Spencer Gulf Research Initiative:

Development of an Ecosystem Model for Fisheries and Aquaculture

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Abbreviations

CPUE	Catch per unit effort
CS	Control nutrient scenario (current nutrient inputs)
DDF	Deposit detritovore feeder
DEM	Digital elevation model
DEWNR	Department of Water, Environment and Natural Resources
DIN	Dissolved inorganic nitrogen
DL	Large detritus
DN	Dab net
DO	Dissolved oxygen
DOM	Dissolved organic matter
DS	Small detritus
EE	Ecotrophic efficiency
F	Fishing mortality
FF	Forcing functions
FIB	Fishing in balance
GN	Gill net
HL	Hand line
HN	Haul net
LL	Long line
MISA	Marine Innovations Southern Australia
mTLC	Mean trophic level of the catch
NH_4	Ammonium
NO_3	Nitrate
NPZD	Nutrient, phytoplankton, zooplankton, detritus
P	Phytoplankton
P/B	Production/biomass
PCS	Prawn catch scenario; 1, 2, & 3 represent 10, 20 & 30% increases in western king prawn catch
PES	Prawn effort scenario; 1, 2, & 3 represent 10, 20 & 30% increases in western king prawn effort
PIRSA	Primary Industries and Regions South Australia
POLR	Proportional odds logistic regression
POM	Particulate organic matter
PS	Purse seine
PZD	Phytoplankton, zooplankton and detritus
ROMS	Regional Ocean Modelling System
S1	Nutrient reduction scenario; only nutrient source was from the shelf
S2	Nutrient increase scenario; 4-fold increase in nutrients from aquaculture
SAIMOS	Southern Australian Integrated Marine Observing System
SAO	Sand-associated ominvore
SAROM	South Australian Regional Ocean Model
SBT	Southern bluefin tuna
SCS	Sardine catch scenario; 1, 2, & 3 represent 10, 20 and 30% increases in sardine catch
SGEDI	Spencer Gulf Ecosystem Development Initiative
SGM	Spencer Gulf model
SS	Sums of Squares
SST	Sea surface temperature
TL	Trophic level
Q	Kempton’s index of biodiversity
Q/B	Consumption/biomass
WWTP	Waste water treatment plant
YTK	Yellowtail kingfish
Z	Zooplankton
ZF	Zooplankton feeding

Executive Summary

What the report is about

A Spencer Gulf ecosystem model has been developed for the fisheries and aquaculture sectors. The model is a trophic mass-balance model that incorporates information from a coupled hydrodynamic-biogeochemical model and habitat distribution models for seagrass and macroalgae. An ecosystem perspective of Spencer Gulf is provided that simulates nutrient inputs from finfish aquaculture, and increasing fishing effort and catches of sardines and western king prawns. Scenarios associated with finfish aquaculture demonstrated how bottom-up changes through additional nutrient loading may affect lower trophic groups, benthic and pelagic ecosystems. Scenarios associated with changes in catch in the sardine fishery demonstrated potential impacts from removal of a key small pelagic fish species. Changes in catch and fishing effort of western king prawns allowed impacts of fishing on benthic communities to be investigated. The model allows additional scenarios to be investigated in future.

Background

Spencer Gulf represents an area of significant economic importance to the fisheries and aquaculture sectors of South Australia. The region is also experiencing considerable industrial growth in terms of mining and other activities. Given this level of proposed development in the region it is imperative that the fisheries and aquaculture sectors have a better understanding of how future developments may impact their industries. This project was undertaken to provide a whole of Spencer Gulf ecosystem model capable of addressing "what if" scenarios for the fisheries and aquaculture sectors. The Spencer Gulf Ecosystem model is needed to provide fisheries and aquaculture managers with sound, evidenced-based information on the impacts of current and future developments in Spencer Gulf.

Objectives

The objectives of this project were to (1) conduct an ecosystem-based assessment of the fisheries and aquaculture industries in Spencer Gulf, which includes the establishment of performance indicators for ecosystem health and (2) develop a suite of linked habitat, biophysical, and trophodynamic models that can be used to assess the future ecological performance of the seafood industry in Spencer Gulf.

