PP forcing No SBT fishery 50% arrow squid fishery
Marine mammals, birds, pelagic sharks	54%	87%	62%	85%	80%
Demersal sharks &rays	4%	35%	3%	3%	4%
Large piscivorous fish	-13%	38%	2%	90%	96%
Cephalopods	4%	95%	-7%	-7%	-23%
Medium-small fish	-4%	78%	1%	1%	1%
Small and meso-pelagic fish	25%	25%	-27%	-27%	-3%
Benthic grazers/filter feeders/detritovores	0%	30%	1%	1%	1%
Zooplankton	0%	37%	1%	1%	1%
Primary production	0%	16%	0%	0%	0%

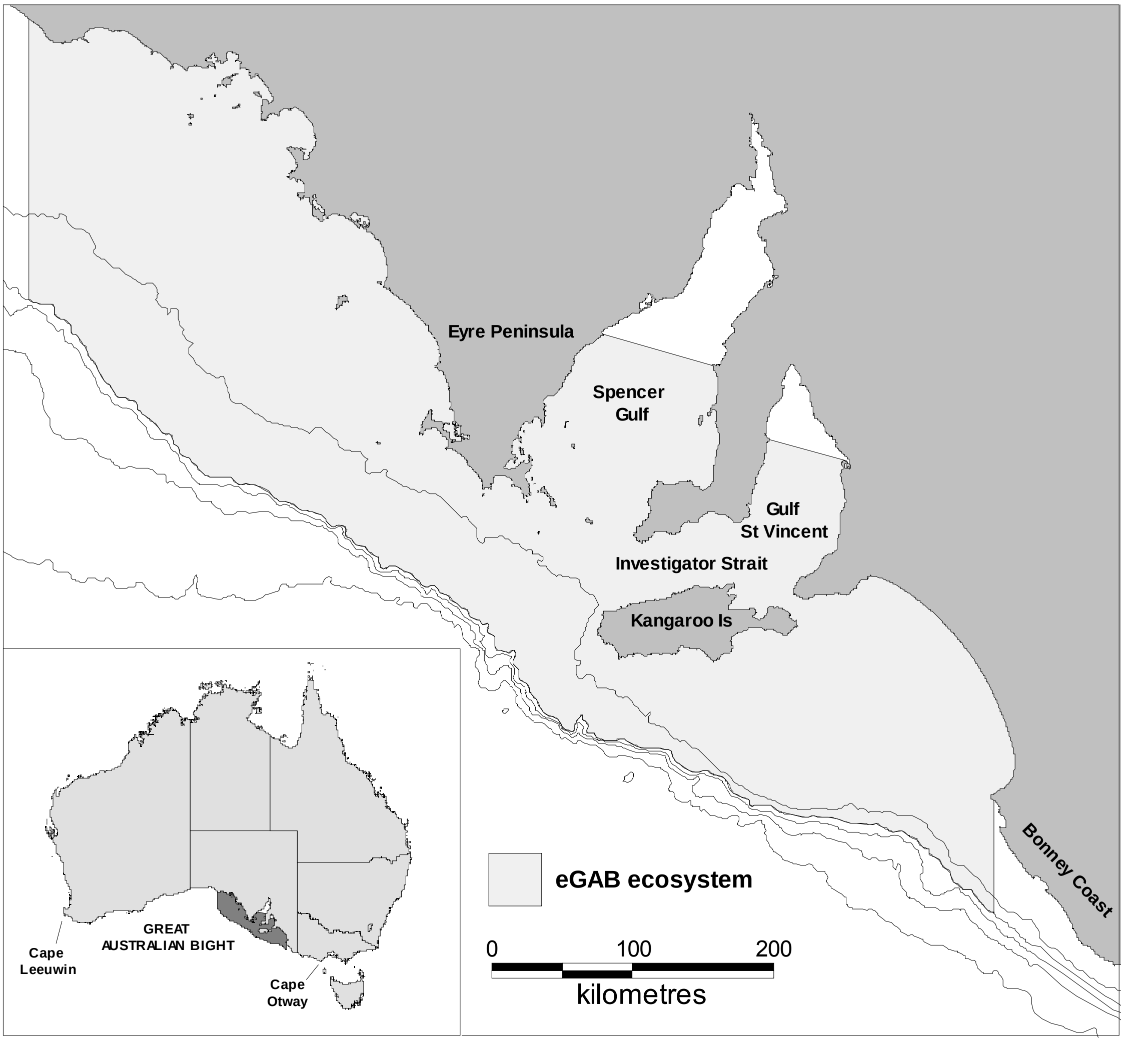

Figure 5.1. Location of the eastern Great Australian Bight (EGAB) ecosystem included in the *Ecopath with Ecosim* model. The depth contours are 100, 200, 500, 1000 and 2000m.

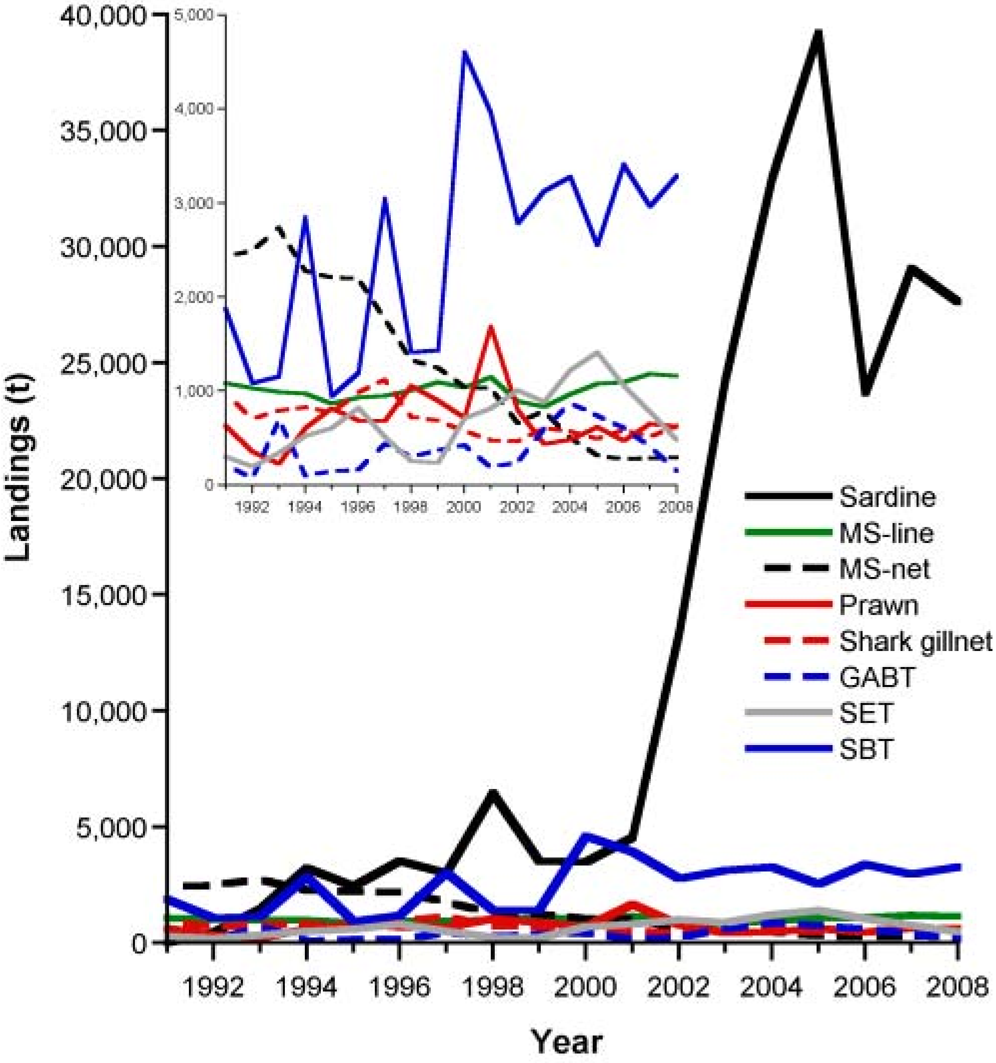

Figure 5.2. Trends in the total landings (catch t y^{-1}) from eight main fisheries in the EGAB ecosystem between 1991 and 2008. The inset graph excludes sardine landings so that trends in the remaining fleets can be more easily resolved.

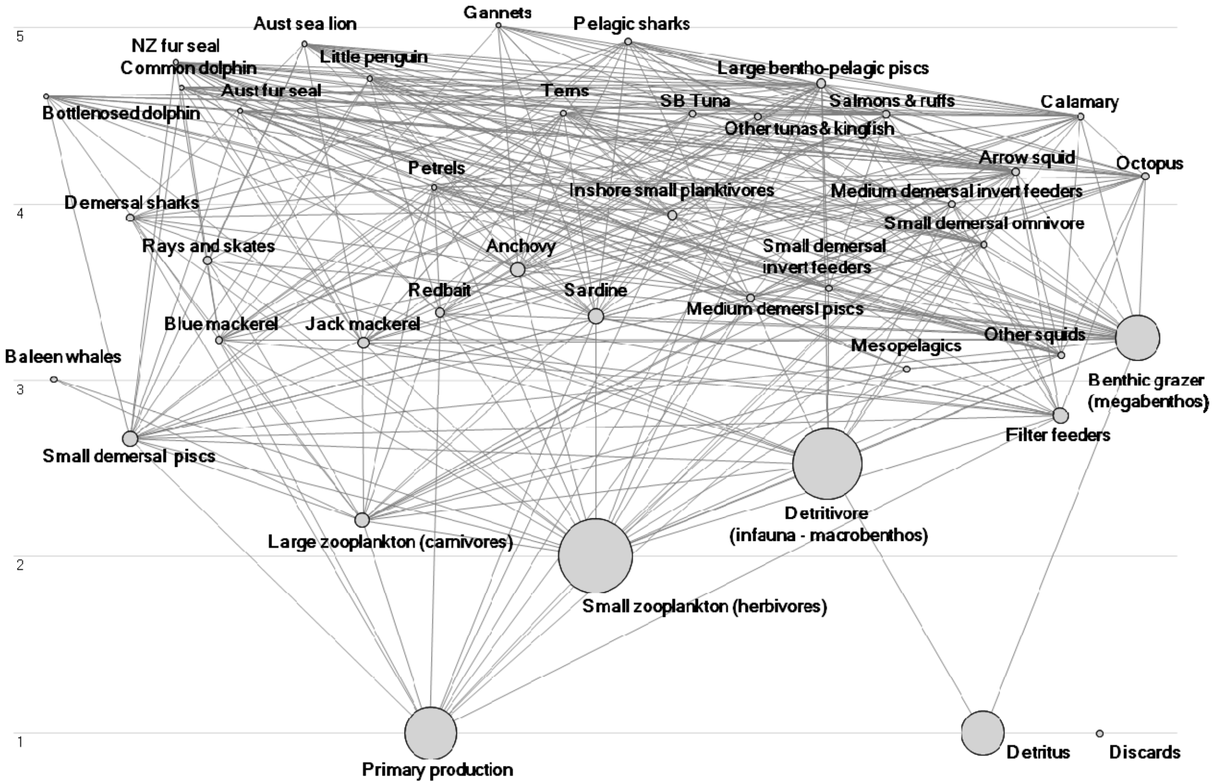

Figure 5.3. Food web diagram of the *Ecopath* model of the EGAB ecosystem. Functional groups are indicated by circles, with size proportional to their biomass. The trophic links between groups are indicated by lines.

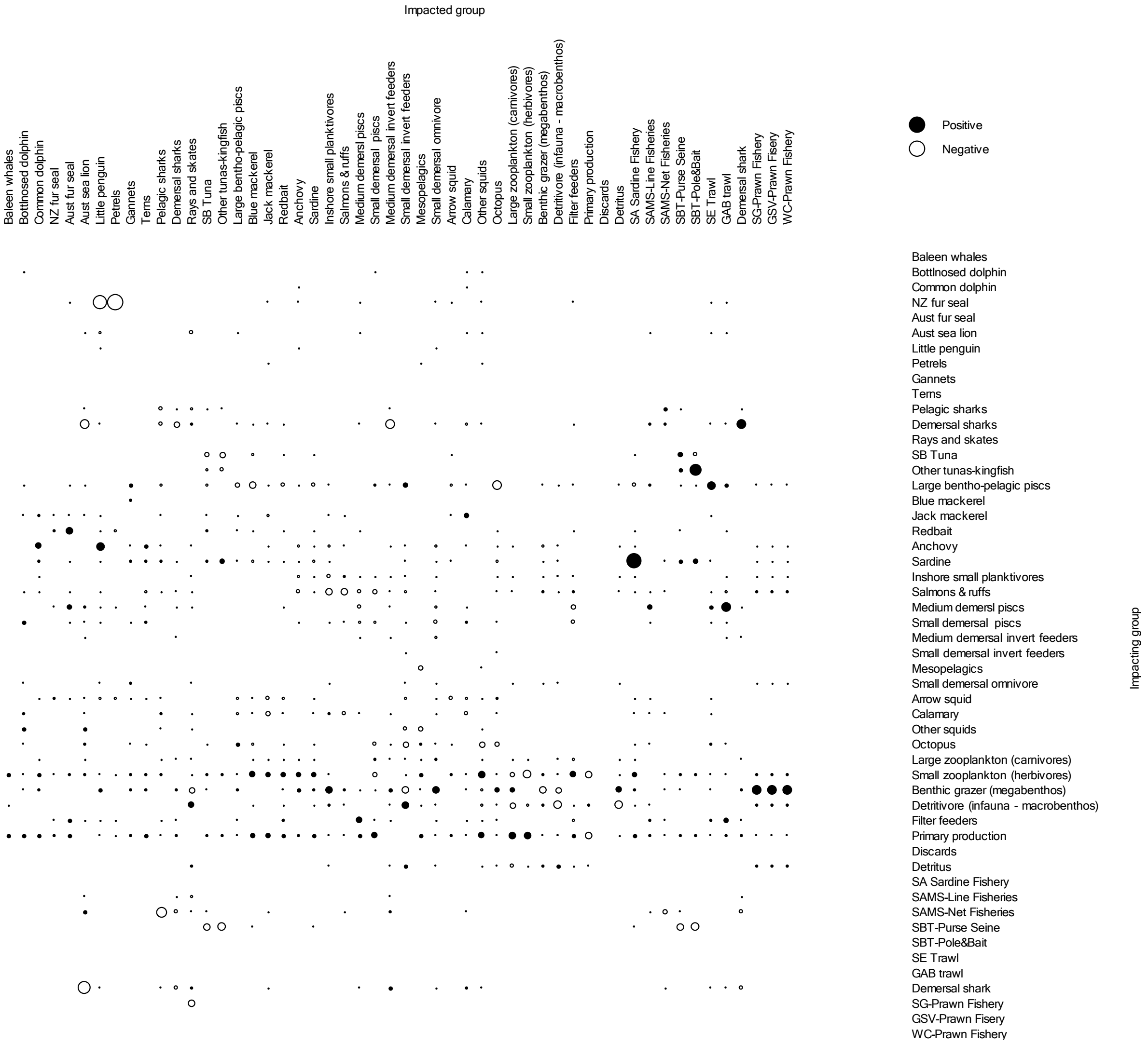

Figure 5.4. Mixed trophic impacts based on the *Ecopath* model of the EGAB ecosystem. It shows the direct and indirect impact that small increase in the biomass of groups listed in rows (Impacting group) have on those listed in columns (Impacted group). Filled circles indicated positive impacts, open circles negative. The size of the circle represents the relative scale of the impact.

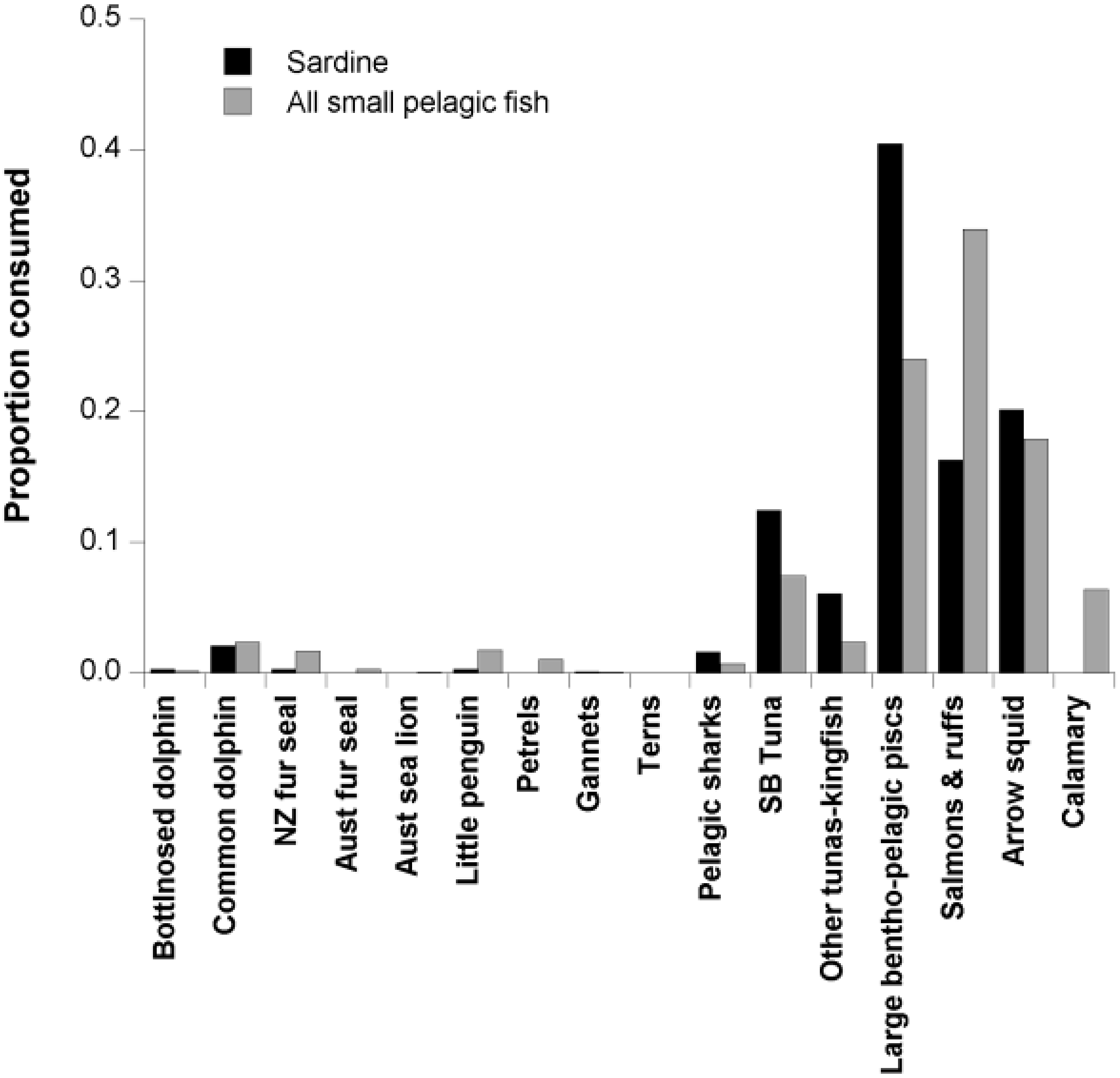

Figure 5.5. Estimated proportional breakdown of total annual conumption of sardines and of all small pelagic fish (sardine, anchovey, blue mackerel, jack mackerel, redbait and inshore planktivores) by their predators based on the *Ecopath* model of the EGAB ecosystem. Total consumption of sardines and of all small pelagic fishes is estimed to be 123,369 t y^{-1} and 396,129 t y^{-1}, respectively.