Methodology

A coupled hydrodynamic-biogeochemical model for Spencer Gulf was developed using the open source Regional Ocean Modelling System. The model used 'whole of gulf' observations on dissolved inorganic nitrogen, nitrate, ammonium, phytoplankton, zooplankton, and small and large detritus. Nutrient scenarios were run to simulate the influence of varying nutrient loads from finfish aquaculture on the lower trophic ecosystem: a control scenario reflecting current nutrient sources that was used for validation purposes; scenario 1 representing nutrient reduction with the only nutrient source being shelf waters, and scenario 2 representing a four-fold increase in nutrients from finfish aquaculture.

Benthic primary production is an important component of the Spencer Gulf ecosystem; however, the majority of benthic habitats within the Gulf remain unsurveyed. Spatial drivers of seagrass and macroalgae cover were investigated to predict Gulf-wide distributions of these habitat types. Available spatial data on seagrass and macroalgae cover were sourced from the South Australian State Benthic Habitats layer and spatial covariates (seabed depth, temperature and salinity, detritus content, current velocity, predicted occurrence of rocky reef habitat) were sourced from a variety of sources. Statistical models were fitted using occurrence data for each habitat type, as well as using percent cover data for seagrasses. A model selection process was used to identify useful predictors of spatial distribution. Statistical models for seagrass performed well and the selected models were used to predict the occurrence and cover of seagrasses throughout unsurveyed regions of Spencer Gulf. Biomass and productivity estimates for seagrasses and macroalgae habitat within Spencer Gulf were derived using parameters from previous studies.

A map of predicted seagrass occurrence and percent cover of seagrasses conditional on seagrass presence was generated for Spencer Gulf and used in the Spencer Gulf ecosystem model. However, macroalgae occurrence could not be predicted confidently due to a lack of reliable information regarding the presence of rocky reef habitat across the unsurveyed regions of Spencer Gulf.

Ecopath with Ecosim software was used to develop a trophic mass-balance model of the Spencer Gulf ecosystem. The *Ecopath* model was constructed for 1991 and the *Ecosim* model developed for a 20 year time period (1991-2010). The model incorporated 78 functional or trophic groups based on similarities in diet, habitat, foraging behaviour, size, consumption and rates of production, as well as 27 fishing fleets for which landings and effort data were available for the 20 year period and two aquaculture industries. Model fitting initially occurred in *Ecosim*, after which four variables (total catch, Kempton's index of biodiversity, mean trophic level of the catch and fishing in balance index) were evaluated to ascertain changes in the marine ecosystem. A preliminary spatially explicit model (*Ecospace*) was also developed.

Three 'what if' scenarios were simulated through trophodynamic modelling: (1) finfish aquaculture (no nutrient inputs from aquaculture, current aquaculture nutrient loadings and a future increased aquaculture nutrient scenario), (2) sardine catch (0, 10, 20 and 30% increases), and (3) western king prawn fishing effort and catch (0, 10, 20 and 30% increases). Changes in biomass of the modelled groups were investigated for all scenarios.

Results

The lower part of the Spencer Gulf food web was dominated by crustaceans, seagrass and macroalgae and the higher part by sharks, Australian gannet, barracouta, yellowtail kingfish, southern bluefin tuna, common dolphin, New Zealand fur seal and southern calamari. The *Ecosim* model estimates of biomass tracked the observed catch per unit effort data reasonably well for most groups, and estimated catches tracked actual catches reasonably well, except for whaler sharks and rock lobster. An increase in total catch over the study period was driven by rapid growth of the sardine fishery which overrode declines in catch in several other fisheries.

Significant changes in biological response were found in relation to changing nutrient conditions. Reduction of anthropogenic discharges (scenario 1) led to slight reductions in nutrients, phytoplankton and detritus, and greater decreases in zooplankton particularly during the winter months. Scenario 2 (4-fold increase in nutrients from aquaculture) resulted in increases in nutrients, phytoplankton and detritus levels with the greatest response in zooplankton abundance.

Model scenarios suggest that increased nutrient loading from aquaculture will results in increases in phytoplankton and detrital levels and a likely increase in gelatinous zooplankton. The maximum nutrient scenario may lead to a reduction in production of benthic systems and ultimately to declines in the abundance of benthic foraging predators. Conversely, under the maximum aquaculture scenario there are increases in the biomass of most small pelagic fish and their predators.