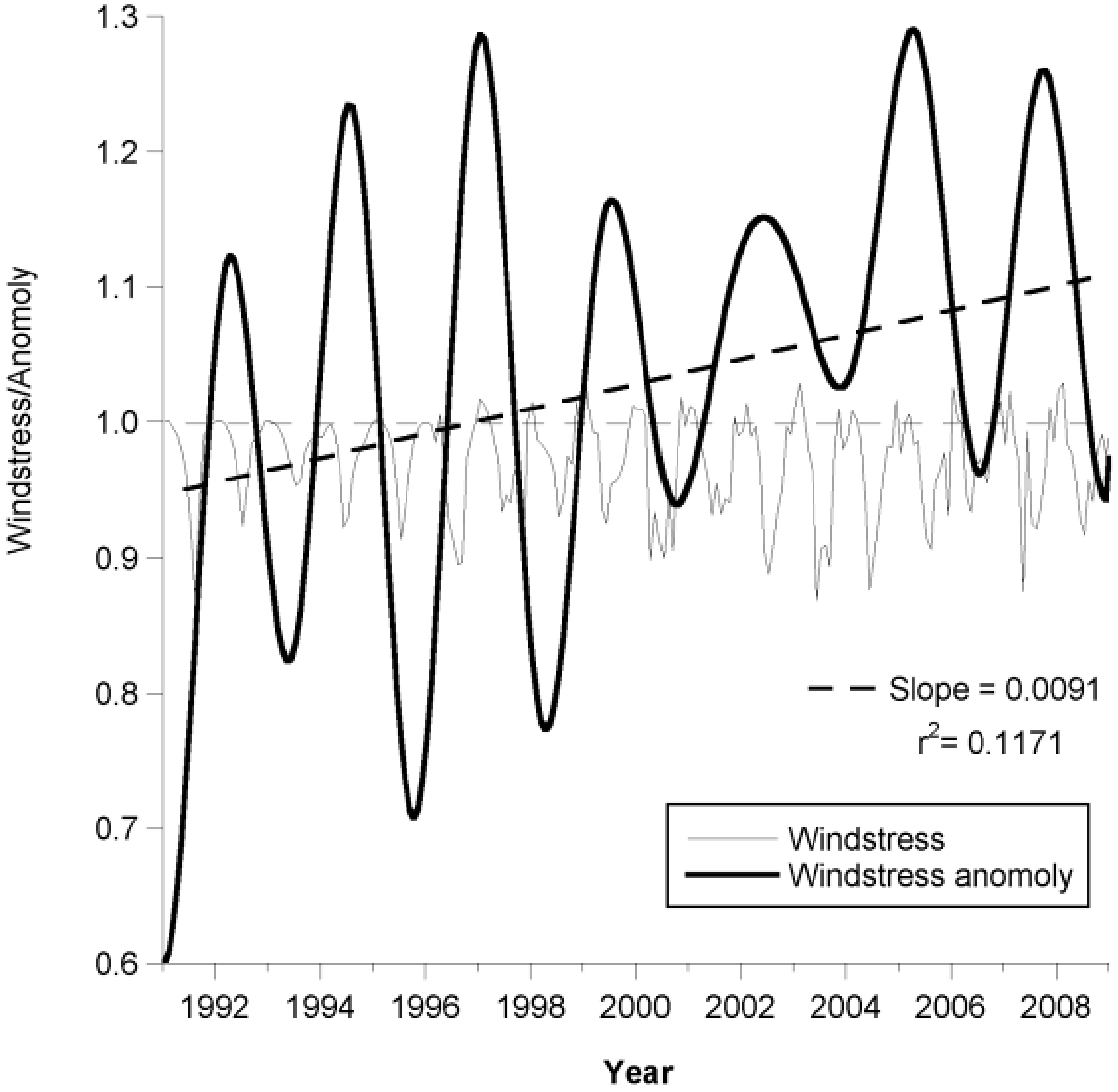

Figure 5.6. Mean monthly windstress values between 1991 and 2008 (scaled to the value for January 1991, grey dotted line), plotted against the *Ecosim* modelled windstress anomoly, with overal trend (dashed black line) and values of the slope and coefficient of variation (r^2).

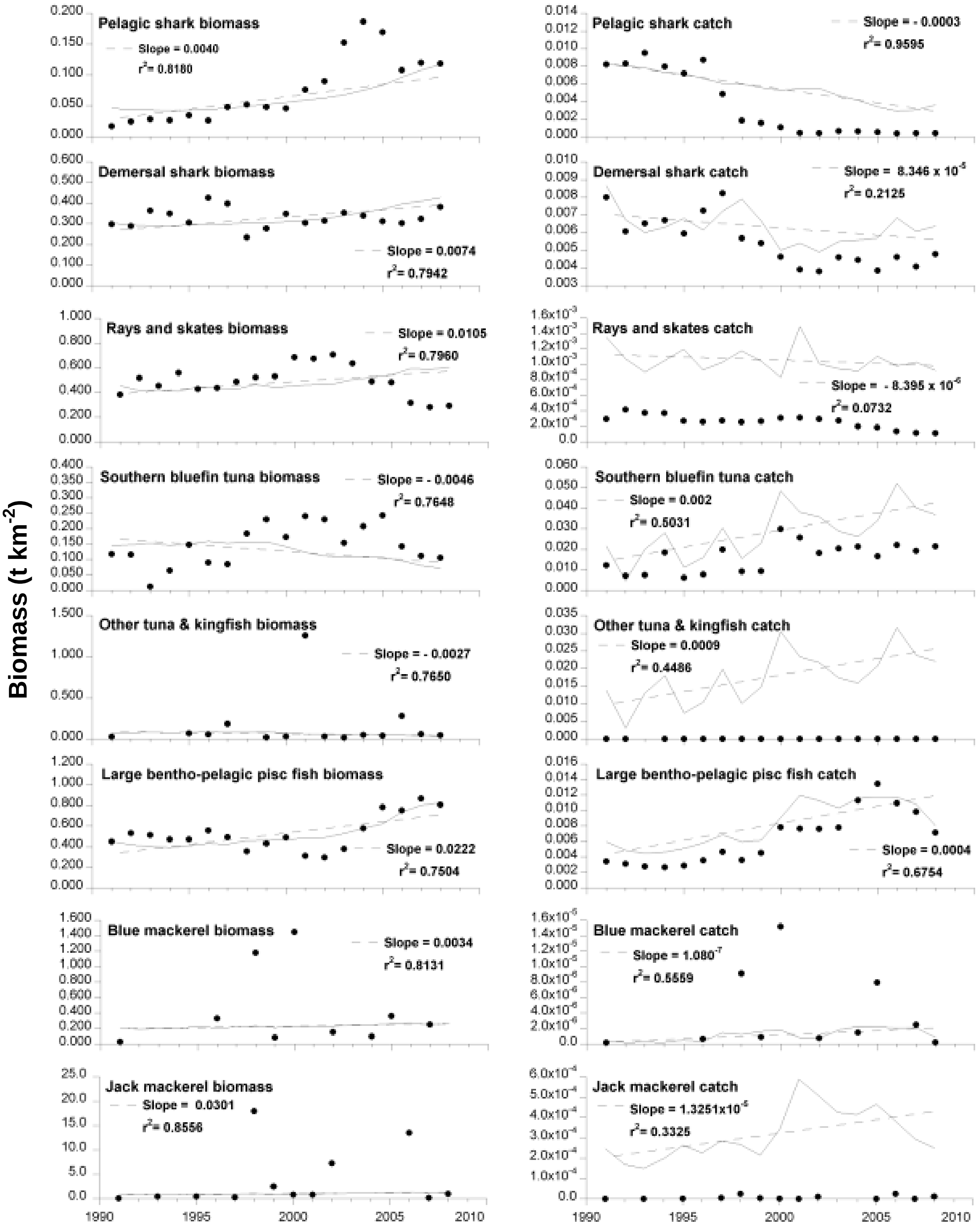

Figure 5.7. Time series fits of the EGAB *Ecosim* model (thin line) to observed biomass (CPUE) and catch data (dots) for 16 functional groups between 1991 and 2008. The modelled trend lines (dashed) are provided, together with values of the slope and coefficient of variation (r^2) (see overleaf for remaining plots).

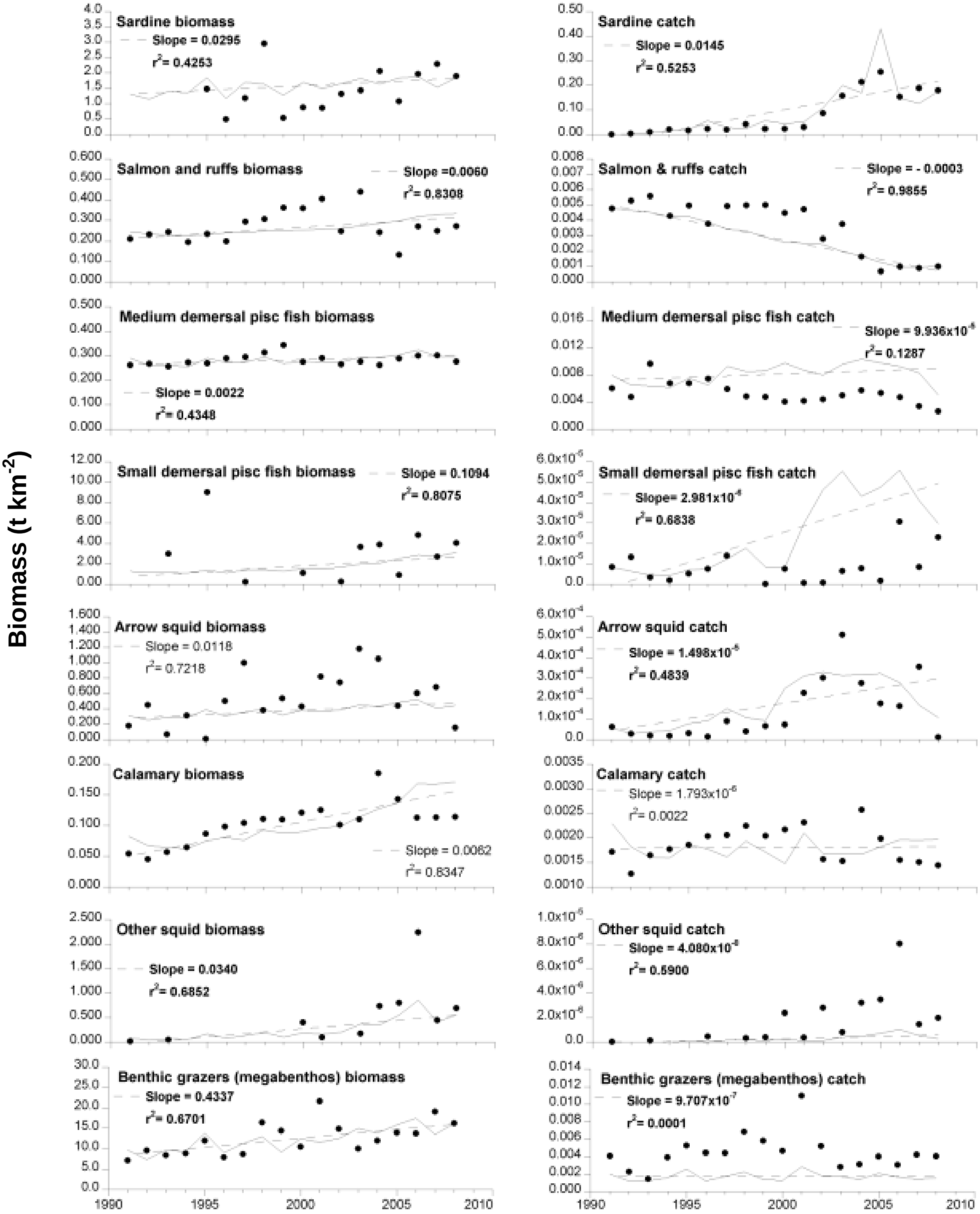

Figure 5.7 cont.

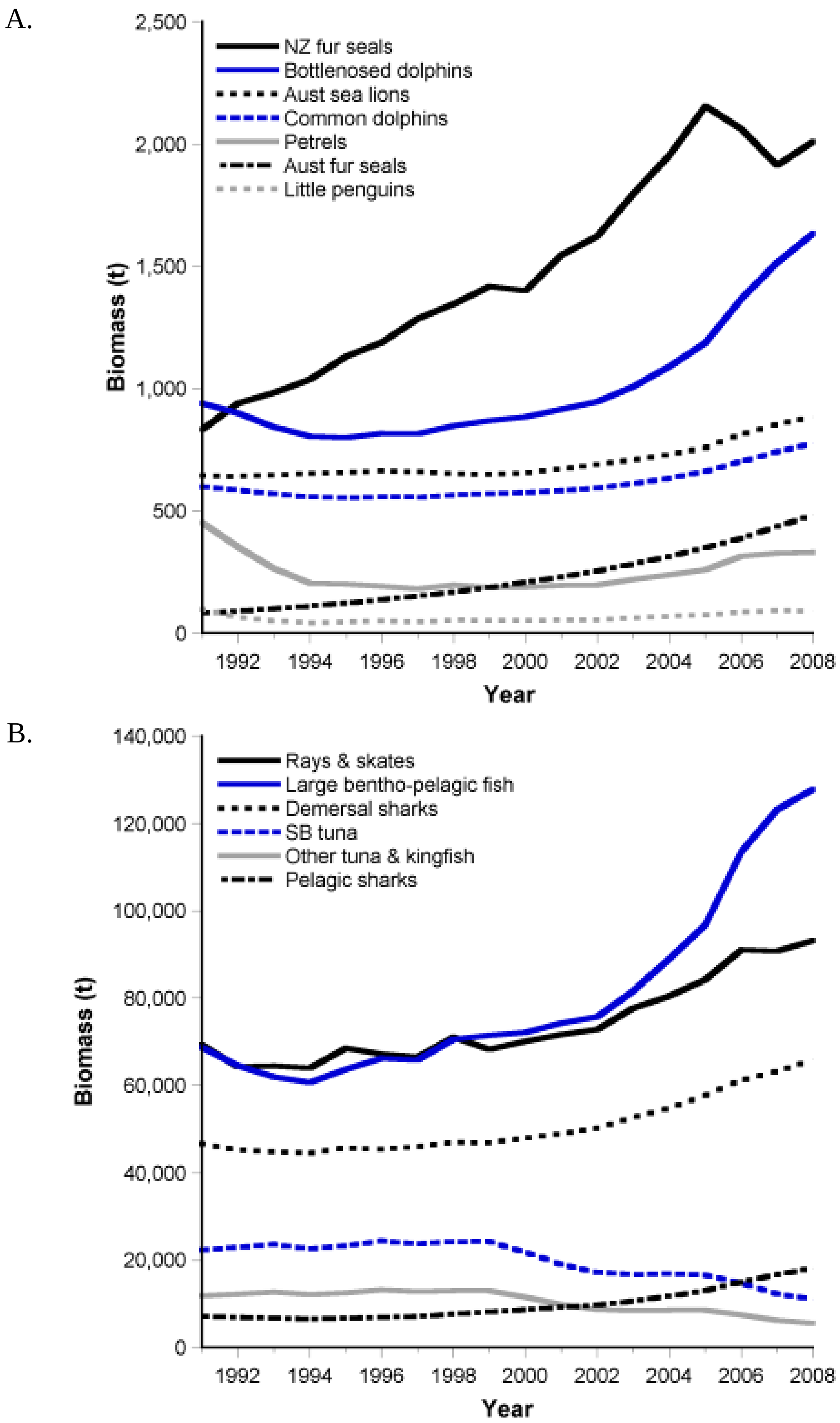

Figure 5.8. Estimated changes in the biomass of high trophic level predators based on the *Ecosim* EGAB model between 1991 and 2008: A) marine mammals and birds; b) sharks, rays and skates, tunas and kingfish. Biomass change in NZ and Australian fur seals is forced in the model as these estimates are based on empircal data for these species in the EGAB ecosystem.

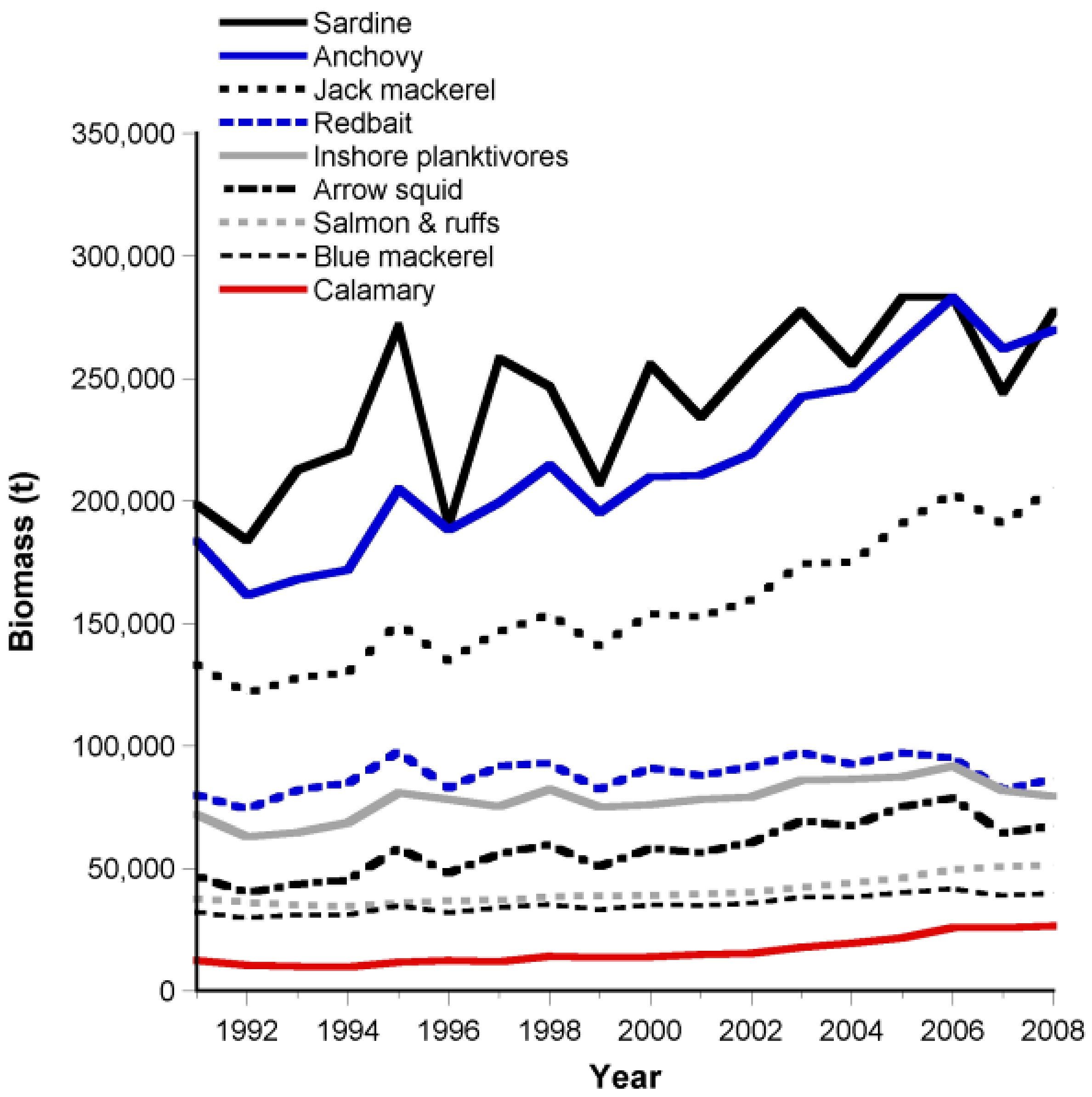

Figure 5.9. Estimated changes in the biomass of small pelagic fish functional groups, arrow squid and calarmary based on the *Ecosim* EGAB model between 1991 and 2008.

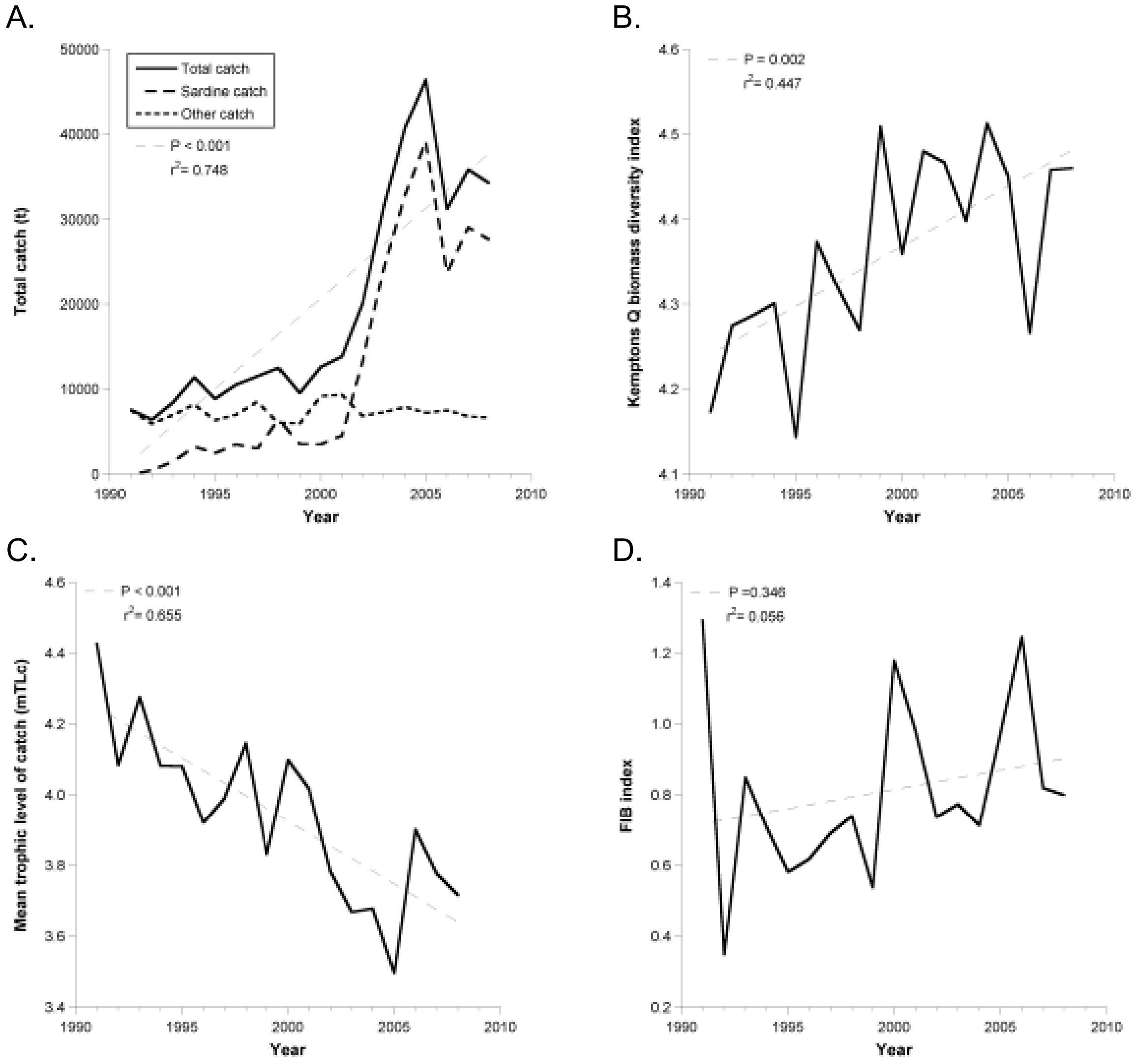

Figure 5.10. Ecosystem indicators calculated from the *Ecosim* EGAB model for the period 1991 to 2008. A. Changes in the landings (catch t y^{-1}) of all fleets (total catch), sardine catch and other catch; B. Kempton's Q biomass diversity index; C. Mean trophic level of the catch and D. Fishing In Balance (FIB) index. Estimated trends are given by dashed lines (regression), with their coefficient of variation (r^2) and level of significance (P).

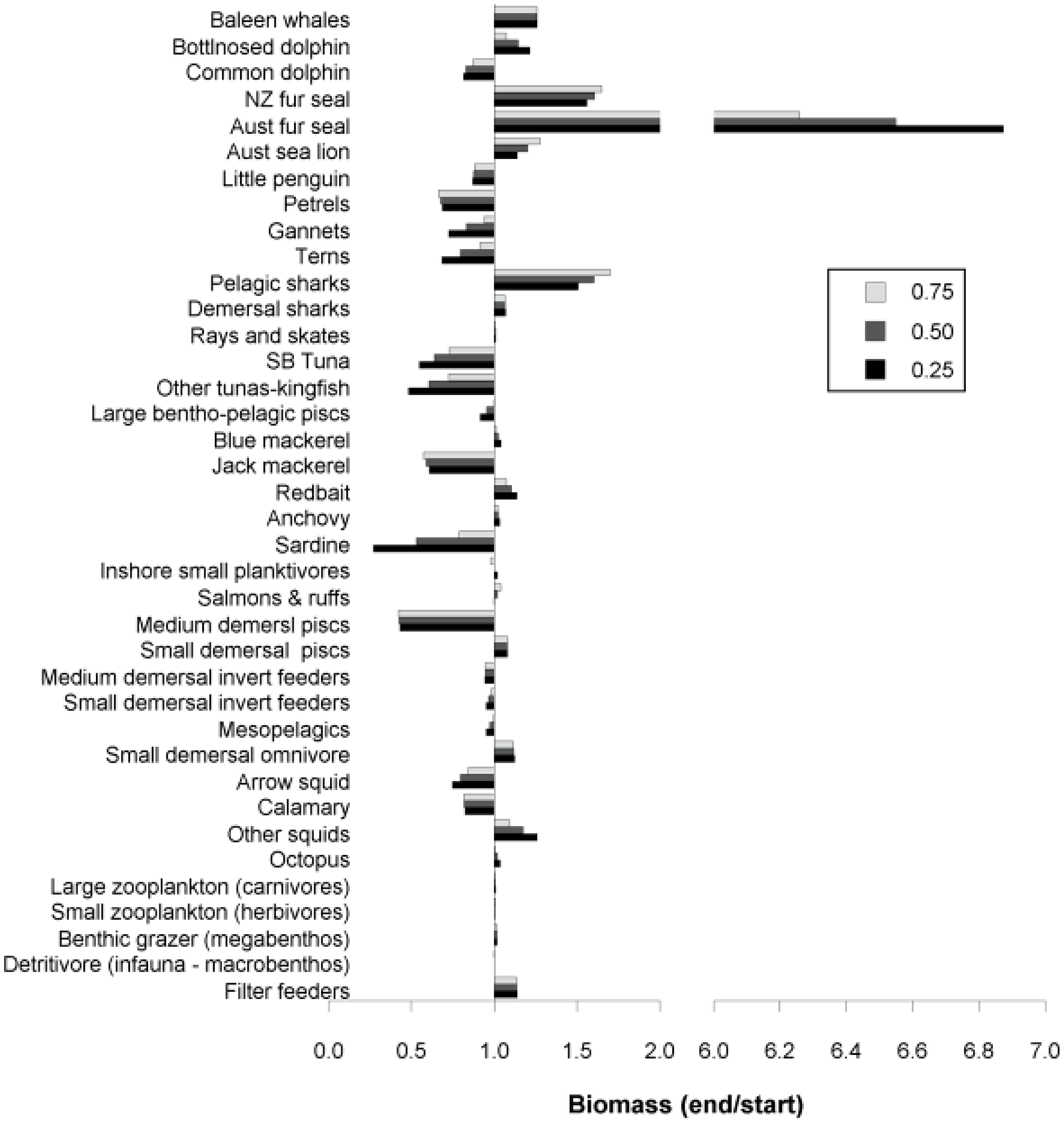

Figure 5.11. Predicted change in the biomass of functional groups in the *Ecosim* EGAB model in 2040 relative to 2008 with sardine biomass reduced to 0.75, 0.50 and 0.25 of 2008 biomass. In this simulation, NZ fur seal and Australian fur seal groups were forced to increase between 2008 and 2040 (in line with observed increases between 1991 and 2008), so that population abundances in the EGAB were about 1.5 and 7.0 times present (2008) levels, respectively (with no change in sardine biomass). Note: the break in the scale of the x-axis.

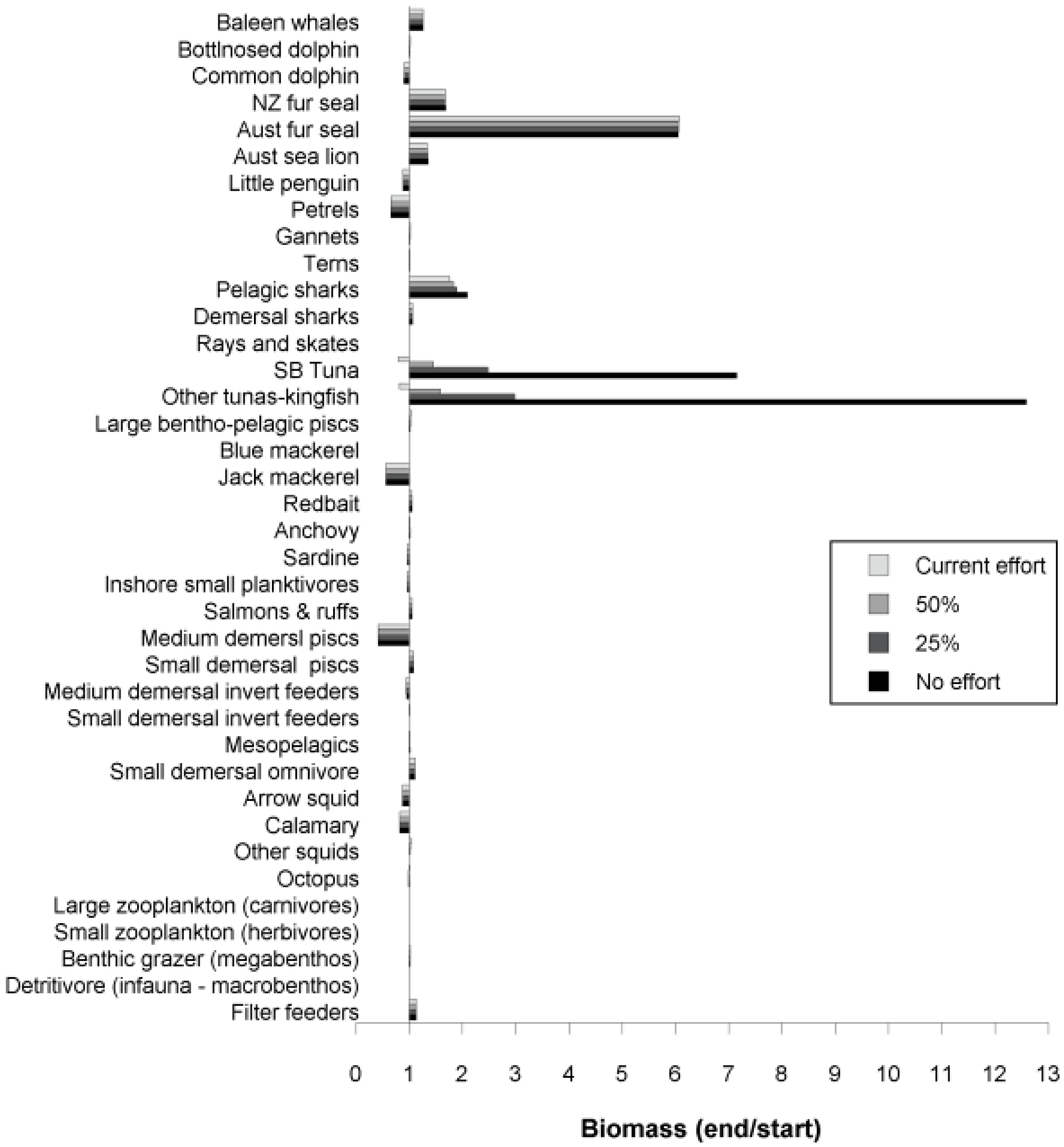

Figure 5.12. Predicted change in the biomass of functional groups in the *Ecosim* EGAB model in 2040 relative to 2008 with biomass increases of SBT. Biomass increases in SBT were simulated by reducing effort in the purse seine fishery by adjusting time series values between 2008 and 2040 under different scenarios of fishing effort (current [2008] levels; 50% 2008 levels; 25% 2008 levels and no fishing effort). In these simulations, NZ fur seal and Australian fur seal groups were forced to increase between 2008 and 2040 (in line with observed increases between 1991 and 2008), so that populations abundances in the EGAB were about 1.5 and 7.0 times present (2008), respectively.

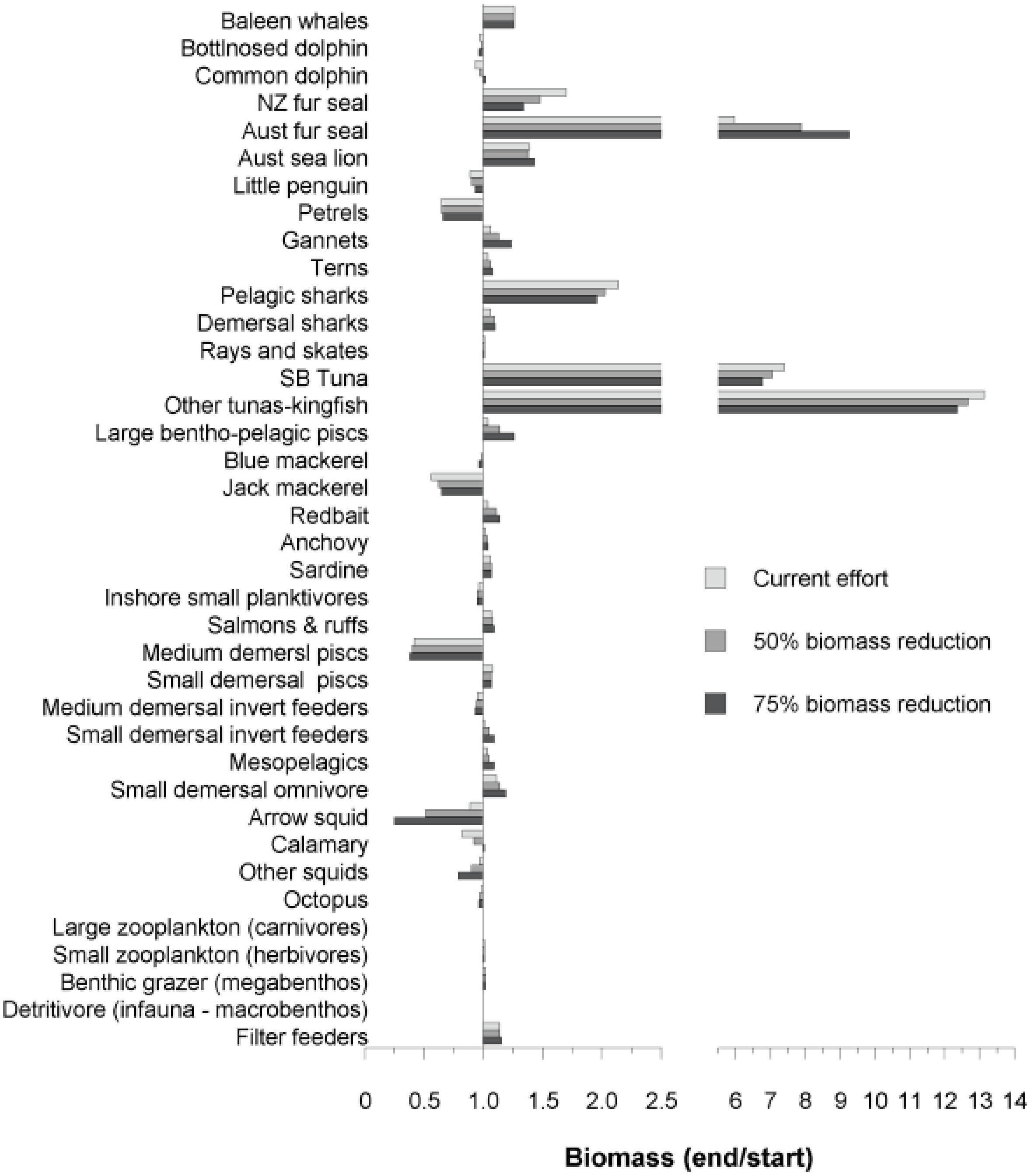

Figure 5.13. Predicted change in the biomass of functional groups in the *Ecosim* EGAB model in 2040 relative to 2008 with different scenarios of arrow squid biomass reduction and with no SBT fishery. Biomass redcution in arrow squid were simulated by adjusting time series values between 2008 and 2040 under different scenarios of fishing effort (current [2008] levels; 50% biomass reduction of 2008 levels; 75% biomass reduction of 2008 levels). In these simulations, NZ fur seal and Australian fur seal groups were forced to increase between 2008 and 2040 (in line with observed increases between 1991 and 2008), so that populations abundances in the EGAB were about 1.5 and 7.0 times present (2008), respectively.

6 IDENTIFICATION OF ECOLOGICAL PERFORMANCE INDICATORS FOR NATURAL PREDATORS OF SARDINE *SARDINOPS SAGAX* IN SOUTHERN AUSTRALIA: ASSESSING THE NEED FOR ECOSYSTEM-BASED MANAGEMENT OF THE SOUTH AUSTRALIAN SARDINE FISHERY

Page B, Goldsworthy SD, Wiebkin A, Shaughnessy PD, McLeay L, Einoder L, McKenzie J, Ward T

Introduction

Global fisheries production has averaged about 80-100 million tonnes per year for the past 20 years, which provides about 16% of the animal protein consumed by humans (FAO 1997, FAO 2005). There is concern about the sustainability of this production because many fisheries are in decline and many are fully-or over-exploited (FAO 2005). Several international strategies aim to improve fisheries management practices and rebuild fish stocks, while helping to preserve biodiversity and sustainable stocks (Mora *et al.* 2009). The most ambitious of these efforts is the attempt to manage fisheries according to the principles of ecologically sustainable development, which requires an understanding of impacts on all species that are affected by fisheries (Fletcher *et al.* 2002).

Data relating to the population dynamics of exploited species and the ecosystem processes that influence them are required for models used for fisheries management (Fletcher 2005, Smith *et al.* 2007). Because data on ecosystem processes are perceived to be both expensive to collect and difficult to incorporate into fishery models, management typically continues with a single-species focus, aimed at ensuring that fish stocks provide the maximum sustainable yield in the long term (e.g. Smith *et al.* 2007). This approach is used effectively in many fisheries, but there is increased recognition at the policy level that improved knowledge of ecosystem processes will reduce the risk of collapses of fisheries and populations of predators that use the same stocks (Hutchings 2000, Sainsbury *et al.* 2000). As a result, there is now considerable interest in the identification, development and application of ecological performance indicators (EPI) of the state of exploited ecosystems (Fulton *et al.* 2005). This has prompted many reviews of the range and suitability of EPIs that can be used in programs aimed at monitoring human impacts (Kabuta and Laane 2003, Hiddink and Kaiser 2005, Cury *et al.* 2005).

For EPIs to be of practical use, they must adequately answer the questions raised by their end users (policy makers, fisheries managers and other stakeholders), by reflecting changes in the ecosystem that is being exploited (Sainsbury *et al.* 2000). The concepts that underlie each EPI must be easy to understand and indicators must be cost effective and able to be reliably measured (Hiddink and Kaiser 2005). Ecological performance indicators may also be related to environmental factors, so it is important to understand these relationships before assuming an EPI reflects human impacts. Most importantly, a significant relationship between an EPI and an indicator of human impact does not confirm that there is a relationship between these indicators, rather it highlights them as potential indicators that require validation (Hiddink and Kaiser 2005).

Because different species of predators vary in their sensitivity to changes in prey availability over both temporal and spatial scales, the development of EPIs to quantify the impacts of humans (e.g. fisheries) on natural predators is not straightforward. Ecological performance indicators are relatively easy to develop and apply if they are based on predator species with specialised diets, restricted foraging ranges when breeding, variable reproductive success or high adult survival rates (Furness and Tasker 2000). Ecosystem impacts are easiest to detect and validate in these predators because they can not compensate for reductions in prey availability by using different prey (Furness and Tasker 2000). Relationships between prey availability and EPIs derived from predators are rarely linear, which increases the data requirements for understanding the nature of a relationship. Differences in the areas used by predators and a fishery, or where stock assessment for a fishery is carried out, can also confound apparent impacts of fishing (Hindell *et al.* 2003). The identification of EPIs therefore requires that a large number of them are monitored simultaneously for several species over several years (de la Mare and Constable 2000, Boyd and Murray 2001). If ecological data from predators show potential as EPIs of fishing impacts, and the relationship can be validated, they can provide robust, independent and cost effective measures that can be used to ensure that ecosystem impacts are detected and minimised (Furness and Tasker 2000, Sainsbury *et al.* 2000, Kabuta and Laane 2003, Mori and Butterworth 2006).