Some change has already occurred in the Spencer Gulf ecosystem. Current levels of fishing effort do not appear to be having major deleterious effects. Several measures of the marine ecosystem increased slightly over time (mean trophic level of catch, fishing in balance index), but in general biodiversity measures were similar throughout the 1990s and 2000s.

Increased exploitation rates of sardines and western king prawns impact most on the target species and their prey. Increased exploitation rates of sardines appear to benefit some trophic groups, namely squid, anchovy and blue mackerel, with concomitant benefits to organisms that prey on these groups. The likely mechanism for the increases in other pelagic fishes is release from competition from sardines. Similarly, increased fishing for western king prawns is associated with small increases in the biomass of squid, anchovy and fish groups. When bycatch of non-target species in the prawn fishery is considered, the increased fishing effort scenarios result in declines in several trophic groups. These were an order of magnitude greater than those observed for scenarios that only considered catches of target species.

Implications for relevant stakeholders

Once further refined the Spencer Gulf ecosystem model provides industry and managers with a tool to investigate drivers of change in ecosystem indicators, as well as changes in catch of key fisheries (e.g. sardines, western king prawns) and levels of aquaculture production. How changing effort in the Spencer Gulf prawn trawl fishery affects important bycatch species can also be investigated in more detail including in relation to bycatch reduction devices.

Recommendations

Spencer Gulf is impacted by a complex array of marine and land-based industries and is an ideal location to develop and establish the integrated scientific frameworks that are needed to underpin the ecologically sustainable management of crowded marine environments. The current project provides proof-of-concept for the coupling of several different models to assess how two types of human activities (i.e. fishing and aquaculture) can interact with each other and affect ecosystem structure and function. This project is an important first step towards the establishment of scientific tools that can be used to determine the individual and cumulative impacts of all industries that operate in and around Spencer Gulf. Such tools are needed to ensure that both the health of these ecosystems is maintained and the socio-economic benefits of these industries to the community is maximised.

Keywords

Spencer Gulf; aquaculture; hydrodynamic-biogeochemical model; macroalgae; seagrass; habitat model; trophic mass-balance model; sardine fishery, western king prawn fishery; *Ecopath with Ecosim*; *Ecospace*

1 Introduction

1.1 Background

Spencer Gulf is recognised for its clean, green image in terms of its seafood production. Both wildcatch (e.g. western king prawns, snapper, garfish, King George whiting, abalone, southern rock lobster, sardines) and aquaculture (southern bluefin tuna, yellowtail kingfish, abalone, oysters, mussels) in Spencer Gulf provide important economic returns to the State and are expanding. Of particular note, is the likely expansion of southern bluefin tuna aquaculture with two new growout zones being finalised in 2011. Furthermore, positive growth in all aquaculture sectors is predicted by industry participants. South Australia, and in particular the Spencer Gulf region, is also on the cusp of a mining boom, with a large number of new mineral/mineral processing ventures possible. A major expansion of existing mining activities is anticipated. Associated with such development will be increased shipping and associated infrastructure (port development, desalination plants, power plants) along with potential biosecurity risks. Spencer Gulf is also the focus of several marine parks and zoning of sanctuary areas is underway. These are likely to conflict with both existing and proposed policy commitments associated with transport, aquaculture and fishing. The region has important relict populations of tropical species (e.g. commercially fished blue crab), and is the only area in the world known to support a breeding aggregation of cuttlefish. It is an important nursery area for fish. These multiple and often competing attributes provide fertile ground for ongoing policy and natural resources management issues, and there are likely to be significant challenges ahead in balancing mineral, fishing/aquaculture, and conservation and recreation needs, while simultaneously delivering on the environmental, social and economic objectives associated with Spencer Gulf

A challenge for Spencer Gulf is ecologically sustainable development and optimal use of its marine natural resources, especially since there are a range of economic interests that profit from use of different resources. Here, our focus is on the fishing and aquaculture industries. Despite the global recognition of the need to adopt a sustainable approach to the management of fisheries and aquaculture (Pikitch et al. 2004), there are still few examples where ecosystem-based management has been adopted to complement the existing single-species/stock management paradigm. Ecosystem-based management aims to sustain healthy marine ecosystems and in particular the fisheries and aquaculture of the region, but no such model currently exists for all of Spencer Gulf. Such models emphasise habitat and ecosystem function and need to incorporate spatial structure and environmental processes. The Spencer Gulf ecosystem model will improve management and use of natural resources to ensure their sustainability.