Two types of EPI are typically used to assess the impacts of fisheries on predators (modified from Hindell *et al.* 2003): 1) Breeding indicators describe the growth and size of offspring, the survival and reproductive success of individuals, trends in population sizes, abundances, distribution and emigration/immigration rates, and 2) Foraging indicators describe the diet, dive effort, at-sea movement patterns and rates of travel. Suites of these EPIs can be used to examine the effects of changes in the sizes of exploited stocks, prey availability, fishery catches or other human impacts (Furness and Tasker 2000, Sainsbury *et al.* 2000, Kabuta and Laane 2003, Mori and Butterworth 2006).

Despite the global interest in the identification and development of EPIs to fulfil the requirements of international and regional conventions, few fisheries use EPIs to inform management decisions (FAO 2005, Garcia 2009, Smith and Fulton 2009, Bensley *et al.* 2010). There are numerous examples where EPIs based on predator biology have been related to the sizes of commercial fish stocks (see review in Einoder 2009, McLeay *et al.* 2009b) and fishery catches (Furness 2002, Kabuta and Laane 2003, Hiddink and Kaiser 2005, Fulton *et al.* 2005), but such EPIs are rarely used by fisheries managers to enhance their understanding of the state of the ecosystem. The Convention on the Conservation of Antarctic Marine Living Resources is a notable exception, because it developed and applied EPIs to assess the impact of the krill (*Euphausia superba*) fishery on its natural predators in the Southern Ocean (e.g. Mori and Butterworth 2006). The *E. superba* fishery is relatively small by global standards and is considered to be under-exploited (Hewitt *et al.* 2004). Ecological performance indicators that describe the breeding and feeding ecology of several *E. superba* predators are monitored in several regions and these EPIs are related to indicators of environmental forcing, *E. superba* production and fishery catches (Constable 2005, Mori and Butterworth 2006, Heywood *et al.* 2006). Similarly, EPIs are used to assess human impacts on the Dutch part of the North Sea to help resource managers evaluate the impacts of their policies on the health of the ecosystem (Kabuta and Laane 2003). The impact of the North Sea sandeel (*Ammodytes marinus*) fishery on the breeding performance of black-legged kittiwakes (*Rissa tridactyla*) led to long term fishery closures and ongoing monitoring of EPIs based on *R. tridactyla* production, environmental forcing and fishery impacts (Kabuta and Laane 2003, Frederiksen *et al.* 2004). The maintenance of these long-term monitoring datasets provides an opportunity to assess human impacts and environmental forcing on the health of these exploited ecosystems.

The South Australian sardine fishery has been operating since the early 1990s, with most of its catch coming from the southern Spencer Gulf. The fishery uses purse seine nets and has 14 licence holders, which currently share the annual allowable catch of 30,000 t (Ward *et al.* 2008). Australian Government legislation encourages the implementation of ecosystem-based fisheries management, which requires an understanding of how fisheries compete with and affect other components of the ecosystem, rather than only quantifying impacts on target and bycatch species (Fletcher *et al.* 2002). In 2004, the SA sardine fishery licence holders, fishery managers and Australian scientists initiated a broad ecological study, which aimed to assess the impact of the fishery on the natural predators of *S. sagax*, to determine whether an ecological allocation of *S. sagax* was required.

In this chapter, we aim to identify EPIs for the key marine predators, which are abundant in the ecosystem used by the SA sardine fishery. Such EPIs could include measures of foraging and/or reproductive success, which are likely to be affected by changes in the distribution and abundance of *S. sagax*. We also aim to use the results of this study to provide input for revisions of the management plan of the fishery and establish cost effective systems for ongoing monitoring and assessment of the ecological effects of the SA sardine fishery.

Materials and Methods

This study developed and assessed EPIs that described the breeding and feeding ecology of crested terns (*Sterna bergii*), New Zealand fur seals (*Arctocephalus forsteri*), little penguins (*Eudyptula minor*), short-tailed shearwaters (*Puffinus tenuirostris*) and southern blue-fin tuna (*Thunnus maccoyii*) (Table 6.1). These key marine predators were chosen for this study because they are either significant consumers of *S. sagax* (*S. bergii*, *E. minor*, *T. maccoyii*, refer Chapter 4) or because trophic models indicated that predators (*A. forsteri*, *P. tenuirostris,* refer Chapter 5) were indirectly affected by changes in the availability of *S. sagax*.

Because we were primarily interested in assessing whether the SA sardine fishery was having a detectable impact on the breeding or feeding ecology of these predators, we looked for relationships between the EPIs from the marine predators and indicators that described the activity of the SA sardine fishery and the biomass of the spawning stock. Data that described trends in the biomass of *S. sagax* and the annual catch of the fishery were available for the period between 1995-2007 and 1991-2009, respectively (Table 6.1, Ward *et al.* 2008). We initially used data that were recorded in the same year to look for relationships between each EPI from marine predators and the *S. sagax* catch and spawning biomass. To assess whether there was a time lag for these relationships, we related the *S. sagax* catch and biomass from the year before the breeding and feeding EPIs were recorded. We also assessed the relationship between the spawning biomass (but not catch) from the year after the breeding and feeding EPIs were recorded.

To test the significance of a correlation, data from at least 3 years were required for each EPI. If the data were available, we used several EPIs to describe similar aspects of the breeding and feeding ecology of the marine predators. For example, eight EPIs were used for analyses of the number of *E. minor* coming ashore at Granite Is, because monthly averages were available between April-November (Granite Island Penguin Monitoring Group, Gilbert and Brandle unpublished data). To assess these relationships we used

Spearman's correlation coefficients, because the data were not normally distributed, and we summarised all of the relationships for which $P < 0.05$. Significant relationships were interpreted as indicating that the marine predators were potentially affected by changes in the availability of *S. sagax*. To determine whether a predator group was a particularly good indicator, we ran binomial probability tests to determine if the number of significant relationships detected for each group was higher than that expected by chance, given the large number of correlations tested and the expectation that 5% of these would be significant by chance (i.e. $\alpha=0.05$ for each correlation).

The data that described the breeding and feeding ecology of *S. bergii* were taken from McLeay (2009). McLeay (2009) assessed the survival and morphology of adult *S. bergii* on Troubridge Is (Gulf St Vincent), where *S. bergii* have been banded as chicks over the last 25 years. We used EPIs that described the survival rates and morphology (size and weight) of adult *S. bergii*, because these EPIs are thought to be reliable proxies for food availability when these individuals were being provisioned as chicks (McLeay 2009) (Table 6.1). The diets of *S. bergii* were assessed at Troubridge Is and at several islands in Spencer Gulf (McLeay 2009). Diet samples were obtained when adult or chick *S. bergii* regurgitated during capture (McLeay 2009). Data that described the proportions of anchovy *Engraulis australis* and *S. sagax* in the diet, at each site, were used as separate EPIs in this study (Table 6.1).

The data for *A. forsteri* were from Shaughnessy (2010), Shaughnessy (unpublished data), Shaughnessy and McKeown (2002), Page and Goldsworthy (unpublished data) and Haase (2004) (Table 6.1). Data that described the morphology (birth length and weight) of *A. forsteri* pups (Shaughnessy 2010, Page and Goldsworthy unpublished data) and the daily growth rates of *A. forsteri* pups were available from several sites, between: 1) birth to April/May, 2) April/May to weaning in September/October and 3) birth to weaning in September/October (Table 6.1, Shaughnessy 2010, Page and Goldsworthy unpublished data and Haase 2004). To calculate these EPIs, samples of at least 30 pups were selected at random at each site during each of the three periods, following the methods of Goldsworthy (2006) and Haase (2004). The annual rate of change in pup production of *A. forsteri* was available for several colonies on Kangaroo Is, the Neptune Islands and Liguanea Is (Table 6.1, Shaughnessy 2010, Shaughnessy and McKeown 2002, Page and Goldsworthy unpublished data). Pup production of *A. forsteri* was assessed using mark-recapture techniques by Shaughnessy (2010) and Page and Goldsworthy (unpublished data).

The diet of *T. maccoyii* was assessed in the shelf waters off South Australia. Data that described the proportions of *E. australis* and *S. sagax* in the diet of *T. maccoyii* were the only EPIs that were analysed for this species (Table 6.1). Stomach samples were obtained from *T. maccoyii* caught on trolled lures and the remains of prey were recovered and identified. Data that described the proportions of anchovy *Engraulis australis* and *S. sagax* in the diet were used as separate EPIs in this study (Table 6.1).

Data that described the breeding and feeding ecology of *E. minor* were taken from Bool *et al.* (2007), Wiebkin (unpublished data) and the Granite Island Penguin Monitoring Group (Gilbert and Brandle unpublished data). We used EPIs that described the breeding success of *E. minor* at several sites, using data that spanned 16 years at Granite Is and 5 years at Troubridge Is (Table 6.1). Ecological performance indicators of breeding success were calculated to describe the proportions of *E. minor* 1) eggs that hatched, 2) eggs that resulted in fledglings, 3) chicks that resulted in fledglings and 4) the number of chicks fledged per breeding pair (Table 6.1). Ecological performance indicators that described the morphology (size and weight) of fledglings were calculated to compare the condition of chicks at the end of the provisioning period (Table 6.1). The annual rate of change in the population of *E. minor* was available at Granite Is between 2000-2009 (Table 6.1, Gilbert and Brandle unpublished data). The diets of *E. minor* were assessed at Troubridge Is., Pearson Is and Reevesby Is (Wiebkin, unpublished data). The stomach contents of *E. minor* were removed using the stomach lavaging technique (Wilson 1984) and weighed. The total mass of prey that each parent returned to its chick was calculated to compare the feeding conditions around colonies on the day of sampling. Data that described the proportions of *E. australis* and *S. sagax* in the diet at each site and the morphology (size and weight) of these prey species were used as separate EPIs (Table 6.1). Prey were identified and their weight and length were estimated by Wiebkin (unpublished data), following the methods of Gales and Pemberton (1990). The durations of foraging trips by adult *E. minor* were calculated at Troubridge Is and Pearson Is from *E. minor* that were carrying (implanted) tags. Each time these *E. minor* came and went from their nests to forage, they passed aerials and a data logger, which read and stored the identity of the tag and the time (Wiebkin, unpublished data).

Data that described the breeding and feeding ecology of *P. tenuirostris*, which were provisioning chicks at Althorpe Is, were taken from Einoder (2010). We calculated EPIs that described the breeding success of *P. tenuirostris*, using data that spanned 3 years (Table 6.1). Ecological performance indicators of breeding success were calculated to describe the proportions of *P. tenuirostris* eggs that resulted in fledglings (Table 6.1). The growth rates of *P. tenuirostris* chicks were calculated at early and late stages in the provisioning period, based on daily mass measurements (Table 6.1). Growth rates of *P. tenuirostris* chicks were modelled and several EPIs that describe the patterns of growth were calculated (Table 6.1). Short and long foraging trips of *P. tenuirostris* are thought to be related to local and distant foraging locations respectively (refer Einoder 2010). To assess local prey availability, we used EPIs that described the provisioning rate (number of meals per chick) and the number and proportion of short (1, 2 and 3 d) foraging trips by adults that were provisioning chicks. The prey of *P. tenuirostris* were recovered using the stomach lavaging technique (Wilson 1984) and the stomach contents were weighed and identified (Einoder 2010). To compare the feeding conditions, the total mass of prey that each parent brought back to its chick was calculated for short (<3 days) and long foraging trips (>3 days) and also for the digested (oil) and raw components of the stomach contents. Data that described the proportions of *E. australis* and *S. sagax* in the diet were used as separate EPIs (Table 6.1).

Results

In total, 183 EPIs were used to describe the trends in abundance and the breeding and feeding ecology of *S. bergii* (44 EPIs), *A. forsteri* (66), *E. minor* (54), *P. tenuirostris* (17) and *T. maccoyii* (2) (Table 6.1). Annual changes in the EPIs based on predators were compared with the five indicators that described the *S. sagax* spawning biomass (3 indicators) and the catch of the SA sardine fishery (2 indicators), leading to 915 relationships that were tested.

It is not possible to present all of the correlation results, because a total of 915 data sets were analysed. We summarised the results for EPIs that described similar aspects of the biology of a predator and the total number of EPIs that were either negatively or positively related to the catch or biomass of *S. sagax* (Tables 6.1 and 6.2). For the significant positive relationships, we present the minimum R value for grouped EPIs and for significant negative relationships, we present the maximum R value for grouped EPIs.

Of the 220 correlations that related the fishery catch/biomass to the breeding and feeding ecology of *S. bergii*, 8 were significant (Tables 6.1 and 6.2). This number of significant correlations did not differ from the number that would be expected by chance alone (P = 0.225). In Spencer Gulf, the proportion of *E. australis* in the diet was negatively related to the *S. sagax* spawning biomass (R = -1.00) and the *S. sagax* catch in the previous year (R = -1.00), but positively related to the *S. sagax* catch in the same year (R = 1.00). At Troubridge Is, the proportion of *E. australis* in the diet showed contrasting relationships with *S. sagax* catch (current and previous years), being positively related in the middle of the breeding season (R = 1.00) and negatively related late in the season (R = -1.00). The bill and head morphology of *S. bergii* chicks were negatively related to the spawning biomass in the previous year (R = -0.90).

Of the 330 correlations that related the fishery catch/biomass to the breeding and feeding ecology of *A. forsteri*, 15 were significant (Tables 6.1 and 6.2). This number of significant correlations did not differ from the number that would be expected by chance alone (P = 0.414). The birth morphology of pups on Kangaroo Is and the Neptune Is was negatively related to the spawning biomass in the current (3 EPIs, R = -1.00) and previous years (1 EPI, R < -0.94) and the catch in the current (6 EPIs, R < -0.47) and previous (2 EPIs, R < -0.52) years. The growth rate of *A. forsteri* pups on Kangaroo Is was positively related to the catch in the previous and current years for 1 EPI (R > 0.97) and negatively related for another (R = -1.00, Tables 6.1 and 6.2).

In total, 10 correlations were run to compare the diet of *T. maccoyii* with the fishery catch/biomass and none of these were significant (Tables 6.1 and 6.2).

Correlations were run to compare the 270 EPIs that described the breeding and feeding ecology of *E. minor* with the fishery catch/biomass indicators. Of these, 49 were significant (Tables 6.1 and 6.2). This number of significant correlations was significantly greater than the number that would be expected by chance alone (P < 0.001). The morphology of *E. australis* in the diet of *E. minor* was negatively related to the spawning biomass in the previous year (R = -1.00). The breeding success (eggs to hatch) at Troubridge Is was positively related to the spawning biomass in the current year (R = 0.99) and the catch in the previous year (R = 0.88). The morphology of fledglings at Pearson Is (Great Australian Bight) was positively related to the spawning biomass in the previous year (R = 1.00). The change in the population size of *E. minor* at Granite Is was negatively related to the spawning biomass in the current (R < -0.72), next (R < -0.79) and previous (R < -0.75) years (each had >7 significant EPIs) and the catch in the current (8 EPIs, R < -0.77) and previous (6 EPIs, R < -0.67) years. The foraging trip duration of *E. minor* at Troubridge Is was positively related to the catch in the previous year (R = 1.00, Tables 6.1 and 6.2).

Of the 85 correlations that related the fishery catch/biomass to the breeding and feeding ecology of *P. tenuirostris*, 2 were significant (Tables 6.1 and 6.2). This number of significant correlations did not differ from the number that would be expected by chance alone (P = 0.196). The daily mass change and growth rates of *P. tenuirostris* chicks were negatively related to the catch in the current year (R = -1.00 in both cases, Tables 6.1 and 6.2).

Table 6.1. The number of EPIs that were significant for each correlation. EPIs that describe similar relationships are grouped, but each relationship is for a different test, based on different data sets (e.g. the EPI for crested terns *anchovy proportion in diet* summarises the relationship between the diet of crested terns, at several different times of the year versus the fishery parameters).

Predator group and indicator	**Years**	**No. indicators**	**Spawning biomass (current year)**		**Spawning biomass (next year)**		**Spawning biomass (prev. year)**		**Catch (current year)**		**Catch (prev. year)**	
			Positive R	**Negative R**	**Positive R**	**Negative R**	**Positive R**	**Negative R**	**Positive R**	**Negative R**	**Positive R**	**Negative R**
SA sardine fishery												
Spawning biomass	1995-2007	3	-	-	-	-	-	-	-	-	-	-
Catch	1991-2009	2	-	-	-	-	-	-	-	-	-	-
	Total	**5**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**
Crested tern - *Sterna bergii*												
Anchovy propn in diet (early, mid and late season, GSV)	2003-2007	2	0	0	0	0	0	0	0	0	0	0
Anchovy propn in diet (early, mid and late season, GSV)	2004-2007	2	0	0	0	0	0	0	0	0	0	0
Anchovy propn in diet (early, mid and late season, GSV)	2005-2007	4	0	0	0	0	0	0	1	1	1	0
Anchovy propn in diet (early, mid and late season, Spencer)	2005-2007	5	0	1	0	0	0	0	1	0	0	1
Bill and head morphology and body mass, GSV	1985-2000	8	0	0	0	0	0	2	0	0	0	0
Sardine propn in diet (early, mid and late season, GSV)	2003-2007	2	0	0	0	0	0	0	0	0	0	0
Sardine propn in diet (early, mid and late season, GSV)	2004-2007	2	0	0	0	0	0	0	0	0	0	0
Sardine propn in diet (early, mid and late season, GSV)	2005-2007	4	0	0	0	0	0	0	0	0	0	0
Sardine propn in diet (early, mid and late season, Spencer)	2005-2007	5	0	0	0	0	0	0	0	0	0	0
Survival rate (observed minus modelled), GSV	1985-2000	6	0	0	0	0	0	0	0	0	0	0
Survival rate (observed minus modelled), GSV	1985-2002	2	0	0	0	0	0	0	0	0	0	0
Survival rate (observed minus modelled), GSV	1986-2003	2	0	0	0	0	0	0	0	0	0	0
	Total	**44**	**0**	**1**	**0**	**0**	**0**	**2**	**2**	**1**	**1**	**1**
New Zealand fur seal - *Arctocephalus forsteri*												
Birth morphology, KI	1988-2009	4	0	0	0	0	0	0	0	4	0	2
Birth morphology, KI	1989-2007	4	0	0	0	0	0	0	0	0	0	0
Birth morphology, KI	2004-2008	2	0	0	0	0	0	1	0	0	0	0
Birth morphology, N Neptune	2006-2008	4	0	0	0	0	0	0	0	2	0	0
Birth morphology, N&S Neptune	2005-2008	4	0	3	0	0	0	0	0	0	0	0
Growth rate cm per day, KI	1995-2006	1	0	0	0	0	0	0	0	0	0	0
Growth rate cm per day, KI	1995-2008	1	0	0	0	0	0	0	0	0	0	0
Growth rate cm per day, KI	1995-2009	2	0	0	0	0	0	0	0	0	0	0
Growth rate cm per day, KI	2001-2007	4	0	0	0	0	0	0	0	0	0	0
Growth rate cm per day, KI	2004-2006	2	0	0	0	0	0	0	0	0	1	0
Growth rate cm per day, KI	2004-2007	5	0	0	0	0	0	0	1	0	0	0
Growth rate cm per day, KI	2004-2010	4	0	0	0	0	0	0	0	0	0	0
Growth rate cm per day, KI	2005-2007	8	0	0	0	0	0	0	0	0	0	1
Growth rate cm per day, KI	2005-2009	7	0	0	0	0	0	0	0	0	0	0
Growth rate cm per day, KI	2005-2010	4	0	0	0	0	0	0	0	0	0	0
Growth rate cm per day, N Neptune	2005-2007	2	0	0	0	0	0	0	0	0	0	0
Population change KI	1989-2007	2	0	0	0	0	0	0	0	0	0	0
Population change KI	1989-2009	1	0	0	0	0	0	0	0	0	0	0
Population change N Neptune	2005-2007	1	0	0	0	0	0	0	0	0	0	0
Population change N&S Neptune, Liguanea	2005-2007	2	0	0	0	0	0	0	0	0	0	0
Population change S Neptune	2005-2007	1	0	0	0	0	0	0	0	0	0	0
Population change, Liguanea	2005-2007	1	0	0	0	0	0	0	0	0	0	0
	Total	**66**	**0**	**3**	**0**	**0**	**0**	**1**	**1**	**6**	**1**	**3**
Bluefin tuna - *Thunnus maccoyii*												
Anchovy propn in diet, SA	1999-2006	1	0	0	0	0	0	0	0	0	0	0
Sardine propn in diet, SA	1999-2006	1	0	0	0	0	0	0	0	0	0	0
	Total	**2**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**

Table 6.1. (cont.)