Worldwide, significant progress is now being made on ecosystem based fisheries management, particularly in the USA (e.g. Alaska), Canada and the European Union. In the southern hemisphere the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) and within Australia the Commonwealth managed Southern and Eastern Scalefish and Shark Fishery (SESSF) are also making progress. Goldsworthy et al. (2011, 2013) developed ecosystem models for the eastern Great Australian Bight (GAB). Although these models included the lower half of Spencer Gulf, they were principally models that focused on the pelagic components of this large marine domain.

Marine Innovation SA (MISA) has also identified Spencer Gulf as a key area in which to develop knowledge platforms that inform sustainable marine management and decision-making in South Australian waters. Identified needs include an ecosystem model with the capability to address “what if” scenarios. Currently our ability to predict what could happen under various scenarios is limited, but essential for improved management and use of fisheries and aquaculture.

1.2 Need

Spencer Gulf represents an area of significant economic importance to the fisheries and aquaculture sectors of South Australia. The region also has the potential for considerable industrial growth in terms of mining and other activities. For example, in 2011 in Spencer Gulf, there was the release of BHP-Billiton's supplementary Environmental Impact Statement for the massive Olympic Dam expansion including a coastal desalination plant (now delayed, and approaches for obtaining water being reviewed), the State government revived plans for a deep water port at Port Bonython (in Upper Spencer Gulf) which is predicted to be a major hub for the export of mineral resources (currently awaiting approval), and IronClad mining announced plans for a floating harbor south of Port Bonython (the proposal is for transshipment of material from Lucky Bay Harbour Panamax size export vessels). Since then there have been further companies indicating they propose to build deepwater ports in Spencer Gulf. Given this level of proposed development it is imperative that the fisheries and aquaculture sectors have a better understanding of how future developments may impact their industries. However, as a first step, an ecosystem model focusing around fisheries and aquaculture is required.

This project was needed to provide a whole of Spencer Gulf ecosystem model to the fisheries and aquaculture sectors with the capability to address "what if" scenarios around impacts on the ecosystem. A suite of linked habitat, biophysical, trophodynamic models have been developed that can be used to assess the future ecological performance of the seafood industry in Spencer Gulf.

The area of the Spencer Gulf Ecosystem model used for oceanographic, habitat and trophodynamic models is shown in Figure 1.1.

Figure 1.1 Area of the Spencer Gulf Ecosystem (SGE) (shaded blue) used in the development of the various models.

1.3 Objectives

The objectives of this project were to:

(1) Conduct an ecosystem-based assessment of the fisheries and aquaculture industries in Spencer Gulf, which includes the establishment of performance indicators for ecosystem health[1].

(2) Develop a suite of linked habitat, biophysical and trophodynamic models that can be used to assess the future ecological performance of the seafood industry in Spencer Gulf.

This following sections of the report focus on the hydrodynamic-biogeochemical model (section 2), seagrass and macroalgal distributions (section 3) and the ecosystem model using Ecopath with Ecosim software (section 4) for the Spencer Gulf region. Each section has a brief introduction; detailed methods and results along with a discussion are provided. These sections are brought together in the conclusion, implications, recommendations and further development sections.

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[1] A healthy ecosystem can be described as a system that is resilient, maintains intact ecosystem functions (population survivorship, growth and replenishment), displays functional response diversity and is able to provide goods and services. Further, a healthy ecosystem has a suite of trophic levels and good spatial connectivity among subsystems. This definition is taken from the white paper on Ecosystem Health which formed one of the white papers for the Biodiversity Conservation and Ecosystem Health paper submitted to the National Marine Science Committee in 2014.

2 A three-dimensional hydrodynamic-biogeochemical model for Spencer Gulf

2.1 Introduction

Spencer Gulf encompasses 21,700 km^2, has an average depth of ~24 m and is the largest estuary along the southern Australian coastline. Characterised by a semi-arid climate (Schwarz 2003), the Gulf experiences strong seasonal heating and high evaporation. This climatology drives an inverse estuarine circulation pattern (Nunes and Lennon 1986) which is accompanied by generally low primary productivity (Smith and Veeh 1989, Nunes Vaz et al. 1990, van Ruth and Doubell 2013) and relatively oligotrophic waters (Smith and Atkinson 1983, van Ruth and Doubell 2013). Spencer Gulf is also the location for an active finfish aquaculture industry worth approximately $228M (Econsearch 2012) (Fig. 2.1).

In contrast to typical (classical) estuaries where freshwater inputs of allochthonous and anthropogenic nutrients help sustain high biological productivity (Hobbie 1988), Spencer Gulf does not receive significant inputs from rivers. For this reason, the current and expected anthropogenic nutrient loads associated with the expanding use of the Gulf are becoming increasingly recognised as a major factor potentially influencing water quality and ecosystem structure and productivity (Middleton et al. 2013). For temperate coastal marine systems, nitrogen is considered the key nutrient which limits phytoplankton growth (Nixon 1995, Howarth and Marino 2006). The cycling of nitrogen by microbes and plankton can have a major impact on the fate and distribution of biological and chemical components in marine ecosystems. In Spencer Gulf, nitrogen from the finfish aquaculture industry provides the largest source of dissolved nutrients discharged into the Gulf (Gaylard 2014). It is estimated the sea-cage farming of southern bluefin tuna (SBT) and yellowtail kingfish (YTK) account for approximately 95% of the Gulfs annual anthropogenic nitrogen load.

Recently, a coupled hydrodynamic-biogeochemical model for Spencer Gulf (SGM) was developed and validated to study the effect of anthropogenic nutrient discharges, particularly those from finfish aquaculture, on eutrophication and carrying capacity in Spencer Gulf (Doubell et al. 2013). Scenario studies using the model provided a tool for assessing the integrated effects of varying aquaculture nutrient loads on the distribution and transport of dissolved nutrients and their influence on the lower trophic ecosystem. This section summarizes the SGM, and presents results for three model scenario studies which show the planktonic ecosystem response to different nutrient loadings associated with varying levels (or none) of finfish aquaculture production. For each scenario study, predicted values and concentrations for a range of physical (e.g. temperature, salinity, currents) and biological (e.g. phytoplankton, zooplankton and detritus) variables were then provided for incorporation into habitat (see section 3) and higher trophic level models (see section 4).

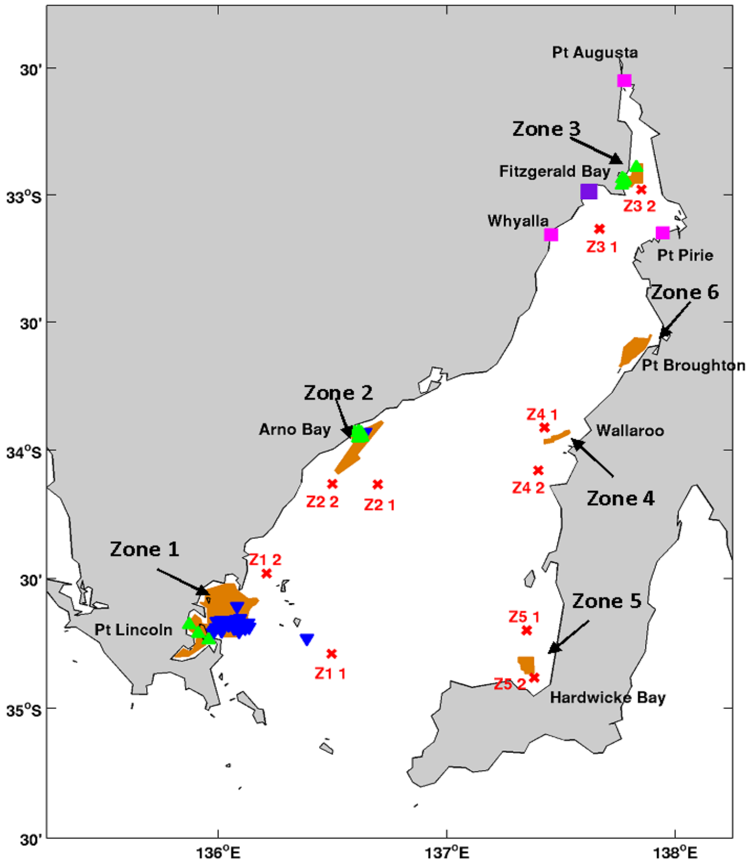


Figure 2.1 Map of Spencer Gulf showing the location of the six aquaculture zones (brown shaded regions), field survey sites (red crosses), tuna (blue triangles) and finfish (green triangles) aquaculture leases, waste water treatment plants (pink squares) and the OneSteel steel works (purple square). The location of aquaculture leases is plotted for the 2010/11 period. The Port Lincoln WWTP was not included as it is a relatively minor source of nutrients.