Predator group and indicator	**Years**	**No. indicators**	**Spawning biomass (current year)**		**Spawning biomass (next year)**		**Spawning biomass (prev. year)**		**Catch (current year)**		**Catch (prev. year)**	
			Positive R	**Negative R**	**Positive R**	**Negative R**	**Positive R**	**Negative R**	**Positive R**	**Negative R**	**Positive R**	**Negative R**
Little penguin - *Eudyptula minor*												
Anchovy morphology, Pearson	2003-2005	2	0	0	0	0	0	0	0	0	0	0
Anchovy morphology, Reevesby	2003-2005	2	0	0	0	0	0	1	0	0	0	0
Anchovy morphology, Troubridge	2003-2006	2	0	0	0	0	0	0	0	0	0	0
Anchovy propn in diet, Pearson	2003-2005	1	0	0	0	0	0	0	0	0	0	0
Anchovy propn in diet, Reevesby	2003-2005	1	0	0	0	0	0	0	0	0	0	0
Anchovy propn in diet, Troubridge	2003-2006	1	0	0	0	0	0	0	0	0	0	0
Breeding success, chicks to fledge, Granite	1990-2006	1	0	0	0	0	0	0	0	0	0	0
Breeding success, eggs to fledge, Granite	1990-2006	1	0	0	0	0	0	0	0	0	0	0
Breeding success, eggs to fledge, Granite	1991-2009	1	0	0	0	0	0	0	0	0	0	0
Breeding success, eggs to fledge, Troubridge	2004-2009	1	0	0	0	0	0	0	0	0	0	0
Breeding success, eggs to fledge, Troubridge	2004-2009	1	0	0	0	0	0	0	0	0	0	0
Breeding success, eggs to hatch, Granite	1990-2006	1	0	0	0	0	0	0	0	0	0	0
Breeding success, eggs to hatch, Troubridge	2004-2009	1	1	0	0	0	0	0	0	0	1	0
Breeding success, eggs to hatch, Troubridge	2004-2009	1	0	0	0	0	0	0	0	0	0	0
Breeding success, hatch to fledge, Troubridge	2004-2009	1	0	0	0	0	0	0	0	0	0	0
Breeding success, per pair, Troubridge	2004-2009	1	0	0	0	0	0	0	0	0	0	0
Breeding success, per pair, Troubridge	2004-2009	1	0	0	0	0	0	0	0	0	0	0
Fledging morphology, Pearson	2004-2006	1	0	0	1	0	0	0	0	0	0	0
Fledging morphology, Troubridge	2004-2009	3	0	0	0	0	0	0	0	0	0	0
Meal mass after trip, Pearson	2003-2005	2	0	0	0	0	0	0	0	0	0	0
Meal mass after trip, Reevesby	2003-2005	2	0	0	0	0	0	0	0	0	0	0
Meal mass after trip, Troubridge	2003-2006	2	0	0	0	0	0	0	0	0	0	0
Population change Granite	2000-2007	8	0	7	0	7	0	7	0	5	0	4
Population change Granite	2000-2008	1	0	1	0	0	0	1	0	1	0	1
Population change Granite	2001-2008	3	0	3	0	0	0	3	0	1	0	0
Population change Granite	2001-2009	1	0	0	0	0	0	1	0	1	0	1
Sardine morphology, Reevesby	2003-2005	2	0	0	0	0	0	0	0	0	0	0
Sardine morphology, Troubridge	2003-2006	2	0	0	0	0	0	0	0	0	0	0
Sardine propn in diet, Pearson	2003-2005	1	0	0	0	0	0	0	0	0	0	0
Sardine propn in diet, Reevesby	2003-2005	1	0	0	0	0	0	0	0	0	0	0
Sardine propn in diet, Troubridge	2003-2006	1	0	0	0	0	0	0	0	0	0	0
Trip duration, Pearson	2004-2006	2	0	0	0	0	0	0	0	0	0	0
Trip duration, Troubridge	2004-2006	1	0	0	0	0	0	0	0	0	1	0
Trip duration, Troubridge	2004-2009	1	0	0	0	0	0	0	0	0	0	0
	Total	**54**	**1**	**11**	**1**	**7**	**0**	**13**	**0**	**8**	**2**	**6**
Short-tailed shearwater - *Puffinus tenuirostris*												
Anchovy propn in diet, Althorpe	2004-2006	1	0	0	0	0	0	0	0	0	0	0
Breeding success, eggs to fledge, Althorpe	2004-2006	1	0	0	0	0	0	0	0	0	0	0
Daily mass change, mid and late season, Althorpe	2004-2006	2	0	0	0	0	0	0	0	1	0	0
Growth rate, peak mass, r-squared, g/d, Althorpe	2004-2006	3	0	0	0	0	0	0	0	1	0	0
Meal mass after trip (short and long), Althorpe	2004-2006	2	0	0	0	0	0	0	0	0	0	0
Meal weight (oil + raw, raw only), Althorpe	2003-2006	2	0	0	0	0	0	0	0	0	0	0
No. successive short trips, late in season, Althorpe	2004-2006	1	0	0	0	0	0	0	0	0	0	0
Propn of 1, 2, 3 d short trips, Althorpe	2004-2006	3	0	0	0	0	0	0	0	0	0	0
Provisioning rate, Althorpe	2004-2006	1	0	0	0	0	0	0	0	0	0	0
Sardine propn in diet, Althorpe	2004-2006	1	0	0	0	0	0	0	0	0	0	0
	Total	**17**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**2**	**0**	**0**
Total (all predators, excl. fishery)		**183**	**1**	**15**	**1**	**7**	**0**	**16**	**3**	**17**	**4**	**10**

Table 6.2. Correlation coefficient for significant correlations (maximum coefficient for negative correlations and minimum for positive correlations). EPI that describe similar relationships are grouped, but each relationship is for a different test, based on different data sets (e.g. the EPI for crested terns *anchovy proportion in diet* summarises the relationship between the diet of crested terns, at several different times of the year versus the fishery parameters).

Predator group and indicator	**Positive or negative correlation**	**Spawning biomass (current year)**		**Spawning biomass (next year)**		**Spawning biomass (prev. year)**		**Catch (current year)**		**Catch (prev. year)**	
		Positive R	**Negative R**	**Positive R**	**Negative R**	**Positive R**	**Negative R**	**Positive R**	**Negative R**	**Positive R**	**Negative R**
Crested tern - *Sterna bergii*											
Anchovy propn in diet (early, mid and late season, GSV)	Negative	-	-	-	-	-	-	-	-1.00	-	-
Anchovy propn in diet (early, mid and late season, GSV)	Positive	-	-	-	-	-	-	1.00	-	1.00	-
Anchovy propn in diet (early, mid and late season, Spencer)	Negative	-	-1.00	-	-	-	-	-	-	-	-1.00
Anchovy propn in diet (early, mid and late season, Spencer)	Positive	-	-	-	-	-	-	1.00	-	-	-
Bill and head morphology and body mass, GSV	Negative	-	-	-	-	-	-0.90	-	-	-	-
New Zealand fur seal - *Arctocephalus forsteri*											
Birth morphology, KI	Negative	-	-	-	-	-	-0.94	-	-0.47	-	-0.52
Birth morphology, N Neptune	Negative	-	-	-	-	-	-	-	-1.00	-	-
Birth morphology, N&S Neptune	Negative	-	-1.00	-	-	-	-	-	-	-	-
Growth rate cm per day, KI	Negative	-	-	-	-	-	-	-	-	-	-1.00
Growth rate cm per day, KI	Positive	-	-	-	-	-	-	0.97	-	1.00	-
Little penguin - *Eudyptula minor*											
Anchovy morphology, Reevesby	Negative	-	-	-	-	-	-1.00	-	-	-	-
Breeding success, eggs to hatch, Troubridge	Positive	0.99	-	-	-	-	-	-	-	0.88	-
Fledging morphology, Pearson	Positive	-	-	1.00	-	-	-	-	-	-	-
Population change Granite	Negative	-	-0.72	-	-0.79	-	-0.75	-	-0.77	-	-0.67
Trip duration, Troubridge	Positive	-	-	-	-	-	-	-	-	1.00	-
Short-tailed shearwater - *Puffinus tenuirostris*											
Daily mass change, mid and late season, Althorpe	Negative	-	-	-	-	-	-	-	-1.00	-	-
Growth rate, peak mass, r-squared, g/d, Althorpe	Negative	-	-	-	-	-	-	-	-1.00	-	-

Discussion

Our study compared the fishery catch and spawning biomass of *S. sagax* with EPIs that described the breeding and feeding ecology of key predators in the same region. The fishery catch of *S. sagax* and the spawning biomass were related to many of the EPIs that described the biology of the predators and as such, these relationships could potentially be used by fishery managers to monitor changes in the availability of *S. sagax* and to aid in the assessment of the need for an ecological allocation of *S. sagax*. Potential EPIs for the SA sardine fishery are listed in Table 6.3 and discussed below.

Table 6.3. Potential EPI to implement EBFM in the SA sardine fishery. The EPIs are based on the breeding and feeding ecology of *E. minor, P. tenuirostris, S. bergii* and *A. forsteri,* and were found to be either negatively related to the fishery catch of *S. sagax* or positively related to the spawning biomass of *S. sagax.* Estimates of the number of days required to collect the data are provided.

Parameter	Location	Method to record parameter annually
E. minor		
Breeding success	Troubridge Is or Lewis Is or Reevesby Is	Record survival rates of eggs and chicks for 30 burrows (30 days)
P. tenuirostris		
Growth rate (g/d) of chicks	Althorpe Is or North Islet	Weigh 50 chicks every 2 days in January (30 days)
S. bergii		
Morphology of fledglings	Troubridge Is and Donington Rock	Measure bill length and weigh 200 chicks (16 days)
Survival of fledglings	Troubridge Is and Donington Rock	Band 1000 chicks (10 days) and recapture 200 known-age adults (16 days)
A. forsteri		
Birth mass and length of pups	Cape du Coedic, Kangaroo Is or North Neptune Is	Measure length and weigh 60 pups in late January (4 days)
Growth rate (g/d, cm/d) of pups	Cape du Coedic, Kangaroo Is or North Neptune Is	Measure length and weigh 60 pups in January, April and September (12 days)

Practical EPIs need to be sensitive to changes in the ecosystem that is being monitored, whether they are caused by environmental or anthropogenic impacts, but should not covary with other factors (Hiddink and Kaiser 2005). We did not assess whether the EPIs were related to factors such as the catch or biomass of other exploited fish stocks, strength and timing of seasonal upwelling (e.g. Middleton 2007), abundance or distribution of chlorophyll, sea surface temperature, or other factors. Covariates reduce the reliability of EPIs and increase the number and cost of EPIs that need to be monitored. The most practical EPIs based on predator ecology have simple, linear relationships with fishery catches or stock sizes (reviewed in Einoder 2009). Our analyses were only able to detect linear relationships and so we would not have detected relationships that were non-linear, which are common in ecology (e.g. Reid *et al.* 2005). Nonetheless, the simple, linear EPIs, which we found to be significantly related to the catch and biomass of *S. sagax*, serve as a starting point to determine if ecological allocation in needed for *S. sagax*.

Because both the SASF catch and spawning biomass have typically increased from one year to the next, any EPIs that were increasing or decreasing over the same period are likely to result in a significant relationship. Thus there is a risk that some of the relationships may be coincidental, because a significant correlation cannot be taken to mean causation. For example, the population of *E. minor* at Granite Is (about 250 km from the centre of the fishery) has been in decline for about 10 years and so the correlations between the indices that described the size of the penguin population and *S. sagax* availability were highly significant (Tables 6.1 and 6.2). The cause of the decline in the penguin population at Granite Is is not known, but predation by introduced rats (*Rattus rattus*) and increasing populations of *A. forsteri*, and disturbance caused by tourists may be partly responsible (Bool *et al.* 2007). The low number of data points available for some of the EPIs (n = 3 for many relationships, refer Table 6.1), exacerbates this issue, and resulted in many correlations with R=1 or R=-1. These EPIs all need to be confirmed with additional data over the coming years.

It is important to highlight that several of the significant relationships between EPIs that described the breeding and feeding ecology of predators and the catch and biomass of *S. sagax* were contrary to our expectations (i.e. positively related to spawning biomass or negatively related to fishery catch). For example, the morphology of *S. bergii* was negatively related to the size of the spawning biomass of *S. sagax* in the previous year. This finding does not support those of McLeay *et al.* (2009b), who showed that *S. bergii* were smaller and had lower survival rates if they were from years when *S. sagax* availability was very low, which was the case after the two mass mortalities of *S. sagax* (Ward *et al.* 2008). Our analyses may have been too crude to detect these differences, possibly because the biomass of *S. sagax* and the breeding performance of *S. bergii* quickly recovered following the mass mortalities of *S. sagax* (Ward *et al.* 2008, McLeay *et al.* 2009b). Other unexpected results included: 1) the negative relationship between the spawning biomass and the morphology of *A. forsteri* pups at the Neptune Islands and on Kangaroo Is, and 2) the negative relationships between the spawning biomass of *S. sagax* and the EPIs that described the breeding and feeding ecology of *E. minor* at Granite Is, as discussed above. These relationships may also be coincidental or they may reflect the benefits of reduced availability of *S. sagax* for these predators. Some other significant relationships between EPIs that described the breeding and feeding ecology of predators and the catch and biomass of *S. sagax* were indicative of an abundance of resources in the ecosystem. For example, the positive relationships between fishery catch and the 1) growth rates of *A. forsteri* pups on Kangaroo Is, and 2) *E. minor* breeding success and trip duration on Troubridge Is are indicative of abundant prey for these predators and the fishery.

Because our data analyses were conducted to find EPIs that were related to the availability of *S. sagax*, it is important to assess how well the suite of indicators met those expectations. The binomial probability tests indicated that, for *S. bergii*, *A. forsteri*, and *P. tenuirostris* , there is a high likelihood that the significant correlations detected occurred by chance alone. This issue would be resolved with further monitoring of the same relationships.

The population parameters of some predators were related to the availability of *S. sagax*. For predator species that were adversely impacted by a reduction in the availability of *S. sagax*, we expected to find positive correlations between *S. sagax* spawning biomass and EPIs that described the breeding and feeding ecology of predators. We also expected the EPIs of these predators to be negatively correlated with the catch of the fishery. We did not have any data to compare the breeding ecology of the juvenile *T. maccoyii* to the catch or biomass of *S. sagax* and none of the EPIs that described the feeding ecology of *T. maccoyii* were related to the catch or biomass of *S. sagax*. For the other predators, the EPIs positively related to spawning biomass or negatively related to fishery catch included those that described the morphology and growth rates of *A. forsteri* pups on Kangaroo Is and Neptune Is, the breeding success of *E. minor* at Troubridge Is, the morphology of crested tern fledglings from Troubridge Is and the growth of *P. tenuirostris* chicks on Althorpe Is. These are potentially useful EPIs of the availability of *S. sagax* and the potential impacts of the *S. sagax* fishery in this region. Findings such as these, which summarise relationships between the breeding and feeding ecology of predators and catch and spawning biomass of a fishery, are common in the literature, but cases where such findings are used to improve the fisheries management are rare (e.g. Sainsbury *et al.* 2000).

The relationship between the spatial distributions of *S. sagax* and *E. australis* has received much research attention, because they are morphologically similar and they appear to occupy a similar ecological niche. Diet studies conducted on these species in South Australia indicate that their diets are broadly similar, but there are significant differences in some of their prey (Daly 2007). In an attempt to understand how *S. sagax* and *E. australis* compete for space, studies have monitored the distribution of eggs, larvae and adult fish (e.g. Barange and Hampton 1997). In South Australia, *S. sagax* and *E. australis* are not spatially segregated and they are thought to compete for spawning habitats, which was inferred from changes in the relative abundance of the eggs and larvae of the two species (Ward *et al.* 2001 a,b). Our results indicate that when the spawning biomass of *S. sagax* was relatively high (or the catch of *S. sagax* was relatively low), the proportion of *E. australis* decreased in the diet of *S. bergii* in Spencer Gulf and at Troubridge Is. In addition, when the spawning biomass of *S. sagax* in the previous year increased, the morphology of individual *E. australis* was relatively small in *E. minor* samples from Reevesby Is. These results support the findings of previous studies, which indicate that adult *E. australis* are displaced from near-shore to offshore waters when *S. sagax* biomass is relatively high (Ward *et al.* 2001 a,b and Barange and Hampton 1997).

The EPIs that were related to the *S. sagax* biomass or fishery catch require further monitoring to confirm that the relationships are real. If validated, the EPIs could be incorporated as reference points in the management plan for the SA sardine fishery. If a reference point of an EPI was triggered, it would require investigation to confirm whether this was caused by a reduction in the availability of *S. sagax*. If subsequent investigations indicate that reductions in the biomass of *S. sagax* were not responsible, the reference points may need to be adjusted. Such ongoing data collection for the EPIs listed in Table 6.3 could inform and refine the management plan for the SA sardine fishery.

The EBFM for the SA sardine fishery could involve the monitoring of aspects of the breeding and feeding ecology of *S. bergii*, *A. forsteri*, *E. minor* and *P. tenuirostris* to reduce the risk of the fishery having long-term impacts on their populations. Table 6.3 details a potential monitoring program for the SA sardine fishery, including estimates of the number of days required to record these data. Monitoring of the EPIs derived from *E. minor* and *P. tenuirostris* would be focused on EPIs that describe breeding success (*E. minor*) and growth rates (*P. tenuirostris*) (Table 6.3). These predators are synchronous breeders, enabling data to be obtained efficiently at any breeding colony. These EPIs could be efficiently monitored at sites that are close to the centre of the fishery, such as Lewis Is or Reevesby Is (*E. minor*) and Althorpe Is or North Islet (*P. tenuirostris*) (Table 6.3).

The EPIs derived from *A. forsteri*, which were related to the availability of *S. sagax*, were based on the birth morphology and growth rates of pups (Table 6.3). Pups remain ashore while lactating females forage at sea. Because lactating females are central place foragers that use shelf waters between summer and winter (Page *et al.* 2006, Baylis *et al.* 2008b, Page and Goldsworthy unpublished data), the growth rates of pups integrate information about the availability of prey. Samples of *A. forsteri* pups can be weighed easily and quickly, so that rates of growth can be calculated (e.g. Goldsworthy *et al.* 2006). Colonies of *A.*

forsteri on Kangaroo Is or North Neptune Is are the most appropriate sites for further monitoring because: 1) there are data sets that span more than 20 years for these EPIs, 2) the colonies are easy to access, and 3) females from these colonies use waters immediately to the south of Spencer Gulf, where the fishery is centred (Baylis *et al.* 2008 a,b, Page and Goldsworthy unpublished data) (Table 6.3).

Based on our understanding of the trophic (Chapter 3 and 5) and spatial (Chapter 4) dynamics of *S. bergii*, *A. forsteri*, *E. minor* and *P. tenuirostris*, the most likely species that would be affected by a change in the availability of *S. sagax* is *S. bergii*. As discussed in chapter 3, *S. sagax* are a very small part of the diet of *A. forsteri*, *E. minor* and *P. tenuirostris* and so even if future research showed that EPIs based on these predators were significantly related to *S. sagax* availability, the relationship is not likely to be direct. The results of the trophodynamic model support these conclusions, because *S. bergii* demonstrated the greatest sensitivity to reduction in *S. sagax* biomass (Chapter 5). Given these findings, the most appropriate form of EBFM for the SA sardine fishery would include the monitoring the EPIs based on *S. bergii*, as outlined in Table 6.3.