2.2 Methods

2.2.1 Hydrodynamic model

The coupled hydrodynamic-biogeochemical model for Spencer Gulf was developed using the open-source ROMS (Regional Ocean Modelling System; www.myroms.org). The hydrodynamic model was run over a series of nested spatial domains. The Spencer Gulf model (SGM) used in this work has a horizontal grid spacing of 1200m and 7 vertical layers. Conditions for velocity, temperature and salinity at the open boundaries were obtained from the large-scale South Australian Regional Ocean Model (SAROM) and a time step of 150 seconds allowed the model to solve the dominant tidal currents occurring in the Gulf. Full details regarding the hydrodynamic model setup, forcing and validation are given in Luick et al. (2013).

2.2.2 Biogeochemical model

The ROMS biogeochemical model of Fennel et al. (2006) was coupled to the SGM and adapted to the ecology of the Spencer Gulf marine system (Doubell et al. 2013). The Fennel model is specifically designed to understand and quantify biogeochemical cycling through the lower trophic ecosystem in coastal and estuarine systems (Fennel et al. 2006, Bianucci et al. 2012). The model provides a representation of the pelagic nitrogen cycle using seven state variables (Figure 2.2); dissolved inorganic nitrogen (DIN), nitrate (NO_3), ammonium (NH_4), phytoplankton (P), zooplankton (Z), small detritus (DS) and large detritus (DL). All state variables have common units (mmol N m^{-3}). The model also tracks dissolved oxygen (DO) and includes a sediment component to simulate microbial benthic mineralisation processes which can significantly influence the cycling of nitrogen (Nixon and Pilson 1983, Bianucci et al. 2012).

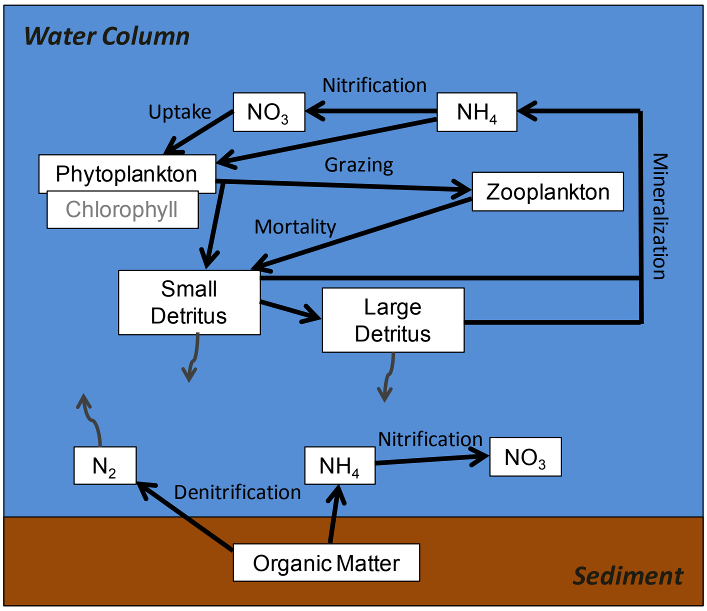


Figure 2.2 Schematic representation of the Fennel et al. (2006) biogeochemical model.

The biogeochemical model was developed following the first, extensive, 'whole of gulf' observational program (van Ruth and Doubell 2013). This field program consisted of 10 monthly surveys made over a period of 1 year between July 2010 and July 2011. In situ measurements of physical, biological and chemical parameters as well as seasonal studies to quantify the rates of key biological processes (e.g. primary and productivity, zooplankton grazing and phytoplankton mortality) were conducted. The collected data provided a complete assessment of the Spencer Gulf lower trophic ecosystem which was necessary for model parameterization.

The biogeochemical model was run for a period of one year commencing on 1st July 2010. Initial values for nutrients and phytoplankton were set to spatially constant values equal to the mean winter concentrations measured during the field surveys. All other state variables were initially set to a constant, small value of 0.1 mmol N m^{-3}. Conditions for nutrients and phytoplankton along the open model boundaries were derived from observations taken through the Southern Australian Integrated Marine Observing System (SAIMOS).

The model was first run for a 'spin up' period of 1 year. Adjustment time scales for each of the model's state variables were short (on the order of weeks). Daily averaged values for each state variable output on 31st June 2011 were then used as the initial conditions for all subsequent model simulations beginning on 1st July 2010. The physics of the hydrodynamic model were the same for each model run. Validation against data collected during the 2010-2011 field survey period demonstrated the model is capable of reproducing the general distribution of nutrients, phytoplankton, zooplankton and dissolved oxygen. Full details regarding the configuration and validation of the ecosystem model are given in Doubell et al. (2013).

2.2.3 Anthropogenic nutrient sources and scenario studies

A series of scenario studies were run using the model to simulate the influence of varying nutrient loads from supplementary fed finfish aquaculture on the lower trophic ecosystem. Table 2.1 summarises these scenarios and their corresponding nutrient sources and annual loads.

Table 2.1 Summary of the model scenario studies, associated sources and annual anthropogenic nutrient loads. Nutrient loads for dissolved inorganic nutrients are in units of kilotons year^{-1} (kT y^{-1}).

Scenario study	Nutrient load (k T y-1) Shelf (natural)	Aquaculture (SBT & YTK)	Other (WWTP & OneSteel)
Control (CS)	16.9	1.4	0.1
1 (S1)	16.9	0.0	0.0
2 (S2)	16.9	6.0	0.1

For the control scenario (CS), the model included the major sources of anthropogenic nitrogen in Spencer Gulf (Gaylard 2014) and included inputs from: finfish aquaculture, three SA Water waste water treatment plants (WWTP) and the OneSteel steelworks. Aquaculture monthly feed data for individual finfish leases were provided by PIRSA Fisheries and Aquaculture. Monthly feed data were converted into model units using the relationships given by Fernandes et al. (2007) for baitfish fed to southern bluefin tuna (SBT) and Fernandes and Tanner (2008) for pellets fed to yellowtail kingfish (YTK). Feed nitrogen contents of 3.25 and 7.10 % were used for baitfish and pellets, respectively. The amount of soluble nitrogen released from feeds was assumed to be 86% and 72% for SBT and YTK, respectively. As the soluble nitrogen released during farming is primarily the result of excretion, faecal leaching and sediment remineralisation, dissolved nitrogen inputs from aquaculture were assumed to be in the form of ammonium (Avnimelech 1999, Schendel et al. 2004). Nutrient loads from WWTP's and OneSteel were significantly smaller compared to those from aquaculture, and contributed less than 10% of the annual anthropogenic load. Nutrient loads from aquaculture were highest through the period March to July 2011.

Quantification of the annual load of dissolved inorganic nitrogen imported naturally from the shelf into the Gulf was estimated by calculating model-simulated fluxes across a boundary at the Gulf's entrance. Other natural sources of nitrogen include inputs derived through pelagic nitrogen fixation (Messer et al. 2014) which have yet to be quantified. Moreover, only readily quantifiable anthropogenic nutrient loads from point sources were included in the simulations. Smaller loads of anthropogenic nitrogen, such as those from stormwater, septic tank systems and agricultural run-off which are influenced by the low, ephemeral rainfall characteristic of the region are expected to have an intermittent enrichment effect on nearshore waters (Gaylard 2014).

The control scenario provided a simulation of the 2010-11 period and was used for validation purposes (discussed above). Scenario 1 (S1) is an example of a load reduction experiment; the import of nutrients and organic matter from the shelf region are the sole source of nitrogen for Spencer Gulf. No anthropogenic sources are included. Scenario 2 (S2) is an example of an increased aquaculture nutrient load experiment; finfish aquaculture loads were increased based on current PIRSA Fisheries and Aquaculture estimates of maximum stocking densities of 6 and 15 tonne ha^{-1} for SBT and YTK, respectively. Feed inputs for each lease and month were increased to reflect these production limits for a typical annual production cycle giving an approximate 4-fold increase in the annual nutrient load from aquaculture. The anthropogenic loads from WWTP's and OneSteel remained the same in the control scenario and Scenario 2.