Because the morphology and survival of *S. bergii* at Troubridge Is have been shown to be related to the availability of *S. sagax* (McLeay *et al.* 2009b), this species requires ongoing monitoring at Troubridge Is (Table 6.3). Colonies of *S. bergii* are readily accessible and at breeding colonies, large proportions of the adult birds can be captured, identified and/or aged (from bands applied when they were chicks), measured and released in a relatively brief period, making this a cost-effective species for monitoring. Efforts should also be made to monitor the morphology and survival of *S. bergii* at a colony in Spencer Gulf (e.g. Donington Rock), near the centre of the SA sardine fishery, because the effects of localised depletions of *S. sagax* are likely to be most apparent there (Table 6.3).

This study focussed on EPIs that described the relationships between the biology of predators and the availability of *S. sagax*. Goldsworthy *et al.* (Chapter 5) used an alternative approach – trophodynamic modelling – to address the same question from a broader perspective. Their model simulated the dynamics of the ecosystem by incorporating all available data on the ecosystem, including: 1) the diets and demographics of the predators in the region, 2) annual changes in environmental features such as upwelling, 3) the annual catches of the fisheries in the region. In addition to confirming the potential for *S. bergii* as an EPI, Goldsworthy *et al.* (Chapter 5) indicated that the current level of the SA sardine fishery was unlikely to be adversely affecting the ecosystem. The ability to evaluate the role of multiple components of the ecosystem, including the impacts of different fisheries, highlights the benefit of using this broad-based approach to monitor changes in key EPIs of the ecosystem. Periodic updates, for example every five years, of the trophodynamic model, including further data inputs (based on the biology of predators, catches of fisheries and environmental features) and improvements to the design of the model, together with the annual monitoring of the EPIs discussed in this chapter, will ensure that significant progress is made toward the EBFM of the SA sardine fishery.

Monitoring of this ecosystem in such a manner would provide fishery managers with data that may help them alleviate the impacts of localised depletions caused by the fishery, for example through quota and/or spatial management of fishing activity. This approach is employed by fishery managers of: 1) the sandeel (*A. hexapterus*) fishery in the North Sea (Kabuta and Laane 2003), and 2) *E. superba* fishery in the Antarctic to minimise the impact on the natural predators of *E. superba*, as part of CCAMLR (Convention on the Conservation of Antarctic Marine Living Resources). Annual monitoring of EPIs for *S. bergii* and periodic updates of trophodynamic models will also ensure that monitoring is focussed on cost effective methods, which provide fishery managers with a means to assess the effects of their management strategies.

7 BENEFITS AND ADOPTION

Industry/community sectors benefiting from research

The SASF has a high public profile and is subject to considerable external scrutiny because it is the largest single-species fishery in Australia and targets a forage species that is widely acknowledged to play a critical role in converting planktonic biomass into a form that is accessible to large predatory species, which have high conservation value.

The main findings of this study, i.e. that current harvest levels in the SASF are not adversely affecting ecosystem function or the populations of key predators, are critically important to licence-holders in the SASF and the tuna aquaculture industry of southern Spencer Gulf, which utilises much of the catch, because they help to ensure ongoing access to this valuable resource.

The benefits to conservation agencies and other fisheries are also significant because this project delivers the type of ecosystem-based assessment of the potential impacts of fishing that is often requested or recommended, but rarely undertaken at the scale of this project.

The findings in this report provide conservationists and the general public with evidence that fisheries management of the SASF is functioning effectively, despite the gloomy global prognosis for many fisheries.

The State and Commonwealth agencies responsible for the management of Australia’s living marine resources also benefit from the evidence that shows not only is the SASF being managed according to the principle of ESD, but that the valuable Flinders Current Ecosystem off southern Australia is functioning well and remains productive. This study also benefits these agencies by providing, arguably for the first time, the data and tools required to objectively assess the potential benefits of establishing ecological performance indicators and reference points for a pelagic fishery in Australian waters.

One example highlights the international recognition for ecological sustainability that this and previous studies have delivered to the SASF. In October 2009, the Marine Stewardship Council used the SASF as a case study to assist the development of a new Fishery Assessment Methodology for the management of fisheries for low trophic level species.

This project provided substantial opportunities for post graduate research in South Australia. This project collated data from 10 projects (6 PhD and 4 honours), which were entirely or partly funded by this FRDC project. The names of the students and their projects are listed below.

PhD projects

Dr Alastair Baylis – The University of Adelaide. Project title: Seasonal and colony differences in the foraging ecology of New Zealand fur seals.

Dr Luke Einoder – The University of Adelaide. Project title: Feeding and breeding ecology of the short-tailed shearwater in South Australia.

Dr Lachlan McLeay – The University of Adelaide. Project title: The life history characters, reproductive constraints and foraging strategies of the crested tern.

Dr Annelise Wiebkin – The University of Adelaide. Project title: The feeding and breeding ecology of little penguins in South Australia.

Dr Kristian Peters – The University of Adelaide. Project title: Molecular and morphological analyses of Australian sea lion diet.

Mr Paul Rogers – Flinders University. Project title: Movement, diet and population structure of pelagic shark populations in southern Australia.

Honours projects

Dr Robin Caines – The University of South Australia. Project title: Trophic Guild-Structures in Pelagic Fish Communities in the Eastern Great Australian Bight Ecosystem.

Ms Natalie Bool – The University of Adelaide. Project title: What is causing the decline of little penguins (*Eudyptula minor*) on Granite Island, South Australia?

Ms Michelle Roberts – The University of Adelaide. Project title: Assessment of morphological and molecular diet analysis techniques: application of a combined approach to resolve the diet of arrow squid, *Nototodarus gouldi* in the eastern Great Australian Bight.

Ms Kerryn Daly – The University of Adelaide. Project title: The diet and guild structure of the small pelagic fish community in the eastern Great Australian Bight, South Australia.

Adoption of the research by identified beneficiaries

The results of this project will be used to assess the potential benefits of establishing ecological performance indicators and reference points in the management plan for the SASF. The ecosystem model that was developed will be available for ongoing analyses, including regular generation (e.g. every five years) of indicators of ecosystem health (Kempton’s Q biodiversity index, Fishing in Balance index). Similarly, dietary and foraging data collected will provide a basis for developing time series of potential ecological performance indicators such as breeding success of crested terns. The adoption of these findings will help to ensure that Australia’s largest fishery continues to meet, exceed and set internationally-recognised standards for world’s best practice in the ecosystem-based management of fisheries for low trophic level species.

Summary of project extension to beneficiaries

The research and management working groups for the SASF have regularly been updated on the progress of the project. Presentations of preliminary results have also been made to the Resource Assessment Group of the Commonwealth Small Pelagic Fishery. Final results will be presented to both of these groups early in 2011. Preliminary results have also been presented to the Marine Stewardship Council (Washington, DC, October 2009) and Sardine Tri-national Forum (Victoria Canada November 2010).

How benefits and beneficiaries compare to those identified in the original application

The benefits and beneficiaries of the project are consistent with those identified in the original application. The proposal indicated that the results of this study would assist the SASF, PIRSA Fisheries and SARDI Aquatic Sciences to revise the management plan for the SASF and establish cost-effective systems for ongoing monitoring and assessment of the ecological effects of the SASF. This will be done during the revision of the management plan for the SASF, which is scheduled to be completed by 30 June 2013. Potential ecological performance indicators identified in this project include the model-based indices (e.g. Kempton’s Q biodiversity index, Fishing in Balance index) and foraging patterns and reproductive success of the crested tern.

8 FURTHER DEVELOPMENT

The current fishery management strategy appears to be sufficiently conservative to ensure that the fishery is managed according to the principles of ecological sustainable development (ESD). However, the extent to which this positive assessment reflects management of the sardine fishery is uncertain, given that much of the positive changes estimated by the model reflect other changes in the ecosystem, including reductions in fishing effort and mortality in some fleets (SA marine scalefish net and Commonwealth demersal gillnet shark fishery), and a positive trend in primary production over the study period. How the current or alternate management strategies of the sardine fishery would perform under alternate oceanographic conditions (such as more variable upwelling), and/or in response to management changes in other fisheries in the EGAB region, may warrant consideration.

The trophodynamic models developed in Chapter 5 provide a basis for examining additional management measures and ecological questions. The performance indicators produced by the *Ecosim* model also provide a means to assess the potential impacts of the SASF relative to those of other fisheries and environmental change. The ability to resolve and attribute potential impacts from multiple fishing fleets and environmental changes provided by these models will be critical for the development of ecological performance indicators for assessing ESD targets.

The results of this study indicate that one potential EPI warrants further investigation. McLeay et al (2009) showed that large reductions sardine biomass may affect the foraging patterns and reproductive success of the crested tern. This potential for reproductive parameters for this species to be incorporated into the management plan for the SASF warrants further investigation.

9 PLANNED OUTCOMES

Social, economic and ecological advantages

The South Australian community, tuna farmers and participants in the SASF have benefited from ongoing access to local sardines and a reduced need to import baitfish into Australia. The project is helping to maintain the status of the SASF as one of Australia's best managed fisheries, and Australia's position as a world leader in EBFM.

Ecologically sustainable management of the SA sardine fishery

The ecosystem and trophodynamic model developed in this study provides an intellectual framework and practical tool to underpin EBFM of the Flinders Current ecosystem. These models, which take into account the spatial and temporal variability of the eastern GAB ecosystem, will allow fishery managers to investigate the potential consequences of increasing the SASF quote, on ecosystem function and the populations of several protected species. This study also identified that there would be benefits in establishing a monitoring program to assess the potential for using reproductive parameters of the crested tern *S. bergii* as ecological performance indicators in the SASF.

Conservation of seabird and marine mammal populations

Information on the feeding grounds of key predators that this report provided has facilitated a better understanding of the environmental mechanisms that underpin the ecology of this region. The SA Department of Environment and Natural Resources will use this and other information collected in this study to manage marine mammal, penguin and other seabird populations that breed on islands in the GAB ecosystem.

10 CONCLUSIONS

DIETS OF MARINE PREDATORS IN SOUTH AUSTRALIA

This study indicated that, based on their diets, the 37 predator groups considered could be categorised into 8 feeding guilds. The SA sardine fishery (based on its catch) was included in a guild with 10 predator groups (common thresher shark, west Australian salmon, common dolphin, little penguin, bonito, Australian gannet, snook, crested tern, southern bluefin tuna and barracouta), which were considered as the species that are most likely to be directly impacted by competition with the SASF. The importance of sardines to the predators in this guild and to some of the other predators in different guilds (e.g. bronze whaler sharks, Gould's squid and kingfish) highlights the benefits of ongoing monitoring of ecosystem processes in the region where the SA sardine fishery is located.

SPATIAL DISTRIBUTION OF CONSUMPTION EFFORT OF KEY APEX PREDATORS AND THEIR OVERLAP WITH THE SARDINE FISHERY

Most of the consumption by the five key land-breeding apex predators studied consisted of fish (53%), squid (39%) and crustaceans (7%). Small pelagic fish accounted for 28% of the total consumption and 52% of the total fish consumed in the EGAB (Table 4.3). Overall, sardines only made up about 1% of the total prey biomass consumed by the five apex predators, and only 2% of the total fish biomass consumed (Table 4.3). The total estimated consumption of sardines by these predators (753 t/y) is very small (3%) relative to the current annual TACC (30,000 t) of the SA sardine fishery. As such, the catch of sardines by the fishery exceeds the consumption by the five apex predators wherever fishing effort occurs. Outside of the area used by the fishery there are large areas where consumption of sardines by the five apex predators exceeds that of the fishery (Figure 4.2. F).

Based on the analysis of the spatial distribution of prey consumption by five key land-breeding apex predators in the EGAB ecosystem, crested terns were the only species identified for which sardine consumption represented a significant component of total prey consumed (22.7%), and this consumption overlapped with the core region used by the SASF in southern Spencer Gulf and Investigator Strait (Figure 4.1. and 4.2.). Therefore, of all the species investigated, crested terns are most likely to provide potential ecological performance indices for the SA sardine fishery, because spatial and consumption analyses suggest they may integrate information on the availability of sardines into their foraging, reproductive and population ecology.

TROPHODYNAMICS OF THE EASTERN GREAT AUSTRALIAN BIGHT PELAGIC ECOSYSTEM

The growth of the sardine fishery in the EGAB region since its establishment in 1991 has been rapid, and its catch now exceeds that of all other fisheries by a factor of three. Despite this, sensitivity analyses based on mixed trophic impacts detected negligible impacts on other predators. The ecosystem performance indicators produced by the *Ecosim* model provide a means to assess the potential impacts of the sardine fishery relative to those from other fisheries and environmental change. These model-based indicators could be generated every 5 years to monitor the health of the ecosystem. The trophodynamic model indicated that crested terns demonstrated the greatest sensitivity to reduction in sardine biomass, supporting the findings of Chapters 3 and 4 and highlighting their potential to be used as EPIs for the SASF.

The use of trophodynamic models as a tool to provide context to the potential impacts and management strategies of a single fishery relative to the temporal changes of a complex dynamic ecosystem subject to dynamic impacts from multiple fishing fleets and climate change, is highlighted by this study. The model provides a basis from which future improvements in model design and data inputs will enable more complex management and ecological questions to be examined.

ECOLOGICAL PERFORMANCE INDICATORS (EPIS) FOR NATURAL PREDATORS OF SARDINE IN SOUTHERN AUSTRALIA

Most of the datasets that described the ecological performance of predators were collected over the 3 or 4 year period of the study, which is a relatively brief period to assess the state of such a productive and dynamic ecosystem. Thus the relationships that we presented may not be robust, and it is likely that some of the significant relationships are non-significant and vice versa. The results indicated that the morphology and growth rates of New Zealand fur seal pups on Kangaroo Is, the breeding success of little penguins at Troubridge Is, the morphology of crested tern fledglings from Troubridge Is and the growth of short-tailed shearwater chicks on Althorpe Is were negatively correlated with the annual catch of the SASF. However, sardines are a very small part of the diet of New Zealand fur seals, little penguins and short-tailed shearwaters and these predators do not overlap significantly with the area that is typically used by the fishery (chapter 4).

Like previous chapters, the results of this chapter indicated that the species most likely to be impacted by reductions in the availability of sardines is crested terns.

The most appropriate form of EBFM for the SA sardine fishery would include monitoring of the morphology and survival of crested terns together with periodic updates of the EPIs that are derived from the trophodynamic model. Efforts should be made to monitor the EPIs for crested terns at colonies in Spencer Gulf (e.g. Donington Rock) and at Troubridge Is. To provide a broader perspective of the potential impact of the SA sardine fishery in the ecosystem, the trophodynamic model should be updated every five years, to incorporate all of the available data (based on the biology of predators, catches of fisheries and environmental features) and improvements to the design of the model.

Relationships between the breeding and feeding ecology of predators and catch and spawning biomass of a fishery are commonly reported in the literature, but cases where such findings are used to improve the management of a fishery are rare. Validation of the relationships between EPIs based on crested terns versus sardine availability will ensure that monitoring is focussed on data that can be collected efficiently and that can be used by fishery managers to assess the effects of their management strategies. This approach will ensure that the SA sardine fishery remains a world leader in the implementation of EBFM

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12 APPENDICES

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Appendix 2: Intellectual property and valuable information

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Appendix 3: Functional groups (by common names) developed in the EGAB model

Functional Group	Group	Taxa (Family in parentheses)
Baleen whales	1	*Balaenoptera musculus brevicauda*
Bottlenose dolphin	2	*Tursiops aduncus, T. truncates*
Common dolphin	3	*Delphinus delphis*
NZ fur seal	4	*Arctocephalus forsteri*
Aust fur seal	5	*Arctocephalus pusillus doriferus*
Aust sea lion	6	*Neophoca cinerea*
Little penguin	7	*Eudyptula minor*
Petrels	8	*Puffinus tenuirostris, P. carneipes, Pelagodroma marina, Thalassarche melanophrys, T. carteri, T. cauta*
Gannets	9	*Morus serrator*
Terns	10	*Sterna bergii*
Pelagic sharks	11	*Carcharodon carcharias, Isurus oxyrinchus, Prionace glauca, Sphyrna zygaena, Alopias vulpinus, Carcharhinus brachyurus, C. obscurus*
Demersal sharks	12	*Asymbolus vincenti, Centroscymnus spp.,Cephaloscyllium laticeps, Callorhinchus milii, Centrophorus moluccensis, Dalatias licha, Deania spp., Foetorepus calauropomus, Furgaleus macki, Galeorhinus galeus, Hydrolagus ogilbyi, Hypogaleus hyugaensis, Heterodontus portusjacksoni, , Isistius brasiliensis, Mustelus antarcticus, Notorynchus cepedianus, Orectolobus maculates, Parascyllium ferrugineum, Pristiophorus spp., Squalus spp., Sutorectus tentaculatus* (Brachaeluridae, Centrophoridae, Chimaeridae, Dalatiidae, Etmopteridae, Hexanchidae, Pristiophoridae, Squalidae, Somniosidae, Squatinidae, Scyliorhinidae, Triakidae)
Rays and skates	13	*Aptychotrema vincentiana, Dasyatis brevicaudata, Dasyatis thetidis, Dipturus whitleyi, Hypnos monopterygium, Manta birostris, Myliobatis australis, Trygonorrhina fasciata, Urolophus spp.,* (Dasyatidae, Rajidae, Rhinidae, Rhinobatidae)
SBT	14	*Thunnus maccoyii*
Other tunas-kingfish	15	*Thunnus albacores, T. alalunga, T. obesus, Katsuwonus pelamis, Xiphias gladius, Lampris guttatus, Lepidocybium flavobrunneum, Seriola lalandi*
Large bentho-pelagic piscs	16	*Achoerodus viridis, Achoerodus viridis, Argyrosomus japonicas, Asymbolus spp, Cheilodactylus nigripes, Caranx lugubris, Centroberyx spp., Chrysophrys auratus, Cyttus traverse, Dactylophora nigricans, Dannevigia tusca, Diodon nicthemerus, Enoplosus armatus, Epinephelus quoyanus, Etelis carbunculus, Genypterus blacodes, Genypterus spp., Gonorynchus greyi, Hyperoglyphe antarctica, Ichthyscopus barbatus, Kathetostoma leave, Latridopsis spp., Latris lineate, Leviprora inops, Lutjanus erythropterus, Macruronus novaezelandiae, Mora moro, Nelusetta ayraudi, Nemadactylus spp., Neoplatycephalus spp., Neosebastes bougainvillii, Omegophora armilla, Optivus agrammus, Pagrus auratus, Paratrachichthys macleayi, Plectorhinchus spp., Platycephalus spp., Polyprion spp., Pristipomoides filamentosus, Pseudocaranx spp., Pseudocyttus maculates, Pseudophycis bachus, Pseudocaranx wrighti, Rexea solandri, Ruvettus pretiosus, Schedophilus labyrinthicus, Scorpis aequipinnis, Seriolella brama, Siphonognathus radiates, Sphyraena spp.,Thyrsites atun, Thysanophrys cirronasa, Trachichthys australis, Urolophus gigas* (Bathygadidae, Centrolophidae, Colocongridae, Congridae, Cyttidae, Macrouridae, Neosebastidae, Percichthyidae, Serranidae, Trachichthyidae, Uranoscopidae, Zeidae)
Blue mackerel	17	*Scomber australasicus, Scomberesox saurus*
Jack mackerel	18	*Trachurus declivis, Trachurus novaezelandiae*
Redbait	19	*Emmelichthys nitidus, Etrumeus teres*
Anchovy	20	*Engraulis australis*
Sardine	21	*Sardinops sagax*
Inshore small planktivores	22	(Atherinidae, Clupeidae), *Iso rhothophilus*
Salmons & ruffs	23	*Arripis georgianus, Arripis truttacea*

Appendix 3 (cont.)

Functional Group	Group	Taxa (Family in parentheses)
Medium demersal piscs	24	*Acanthaluteres spilomelanurus, Acanthaluteres spilomelanurus, Acanthaluteres vittiger, Aethalopercaspp., Aldrichetta forsteri, Allocyttus spp., Allocyttus verrucosus, Anyperodon spp., Aracana spp., Aulopus cf purpurissatus, Austrolabrus maculates, Beryx splendens, Brachaluteres jacksonianus, Caesioperca Lepidoptera, Cantheschenia longipinnis, Centroberyx spp., Cheilodactylus spp., Cnidoglanis macrocephalus, Contusus brevicaudus, Cyttopsis rosea, Cyttus australis, Dinolestes lewini, Eubalichthys spp., Glaucosoma spp., Helicolenus percoide, Hoplostethus atlanticus , Latris lineate, Lepidotrigla spp., Lipocheilus carnolabrum, Lotella spp., Lutjanus spp., Metavelifer multiradiatus, Meuschenia spp., Neocyttus rhomboidalis, Neocyttus spp , Neoodax balteatus, Neoplatycephalus aurimaculatus, Neosebastes pandus, Notolabrus tetricus, Oplegnathus woodwardi, Othos dentex, Parapercis ramsayi, Parazanclistius hutchinsi, Paristiopterus gallipavo, Pegasus lancifer, Pictilabrus laticlavius, Plagiogeneion spp., Platycephalus spp., Pleuronectidae spp., Pseudopentaceros richardsoni, Pseudophycis barbata, Pterygotrigla polyommata, Satyrichthys cf moluccense, Scobinichthys granulates, Seriolella punctata, Sillaginodes puncata, Siphonognathus spp., Spratelloides robustus, Tetractenos glaber, Trachyscorpia carnomagula, Zanclistius elevates, Zenopsis nebulosus, Zeus faber* (Balistidae, Bothidae, Diodontidae, Labridae, Monocanthida, Odacidae, Pentacerotidae, Tetraodontidae)
Small demersal piscs	25	*Caesioperca lepidoptera, Caesioperca rasor, Callanthias australis, Chelmonops curiosus, Cristiceps australis, Echiodon rendahli, Gymnapistes marmoratus, Lepidoperca pulchella, Odax acroptilus, Oreosoma atlanticum, Parapercis haackei, Parapriacanthus elongatus, Parazanclistius hutchinsi, Pempheris spp., Pentaceropsis recurvirostris, Pentaceros decacanthus, Sillago flindersi, Thamnaconus degeni, Upeneichthys lineatus, Vincentia spp.,* (Gobiidae)
Medium demersal invert feeders	26	*Ammotretis lituratus, Cheilodactylus nigripes, Cynoglossus broadhursti, Girella tricuspidata, Kanekonia queenslandica, Lophonectes gallus, Pseudorhombus jenynsii, Sillago spp., Taratretis derwentensis, Zebrias scalaris,*
Small demersal invert feeders	27	*Glyptauchen panduratus, Histiophryne cryptacanthus, Neopataecus waterhousii, Parequula melbournensis, Phyllophryne scortea, Repomucenus calcaratus, Rhycherus filamentosus, Scorpis lineolata, Siphaemia cephalotes, Sorosichthys ananassa, Tilodon sexfasciatus*
Mesopelagics	28	*Diaphus jenseni, Electrona carlsbergi, Electrona paucirastra, Gymnoscopelus robustus, Krefftichthys anderssoni, Myctophidae, Symbolophorus spp.*
Small demersal omnivore	29	*Filicampus tigris, Hippocampus abdominalis, Histiogamphelus cristatus, Hyporhamphus melanochir, Leptoichthys fistularius, Maxillicosta scabriceps, Paraulopus nigripinnis, Pelates octolineatus, Phycodurus eques, Phyllopteryx taeniolatus, Polyspina piosae, Siphonognathus argyrophanes, Stigmatopora argus, Upeneichthys spp., Upeneichthys vlamingii,* (Mullidae, Scorpaenidae, Sygnathidae,Tetraodontidae)
Arrow squid	30	*Nototodarus gouldi, Todarodes filippovae* (Ommastrephidae, Teuthoidea)
Calamary	31	*Sepioteuthis australis*
Other squids	32	*Sepia spp., Histioteuthis spp., Euprymna tasmanica,* (Sepiidae)
Octopus	33	*Octopus maorum, Argonauta nodosa, Octopus berrima,* (Octopodidae)
Large zooplankton (carnivores)	34	Amphipoda, Maxillopoda, Isopoda, Salpidae
Small zooplankton (herbivores)	35	*Nyctiphanes australis,* Ostracods
Benthic grazer (megabenthos)	36	*Brachyura spp., Jasus edwardsii, Melicertus latisulcatus, Metapenaeopsis palmensis, Pseudocarcinus gigas,* (Crustacea, Decapoda, Pandalidae, Scyllaridae– undifferentiated)
Detritivores	37	(Nereididae), Polychaeta
Filter feeders	38	*Plebidonax deltoids,* non cephalopod Mollusca, Polychaeta
Primary production	39	Phytoplankton, Macroalgae, Algae
Detritus	40	Detritus

Appendix 4: Summary of the biology of the functional groups used in the EGAB model

Appendix 4.1 Baleen whales (model group 1)

The most significant species of baleen whale that forages within the EGAB region is the pygmy blue whale (*Balaenoptera musculus brevicauda*). Southern Right whales (*Eubalaena australis*) breed in South Australia and humpback whales (*Megaptera novaeangliae*) are also common visitors, but neither is thought to forage in shelf waters of the EGAB. Pygmy blue whales are seasonal visitors to shelf and slope waters of the GAB, with major feeding aggregations associated with upwelling occurring between Cape Otway (Victoria) and south and west of the lower Eyre Peninsula (South Australia) between November and May (Gill 2009). The numbers of pygmy blue whales that forage in GAB waters is unknown, but based on estimates of sightability from aerial surveys; they may number 150 and are thought to remain in the upwelling system for approximately 6 months (P. Gill pers. comm.). We have assumed about half of these (75) are distributed in the EGAB ecosystem. Estimates of the mass of pygmy blue whales range between 60 and 150 tonnes. We have used a conservative estimate of 80 t per whale to account for subadults (P. Gill pers. comm.). This gives a biomass estimate of 6,000 t within the EGAB ecosystem.

Prey consumption was estimated using the methods presented by Barlow *et al.* (2008) for cetaceans feeding in the California Current ecosystem. They used models of the average daily ration (R in kg wet wt) and average daily metabolic requirements ($ADMR$ in kJ d^{-1}) as follows:

$$R = ADMR / \{0.8[3900\mathrm{Z} + 5450(1 - \mathrm{Z})]\},$$

where:

$$ADMR = \beta\left(293.1M^{0.75}\right),$$

and 3900 and 5450 are the energy densities of crustaceans and fish, respectively (kJkg^{-1} wet weight), Z is the fraction of crustaceans in the diet, 0.8 is the assimilation efficiency (Leaper & Lavigne 2007) and β = 2.5 (Kenney *et al.* 1997, Hooker *et al.* 2002, Laidre *et al.* 2004).

These models were based on the Kleiber (1975) function for basal metabolic rate (*BMR*) related to the mass (*M*) of homeotherms:

$$BMR = 293.1M^{0.75},$$

and food consumptions models developed by Lavigne (1996) and Leaper and Levigne (2007). Total annual prey consumption was estimated as the product of the mean daily ration (365 x *R*) and the pygmy blue whale abundance (Barlow *et al.* 2008). Following this, we estimated the annual prey consumption of a pygmy blue whale weighing 80 t to be 408 t, and the total annual consumption of 75 whales to be 30,582 t. This provides a Q/B estimate of 5.097.

We have assumed that half (50%) of pygmy blue whale annual intake occurs in the summer/autumn feeding areas of the EGAB ecosystem. As such dietary import is estimated to be 0.50. The pygmy blue whale habitat area is estimated to represent 0.458 of the EGAB ecosystem. As such the pygmy blue whale biomass in the habitat area is estimated to be 0.085 t.km^{-1}.

Trites *et al.* (1999) estimated the P/B ratio of whales to be half the maximum population growth rate (r_{max}), which has been estimated at 4% (Reily & Barlow 1986). Hence we used a P/B of 0.02.

Appendix 4.2 Toothed whales/dolphins (model groups 2 &3)

For continental shelf waters of the EGAB, dolphins are the most common toothed cetacean species. Bottlenose (*Tursiops aduncus* and *T. truncatus*) and common dolphins (*Delphinus delphis*) occur in South Australia (Kemper & Ling 1991, Kemper *et al.* 2006). Dolphin abundance was based on the mean density of dolphins (0.0813/km^2) from aerial surveys conducted in part of the EGAB by Kemper *et al.* (2006). The densities of individual species has not been determined from aerial surveys, so we assumed that the relative abundance was the same as the relative proportions that are recovered from beaches in the same region: 0.4 are *Tursiops* spp. and 0.6 *D. delphis* (Kemper and Gibbs 2001). The mass of each species was estimated to be 188 kg and 80 kg (Barlow *et al.* 2008), giving overall estimates of biomass of 0.00611 and 0.00390 t.km^{-1}, respectively. Estimates of Q/B followed the same approach detailed for baleen whales (in A.1). P/B was estimated at 0.08 and 0.09 for *Tursiops* and *Delphinus* based on Barlow and Boveng (1991).

Appendix 4.3 Seals (model groups 4-6)

Australian sea lion

Australian sea lions (*Neophoca cinerea*) are endemic to Australia and restricted to South and Western Australia. Most of the population occurs within the EGAB ecosystem between Nuyts Reef and The Pages. The most reliable pup production estimates available are those summarised in Goldsworthy and Page (2007), and were assumed to be representative of pup production in 1990, with the exception of Dangerous Reef, The Pages and Seal Bay, where 1990 pup production estimates were based on Gales *et al.* (1994) and McIntosh *et al.* (2006). Total pup production within the EGAB ecosystem in 1990 was hence estimated to be 2,212. Age-specific survival and pup production data were used to estimate the numbers of animals alive at each age stage. Life tables were based on those developed by McIntosh (2007) and modified to achieve stable growth by Goldsworthy *et al.* (2010). A maximum longevity of 24 and 21.5 years for females and males was used (McIntosh 2007). As ASL breed about every 17.5 months (Shaughnessy *et al.* 2006), survival was calculated for every 1.458 years. Age-mass relationships for females and males followed those developed for the species by McIntosh (2007) and were used to estimate population biomass (*B*).

New Zealand fur seal

Estimates for the abundance of New Zealand fur seals (*Arctocephalus forsteri*) in the EGAB region were based on pup production estimates (5,636) obtained for the 1989/90 breeding season (Shaughnessy *et al.* 1994). Pup production at most Kangaroo Island breeding sites has been monitored annually since 1988, and other South Australian colonies less frequently (Shaughnessy *et al.* 1994, Shaughnessy *et al.* 1995, Shaughnessy & Dennis 1999, Shaughnessy & Dennis 2001, Shaughnessy & McKeown 2002, Shaughnessy & Dennis 2003, Shaughnessy 2004, 2005, Shaughnessy *et al.* 2005, Shaughnessy & Goldsworthy 2007). These data were used to provide estimates of annual changes in pup production since 1990. Life-tables were based on those developed by Goldsworthy *et al.* (2003) and Goldsworthy and Page (2007), utilising data available for closely related species. Age-specific survival relationships were: females $S = 0.627 - 0.073a + 0.003a^2 - (5.91 \times 10^{-5})a^3$; males $S = 0.627 - 0.097a + 0.006a^2 - (0.140 \times 10^{-3})a^3$), where S is survival and a is age in years. Maximum ages were 23.4 and 16.7 for females and males, respectively (McKenzie 2006, McKenzie *et al.* 2007). Age-mass relationships for females and males followed those developed for the species by McKenzie *et al.* (2007), and were used to estimate population biomass (*B*).

Australian fur seal

Australian fur seals (*Arctocephalus pusillus doriferus*) principally breed on the Bass Strait islands off Victoria and Tasmania and have been recorded in South Australia mostly as non-breeding visitors but have recently establish a breeding colony at North Casuarina Island (Kirkwood *et al.* 2005, Shaughnessy *et al.* 2010). Estimates of the abundance of Australian fur seals (*Arctocephalus pusillus doriferus*) in the EGAB region in 1990 were based on maximum counts of animals hauled out at Cape Gantheaume, North Casuarina Island and Cape du Couedic (29, 98 and 3, respectively) between 1988 and 1990 (Shaughnessy *et al.* 2010), with seals observed ashore representing about 25.8% of all seals onshore and at sea at any given time (Kirkwood *et al.* 2006). Annual rates of increase of 7.4% and 12.0% at Cape Gantheaume and North Casuarina Island of the mostly 3+ year-old males was assumed (Shaughnessy *et al.* 2010). Based on the mean annual change in pup production from 11 to 29 pups at North Casuarina Island between 2006/07 and 2007/08, respectively, we assumed that breeding commenced in 2001/02 (1 pup), increasing at 61.1% per season between 2001/02 and 2006/07, and at 12.0% per year since 2007/08. This rate approximates those reported for establishing NZFS populations at Cape Gantheaume (Shaughnessy *et al.* 1995). Life-tables were based on those developed by Goldsworthy *et al.* (2003). Maximum ages were 21 and 19 years for females and males, respectively (Warneke 1995). Age-mass relationships for females and males followed those developed for the species by Arnould and Warneke (2002), and were used to estimate population biomass (*B*).

Prey consumption – all species

A mass-based regression equation of field metabolic rate (FMR) based on seven otariid species developed by BF Green, presented in Goldsworthy *et al.* (2003), was used to estimate daily energy requirement (*ER*):

$$ER_{at-sea} = 2.234M^{0.665},$$

where $ER_{at\text{-}sea}$ is MJd^{-1} and *M* is the mean mass of each age-class/sex. The average daily energy requirement of otariid seals is a function of the proportion of time spent at sea and on-shore (Costa & Gales 2000, Winship *et al.* 2002), with daily energy requirements at-sea being about 1.8 times greater than those on-shore ($ER_{on\text{-}shore}$)

(Costa & Gentry 1986). As such the *ER* of each age-class/sex was estimated following Mecenero *et al.* (2006) as:

$$ER = \left(ER_{at-sea} p_{on-shore} + ER_{on-shore} p_{on-shore}\right) / 0.93,$$

Where the proportion of time spent at sea and on-shore is $p_{at\text{-}sea}$, $p_{on\text{-}shore}$, respectively. Estimates of $p_{at\text{-}sea}$, $p_{on\text{-}shore}$, were based on those in Goldsworthy *et al.* (2007), Goldsworthy and Page (2007) and Kirkwood *et al.* (2006). 0.93 is the estimated mean prey assimilation efficiency (Winship *et al.* 2002, Mecenero *et al.* 2006). An average prey energy density of 4.985 MJ/kg (Goldsworthy *et al.* 2003) was then used to estimate the total annual prey consumption (Q ty^{-1}) of age/sex classes as:

$$Q = \left[\left(\mathrm{ER}/4.985\right)365\right]/1000.$$

Estimates of *Q* and *B* per species were used to estimate *Q/B* values. Production (*P*) per Biomass estimates (*P/B*) were estimated as: (current biomass live + dead)/(previous year annual biomass alive)).

Appendix 4.4 Seabirds (model groups 7-10)

Little penguin

Estimated abundance of little penguins (*Eudyptula minor*) in the EGAB region is between 20,000 and 45,000 breeding pairs (Copley 1996). Survival in little penguins is estimated to be 17%, 71% and 78% in each of the first three years, respectively, and 83% per year subsequently (P. Dann pers. comm.). 50% of birds are mature and breed when they are two years of age, with the remaining birds breeding for the first time at three years (Dann & Cullen 1990). A simplified life-table based on these parameters and maximum longevity of ~26 years (Dann *et al.* 2005) suggests juveniles make up 27% of the population, while breeding pairs (adults) make up 73%. Using the median estimate of breeding pairs (32,500), the total population of little penguins in the EGAB is estimated to be 89,633. Assuming a mean mass of 1.2 kg per bird, the total biomass of the population is estimated to be 107.6 t. Non-breeding (juvenile) little penguins were estimated to consume 73.1 kg per year, based on prey consumption of 167 g/kg.d (Costa *et al.* 1986), while breeding little penguins are estimated to consume 114.0 kg of prey each year (including the food requirements for 0.85 chicks/year, 1.7 per pair) (Bethge *et al.* 1997). This provides an estimate of total annual prey consumption (*Q*) in the EGAB of 9,211.8 t, and a *Q/B* of 85.6. A *P/B* estimate of 1.29 was derived from an estimate for Antarctic penguins (Cornejo-Donoso & Antezana 2008). Diet data was based on that detailed for the species in Chapter 3.

Petrels

The dominant petrel species in the EGAB region are the short-tailed shearwater (*Puffinus tenuirostris*) (1.046 – 1.146 million breeding pairs) and the white-faced storm petrel (*Pelagodroma marina*) (241,000 – 251,000 breeding pairs) (Copley 1996). Small numbers (~150 pairs) of flesh-footed shearwaters (*Puffinus carneipes*) also breed in the region (Copley 1996). Common non-breeding species include the black-browed albatross (*Thalassarche melanophrys*) and Indian yellow-nosed albatross (*Thalassarche carteri*) which are present over shelf waters in most months but most frequently in May – October, and the shy albatross (*Thalassarche cauta*) which is most common in April - October (Copley 1996).

Of these species, data on the breeding ecology, diet and at-sea distributions within the EGAB region are only available for the short-tailed shearwater. They undergo major migrations, overwintering in the North Pacific Ocean and Bering Sea, arriving in south-eastern Australia in September/October and leaving again in March/April (Weimerskirch & Cherel 1998). The return rate of fledged chicks at four years of age is estimated at 0.437 and adult annual survival at 0.92 (Wooller *et al.* 1990, Hunter *et al.* 2000). With the mean age of first breeding at ~ 7 years (Hunter *et al.* 2000), a simplified life-table based on these parameters suggests juveniles make up 47% of the population, while breeding pairs (adults) make up 53%. Using the median estimate of the number of breeding pairs (1.095 million, Copley 1996), the total population of short-tailed shearwater within the EGAB region is estimated to be 4.13 million. Assuming a mean mass of 0.7 kg per bird, the total biomass (*B*) of the population is estimated to be about 2,900 tonnes.

The active (965.9 kJ/d) and resting (296.9 kJ/d) metabolic rates for short-tailed shearwaters were estimated from regression equations in Warham (1996). Breeding pairs were assumed to spend 206 days in non-breeding foraging grounds, 14 days pre-incubation in the EGAB and adjacent waters, 55 days incubating the egg (incubation shared equally between the sexes) and 90 days rearing chicks (Weimerskirch & Cherel 1998, Einoder & Goldsworthy 2005, Einoder 2010). In South Australia, short-tailed shearwaters undertake on average 28 short foraging trips over shelf waters and 12 long trips into the Southern Ocean during the 90 day chick-rearing period (Einoder 2010). Assuming individual birds spend about 5 hours ashore in between foraging trips; birds were estimated to spend 10.2% of their time ashore and 89.8% at sea. The prey consumption equation of Daunt *et al.* (2008) was used, assuming an assimilation efficiency of 0.69, and based on information of dietary breakdown, prey energy density and 4.5 kg of prey being fed to the chick by each breeding pair (Einoder 2010). Annual prey consumption (*Q*) was estimated at 438,165 t, but with 70% of foraging time during chick rearing spent on long trips into the Southern Ocean, and 206 days spent undertaking the annual migration into the Northern Hemisphere, most (86.9%) prey consumption is estimated to be imported (derived from outside the EGAB ecosystem). *Q/B* is estimated to be 150.4. A *P/B* estimate of 1.0 was derived from an estimate for Antarctic seabirds (Cornejo-Donoso & Antezana 2008). Diet data was based on that detailed for the species in Chapter 3.

The median estimate of breeding pairs of White-faced storm petrels within the EGAB region is 246,000 (Copley 1996). Assuming breeding pairs make up 2/3^{rds} of the population, the total median estimate of the population is 745,455. White-faced storm petrels are estimated to be present within the EGAB between October and March which includes a 45 day incubation and 51 day chick rearing period (Marchant & Higgins 1990). Assuming a mean mass of 55 g (Marchant & Higgins 1990), adults spending 82% of their time at sea, and at-sea and onshore metabolic rates of 223.7 kJ.d^{-1} and 50.3 kJ.d^{-1}, respectively (estimated from equations in Warham 1996), an assimilation efficiency of 0.69, a prey energy density of 5 MJ/kg, and a mean meal mass fed to chicks of 6.4 g (0.5 meals per night) (Marchant & Higgins 1990); prey consumption per annum is estimated to be 26,244 t (using equations in Daunt *et al.* 2008). Import of prey consumption from outside the EGAB ecosystem was estimated to be 41.9%. Total population biomass is estimated to be 41 t, giving a *Q/B* estimated of 640.1. A *P/B* estimate of 1.0 was derived from an estimate for Antarctic seabirds (Cornejo-Donoso & Antezana 2008). Diet data was based on that detailed for the species by Imber (1981).

The global population sizes of black-browed albatross (1,220,000), Indian yellow-nosed albatross (160,000) and shy albatross (55,000) (population estimates sourced from IUCN 2010). We assumed that approximately 0.5%, 1% and 10% of the global populations of each species occurs within EGAB throughout the year, representing a total of approximately 13,200 individual birds, with biomass of 48 t and consumption of 3,337 t (assuming a mean individual mass of 3.4 kg, FMR of 2390 kJd^{-1}, assimilation efficiency of 0.69 and mean prey energy density of 5 MJkg^{-1}). Diet data was based on that detailed for the shy albatross in Chapter 3.

Parameters for the *Ecopath* model for the petrel group were combined to provide an overall estimate of biomass (3,197 t), biomass in the habitat area (0.00306 t.km^{-2}) and consumption (470,354 t). Based on these values, P/B was estimated to be 147.1. A P/B estimate of 1 was used based on Sakshaug (1997). Dietary data was weighted for each species group based on their proportion to prey biomass consumed in the habitat area.

Australasian gannet

The only breeding colony of Australasian gannets (*Morus serrator*) in the EGAB ecosystem is at Margaret Brock Reef off Cape Jaffa where approximate 300 breeding pairs nest (Lighthouses of Australia Inc 2004). Most gannets in the EGAB ecosystem would be birds originating from this and from breeding colonies in Victoria and Tasmania, which number approximately 6,660 pairs (Marchant & Higgins 1990). We estimated that all of the Margaret Brock Reef population and about 10% of the Victoria and Tasmania populations forage within the EGAB region at any time. With individual gannets weighing approximately 2.5 kg (Daunt *et al.* 2008), EGAB gannet biomass is 4.8t (0.0000308 tkm^{-2} in habitat area). Estimates of the energy needs of breeding and non-breeding birds (4,561 KJd^{-1}), plus the energy costs of egg (201,100 KJ) and chick production (145,000 KJ) were derived from Bunce (2001). Assuming 0.63 chicks per pair, 0.75 assimilation efficiency and a mean prey energy density of 6.7 kJg^{-1} (Bunce 2001), prey consumption was estimated using the formula of Daunt *et al.* (2008) to be 657.5 t. Based on these estimates, Q/B is 138.3. P/B estimate of 1 was used based on Sakshaug (1997). Dietary data was based on (Bunce 2001).

Crested tern

The total population of crested terns (*Sterna bergii*) in the EGAB (57,566) was estimated using the median number of breeding pairs (19,000, Copley 1996) and assuming that adults make up 2/3^{rds} of the total population. Total biomass is estimated to be 19.6 t (0.0000127 t.km^{-2} in habitat area) based on an individual mass of 0.34 kg (Mcleay 2010). From estimates of daily energy needs of adults and chicks (406.3 kJd^{-1}), breeding pairs each raising one chick over a 40 day period, an assimilation efficiency of 0.75 and mean prey density of 6.7 kJg^{-1} (Chiaradia *et al.* 2002, Daunt *et al.* 2008), total prey consumption was estimated at 1,761 tyr^{-1}. Based on these estimates, Q/B is 89.9. P/B estimate of 1 was used based on Sakshaug (1997). Dietary data was based on studies undertaken in South Australia by McLeay *et al.* (2009a), summarised in Chapter 3.

Appendix 4.5 Pelagic sharks (model group 11)

Pelagic shark species considered in the EGAB model included the white shark (*Carcharodon carcharias*), shortfin mako (*Isurus oxyrinchus*), blue shark (*Prionace glauca*), smooth hammer head shark (*Sphyrna zygaena*), common thresher shark

(*Alopias vulpinus*), bronze whalers (*Carcharhinus brachyurus*) and dusky whalers (*C. obscurus*). There was scant incidental catch data for pelagic sharks, and records were likely to have under-estimated actual landings. Pelagic shark species included in the model, for which there were some time series catch data, included shortfin mako, blue, smooth hammerhead and whaler sharks, mainly in the SA line and net marine scalefish fishery, and the demersal gillnet shark fishery. There was limited discard information available. No biomass (*B*) data was available for any component of this model group and this parameter was estimated by the model. *P/B* and *Q/B* estimates of 0.2 and 1.2, respectively were used (Last & Stevens 1994, Bulman *et al.* 2006, Froese & Pauly 2009). Diet data were sourced from SARDI unpublished datasets, Rogers and Huveneers (2009), Huveneers and Rogers (unpublished data), and Page *et al.* (Chapter 3).

Appendix 4.6 Demersal sharks (model group 12)

Demersal sharks are among the most heavily exploited of the large marine fauna that inhabits southern Australian shelf waters. Time series data from logbooks between 1991 and 2008 for annual catch and effort were included in the model for this broad group comprising 33 taxa. It included members of the Family Chimaeridae (Chimeras). Commonwealth fisheries that have significantly harvested this model group include the Gillnet Hook and Trap (GHAT) and Southern Shark Fishery. The main gear types were demersal gillnet, long-line and drop-line. The State managed (< 3 nm from shore) component of the catch was taken in the Marine Scalefish (MSF) fishery using long-lines and hand-lines. Dominant species included in the model were gummy shark (*Mustelus antarcticus)*, school shark (*Galeorhinus galeus)*, *Chimaeridae*, whiskery shark (*Furgaleus macki)* and broadnose shark (*Notorynchus cepedianus).* There was limited discard information for State or Commonwealth fisheries, with the exception of the period between 2006 and 2008 in the shark gillnet component of GHAT fishery and for 2007 in the MSF fishery (Fowler *et al.* 2009). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, based on the upper 95% CL from 12 demersal shark species (0.307 $t.km^{-1}$) (Currie *et al.* 2009, Currie & Sorokin 2010). *P/B* and *Q/B* estimates of 0.18 and 1.8 were sourced from the literature (Walker 1984, Moulton *et al.* 1992, Last & Stevens 1994, Bulman *et al.* 2006, Froese & Pauly 2009) and estimated by the model. Diet data were sourced from Bulman *et al.* (2006). For *Notorynchus cepedianus,* diet data was sourced from the review by Huveneers and Rogers (unpublished data).

Appendix 4.7 Rays and Skates (model group 13)

Catch and effort time series data from logbooks were available between 1984 and 2008 for this broad taxonomic group comprising seven coarsely identified skates and rays. The main gear types that take these taxa as bycatch were demersal gillnet, bottom trawl, and long-line. The State managed (< 3nm from shore) component was mostly taken in the MSF fishery using long-lines. Most skates and rays tend to be discarded with the exception of the southern eagle ray (*Myliobatis australis*) which is occasionally retained. Catch data for this group is patchy and undoubtedly biased by the fact that most large Dasyatidae are released. There was limited discard information available for State or Commonwealth fisheries that take this model group as bycatch, with the exception of the period between 2006 and 2008 for the Commonwealth GHAT fishery, and in 2007 when a dedicated bycatch program was implemented in State waters (Fowler *et al.* 2009). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, using estimates for the upper 95% CL for 10 rays and skate species (0.459 t.km^{-2}) (Currie *et al.* 2009, Currie & Sorokin 2010). *P/B* and *Q/B* estimates of 0.35 and 2.7, respectively were sourced from Froese & Pauly (2009). Diet data for were sourced from Bulman *et al.* (2006) and Currie and Sorokin (2010).

Appendix 4.8 Southern Bluefin Tuna (SBT) (model group 14)

Southern bluefin tuna (SBT), *Thunnus maccoyii* is a highly migratory and pelagic species that occurs between 30°S and 50°S, and nearly to 60°S. Juveniles aggregate in the GAB during each summer and autumn (Young *et al.* 1996, Gunn & Young 1999). Most (99.6%) of the Australian SBT catch is taken in the EGAB region (Wilson *et al.* 2009). The spawning component of this species is considered to have suffered from serious depletion and is currently classified as Critically Endangered (IUCN 2010). Time series of catch and effort data for the period 1991 to 2008 were accessed from Australian Fishery Management Authority (AFMA) databases managed by CSIRO. The main gear types used to catch SBT in the study area have been pole, pelagic long-line and purse-seine. There was limited discard information available for SBT. No biomass data was available, and this parameter was estimated by the model. *P/B* and *Q/B* estimates were 0.2 and 1.6 respectively (Bulman *et al.* 2006, Froese & Pauly 2009). Diet data were sourced from Caines (2005), Ward *et al.*(2006) and Page *et al.* (Chapter 3).

Appendix 4.9 Other tunas and kingfish (model group 15)

The other tunas and kingfish group included species such as yellowfin tuna (*Thunnus albacores*), albacore (*T. alalunga)*, bigeye tuna (*T. obesus*), Australian bonito (*Katsuwonus pelamis*), swordfish (*Xiphias gladius*), opah (*Lampris guttatus*), escolar (*Lepidocybium flavobrunneum*) and yellowtail kingfish (*Seriola lalandi*). The main gear types utilised in these fisheries were pole, long-line and purse-seine. No biomass data was available, and this parameter was estimated by the model. *P/B* and *Q/B* estimates were 0.2 and 1.2 respectively (Bulman *et al.* 2006, Froese & Pauly 2009). Diet data were sourced from Caines (2005), Ward *et al.* (2006) and Page *et al.* (Chapter 3).

Appendix 4.10 Large bentho-pelagic piscivorous fish (model group 16)

The large bentho-pelagic piscivorous fish group describes a range of species that inhabit both pelagic shelf and gulf habitats, and benthic habitats ranging from sand, seagrass and low limestone reefs in the central gulfs, to calcareous limestone and high relief granitic reef in mid-outer shelf waters. Catch and effort time series data from logbooks were available for this broad taxonomic group between 1991 and 2008. The main demersal gear types used to take these taxa were bottom trawl, long-line, hand-line and shark gill-net. The State managed (< 3nm from shore) component of the catch was predominantly taken in the MSF fishery using handlines and long-lines. Typical species in this group include deepwater flathead (*Platycephalus conatus*) and blue morwong (*Nemadactylus valenciennesi*) in shelf waters, and snapper (*Pagrus auratus*) and snook (*Sphyraena novaehollandiae*) in gulf waters. Biomass estimates were based on the upper 95% CL of estimates for 26 species in this functional group in Spencer Gulf (0.452 $t.km^{-2}$) (Currie & Sorokin 2010). *P/B* and *Q/B* estimates were 0.338 and 3.315, respectively (Bulman *et al.* 2006, Froese & Pauly 2009). Diet data were obtained from Currie and Sorokin (2010).

Appendix 4.11 Small pelagic fish (groups 17-22)

The South Australian Sardine Fishery is the biggest Australian fishery by total weight of the catch. The fishery predominantly takes Sardine (*Sardinops sagax*), but other small pelagics are also captured including anchovy (*Engraulis australis*), jack and yellow tail mackerel (*Trachurus* spp.), maray (*Etrumeus teres*), blue mackerel (*Scomber australasicus*), redbait (*Emmelichthys nitidus*) and blue sprat (*Spratelloides *robustus*). The catch is taken at night using purse seine nets. Spawning stock biomass is estimated by SARDI using the Daily Egg Production Method (DEPM) on an annual or bi-annual basis (Ward *et al.* 2009a). Estimates of spawning stock biomass typically range between 146,000 – 264,000 t (Ward *et al.* 2009b). Recent DEPM based estimates of anchovy biomass in SA gulf and shelf waters were 127,000 t, and a biomass of 1.59 t.km^{-2} (upper 95%CL) was used in the model based on the estimate by Dimmlich *et al.* (2009). Sardine biomass datasets collected by the Small Pelagic Fishes SubProgram of SARDI were available between 1995 and 2007. A biomass estimated for 1991 was based on the mean for years before the mass mortality (<1996), post-recovery years (2003 onward) plus 1998 (Ward *et al.* 2001 a,b, Ward *et al.* 2009b). Blue mackerel biomass estimates were based on DEPM estimates of spawning biomass 56,228 t within the EGAB region (Ward *et al.* 2007). Biomass of jack mackerel, redbait and inshore small planktivores was estimated by the model. Time series for catch and effort of sardine was available for 1991 to 2008. Fishing mortality (F) was estimated as the catch divided by spawning biomass. *P/B* and *Q/B* estimates of 1.6 and 5.04 for sardine, 0.70 and 5.040 for anchovy, 0.74 and 2.80 for redbait, 0.74 and 3.30 for jack mackerel, and 0.37 and 3.50 for blue mackerel, respectively were sourced from (Bulman *et al.* 2006, Froese & Pauly 2009). The inshore small planktivores group also included silverside (Atherinidae), sprats (Clupeidae) and the surf sardine (*Iso rhothophilus*). *P/B* and *Q/B* estimates of 1.01 and 7.30 were derived from estimates provided by (Bulman *et al.* 2006, Froese & Pauly 2009). Diet data were sourced from (Daly 2007) and summarised in Chapter 3.

Appendix 4.12 Australian Salmon and Australian herring (model group 23)

Australian Salmon (*Arripis truttaceus*) and Australian Herring (*A. georgianus*) are predominantly found in the gulfs, inshore areas in shelf waters and around offshore islands. Historically, the commercial catch has mostly been extracted from State waters using purse seine nets, gill-nets, haul-nets and hand-lines, and the product is used for bait and human consumption. Catch and effort time series data for these species from SARDI logbook systems were extracted from between 1991 and 2008. Biomass was estimated by the model. *P/B* and *Q/B* estimates of 0.44 and 5.4 were sourced from (Bulman *et al.* 2006, Froese & Pauly 2009). Diet data for *A. truttaceus* were sourced from (Caines 2005) and Page *et al.* (Chapter 3).

Appendix 4.13 Small to medium demersal fishes (model groups. 24–27, 29)

The small to medium demersal fish groups included small demersal piscivores, small demersal invertebrate feeders, medium demersal piscivores, medium demersal invertebrate feeders and small demersal omnivores. Catch and effort time series data from logbook systems managed by AFMA/CSIRO and SARDI were available for 63 taxa within these groups between 1977 and 2008. Typical taxa included *Platycephalus* spp., *Centroberyx* spp. *Neocyttus* spp. in shelf waters, and Monocanthidae and *Silliginodes* spp. in gulf waters. The main gear types used were fish and prawn bottom trawl, shark gill-net and hand-lines in the gulfs. Many of these species are caught when deepwater flathead and other scalefish were targeted on the shelf, and western king prawn were targeted in State waters. Biomass datasets were available for this model group from estimates for Spencer Gulf in Currie *et al.* (2009). For medium demersal piscivores, the upper 95% CL estimate based on 34 species in Spencer Gulf (0.302 t.km^{-2}) was used to estimate biomass as it was the minimum biomass range that achieved model balance (Currie *et al.* 2009, Currie & Sorokin 2010). For small demersal piscivorous fish the upper 95% CL estimate based on 15 species in Spencer Gulf (1.467 t.km^{-2}) was used to estimate biomass (Currie *et al.* 2009, Currie & Sorokin 2010). For small demersal invertebrate feeders, the upper 95% CL estimate based on 8 species in Spencer Gulf (0.0786 t.km^{-2}) was used to estimate biomass, as the mean estimate did not achieve model balance (Currie *et al.* 2009, Currie & Sorokin 2010). For small demersal omnivores, the upper 95% CL (0.13 t.km^{-2}) for 13 species in Spencer Gulf was too low to balance the model, and was raised to 0.17 t.km^{-2} to get EEs below 1. The biomass of the small demersal invertebrate group was estimated by the model. *P/B* estimates for these model groups ranged from 0.1 to 0.6, and *Q/B* ranged from 1 to 12.4 (Froese & Pauly 2009, Bulman *et al.* 2010, Currie & Sorokin 2010). Diet data were sourced from the literature (Currie *et al.* 2009, Currie & Sorokin 2010).

Appendix 4.14 Mesopelagics (model group 28)

Mesopelagic fishes including Myctophidae and *Nansenia* spp. typically occupy the shelf slope, which represents the outer spatial bounds of our modelled region. These species are not targeted by commercial fisheries in the study region, but are incorporated in the diets of some predator groups in the model. There is minimal data available on the distribution, abundance or biomass of this model group in the study region. *P/B* and *Q/B* estimates were 1.005 to 6.673, respectively (Bulman *et al.* 2006, Froese & Pauly 2009). Diet data were sourced from Bulman *et al.* (2001, 2006).

Appendix 4.15 Cephalopods (model group 30–33)

The cephalopod group consists of ‘arrow squids’, southern calamary, ‘other squids’, and ‘octopuses’. All of these groups are commercially harvested but only limited catch data was available for Commonwealth waters for the SET and GABT fisheries. Calamary (*Sepioteuthis australis*) and giant cuttlefish (*Sepia apama*) is targeted in the MSF fishery in State waters. Biomass datasets were unavailable for all model groups within the EGAB and were estimated by the model. *P/B* and *Q/B* estimates typically ranged from 1.95 to 2.5 and from 3.9 to 5.85, respectively (Bulman *et al.* 2006, Froese & Pauly 2009). Diet data were sourced from Braley *et al.* (2010), Bulman *et al.* (2006), Grubert *et al.* (1999) and Page *et al.* (Chapter 3).

Appendix 4.16 Zooplankton (model groups 34–35)

The ‘small zooplankton group’ in our input dataset comprised copepods, pteropods and ostracods, and the ‘large zooplankton’ group consisted of krill (*Nyctiphanes*), copepods and amphipods. These groups are not commercially harvested in SA Commonwealth or State waters and there were no published data on their biomass or production dynamics groups in the study region. Biomass was estimated by the model. We used estimates of 20 and 70, and 5 and 32 for *P/B* and *Q/B*, for small and large zooplankton based on Bulman *et al.* (2006) that were originally derived from studies in the Northern Hemisphere (Guenette and Morato (2002).

Appendix 4.17 Benthic grazers (megabenthos) (model group 36)

The study region is characterised by diverse communities of megabenthos or ‘benthic grazers’. The main commercially exploited species in SA State and Commonwealth waters include southern rock lobster, western king prawns and blue crabs. Commercial catch, effort, discard and bycatch data for southern rock lobster and blue crab were not included for the purpose of this model, but time series of catch and effort for the western king prawn fishery was used between 1991 and 2008. Biomass for this group was estimated by the model. We used estimates of 1.6 and 6.0 for P/B and Q/B, respectively (Bulman *et al.* 2006). Diet data were sourced from the literature (Bundy 2001, Bulman *et al.* 2006).

Appendix 4.18 Detritivores (infauna-macrobenthos) (model group 37)

The detritivore group consists of polychaete worms and other infaunal invertebrates. We had no input data available for the modelled region, and used estimates of 1.6 and 6 for P/B and Q/B, respectively (Bulman *et al.* 2006). Diet data were sourced from the literature (Bundy 2001, Bulman *et al.* 2006).

Appendix 4.19 Filter feeders (model group 38)

This model group consisted of bivalves and molluscs. Estimates of *P/B* and *Q/B* of 1.6 and 6, respectively were used as model input data based on Bulman *et al.* (2006).

Appendix 4.20 Primary Producers (model group 39)

Estimates on phytoplankton biomass and primary productivity (P/B) in the EGAB ecosystem were based on data from van Ruth (2009) and van Ruth *et al.* (2010). We used the mean (14.9 t km^{-2}) of the estimates of phytoplankton biomass from near-shore (25.1t km^{-2}), mid-shelf (12.7 t km^{-2}) and offshore (6.8 t km^{-2}) stations from sites in the eastern, central and western parts of the EGAB.. Primary productivity estimates were derived by taking the mean (745 t y^{-1}) of the median values from near-shore (800 t y^{-1}), mid-shelf (688 t y^{-1}) and offshore (746 t y^{-1}) stations (van Ruth 2009, van Ruth *et al.* 2010).

Appendix 4.21 Detritus (model group 40)

Detritus is comprised of organic matter lost to the benthos. There were no published data on detritus deposition per square km in the study region. We used a ‘global’ biomass estimate of 10 t km^{-2} as used by Bulman *et al.* (2006).