2.3 Results and Discussion

2.3.1 Control scenario: simulation of the annual cycle

Model estimates showing seasonal variation in the circulation and associated hydrographic, chemical and biological fields are displayed in terms of seasonal means for the depth-averaged currents (Fig. 2.3) and monthly means for sea surface temperature and the concentrations of nutrients, phytoplankton, zooplankton and detritus are shown in Figures 2.4-2.9. The simulated annual cycle in

Spencer Gulf begins with elevated concentrations of phytoplankton in the south-western corner during winter (July-August 2010). Nutrients generally show an inverse behaviour compared to phytoplankton, and the supply of nitrates from the shelf is largest during the winter months (Fig. 2.5). During winter, nutrients from the shelf and aquaculture (Fig. 2.6) maintain increased levels of phytoplankton, and both nutrients and phytoplankton are transported northward along the western coastline and eastward into the southern region of the Gulf as a consequence of the inverse estuarine circulation (Fig. 2.3). In early spring (September-October), phytoplankton concentrations increase throughout the Gulf in response to seasonal changes in temperature and irradiance. Whilst the import of nitrate from the shelf is reduced, ammonium supplied from finfish aquaculture, OneSteel and WWTP's helps maintain elevated concentrations of phytoplankton in western and southern Spencer Gulf. The response of zooplankton to phytoplankton is greatest during spring (Fig. 2.8). Detritus levels (Fig. 2.9) typically reflect the cumulative behaviour of phytoplankton and zooplankton.

Nitrate and phytoplankton concentrations throughout the Gulf are lowest in summer (December-February). During summer, phytoplankton growth is maintained by low levels of ammonium supplied by anthropogenic sources and through benthic remineralisation processes. In early autumn (March-April), phytoplankton concentrations begin to increase again in the south-west corner of the Gulf. The inflow of nitrate from the shelf remains limited. Increases in ammonium discharges from aquaculture provide an additional source of nutrients for phytoplankton. By late autumn (May-June), nutrient concentrations increase from both the import of nitrate from the shelf and ammonium discharges from aquaculture. This increase supports phytoplankton growth. Elevated concentrations of nutrients and phytoplankton are again transported northwards along the western coastline and eastwards across the Gulf by the winter circulation. This completes the seasonal cycle.

2.3.2 Alternative scenario studies

Using the model, alternative scenario studies (S1, S2) were performed to estimate the influence of anthropogenic nutrient inputs on the ecosystem response. Table 2.1 summarises these scenarios and their corresponding nutrient sources and annual loads. Whilst, the physics of the system is not affected by the changing nutrient loads, the simulations show significant changes in biological responses. Figure 2.10 demonstrates that differences in the simulated concentration and distribution of organic matter (represented as phytoplankton), under the different scenarios, are not spread evenly across the Gulf, nor were they confined to the nutrient source regions.

To demonstrate the response of the ecosystem at the Gulf scale to decreases (Scenario 1) and increases (Scenario 2) in anthropogenic nutrient loads, changes in the daily averaged inventory of the chemical and biological variables are presented as a percentage of the control simulation (Fig. 2.11). The results for Scenario 1 indicate that the addition of anthropogenic discharges in the control scenario study have led to slight increases (<5 %) in the levels of nutrients, phytoplankton and detritus. Zooplankton levels showed the greatest response and indicate zooplankton biomass may have increased by approximately 10% across most of the year, with biomass levels increasing by as much as 75% in the last quarter of the simulated year.

Scenario 2 included significant increases in the annual (and monthly) nutrient discharges from aquaculture compared to the control scenario. Results indicate nutrients, phytoplankton and detritus respond rapidly to increases in the nutrient discharge from aquaculture. Total daily inventories for these variables increased by approximately 10% in comparison with the control scenario levels. Again, the greatest response was observed in the zooplankton with daily increases in biomass rapidly increasing to greater than 40% and reaching as high as 75% during the late summer/autumn period.

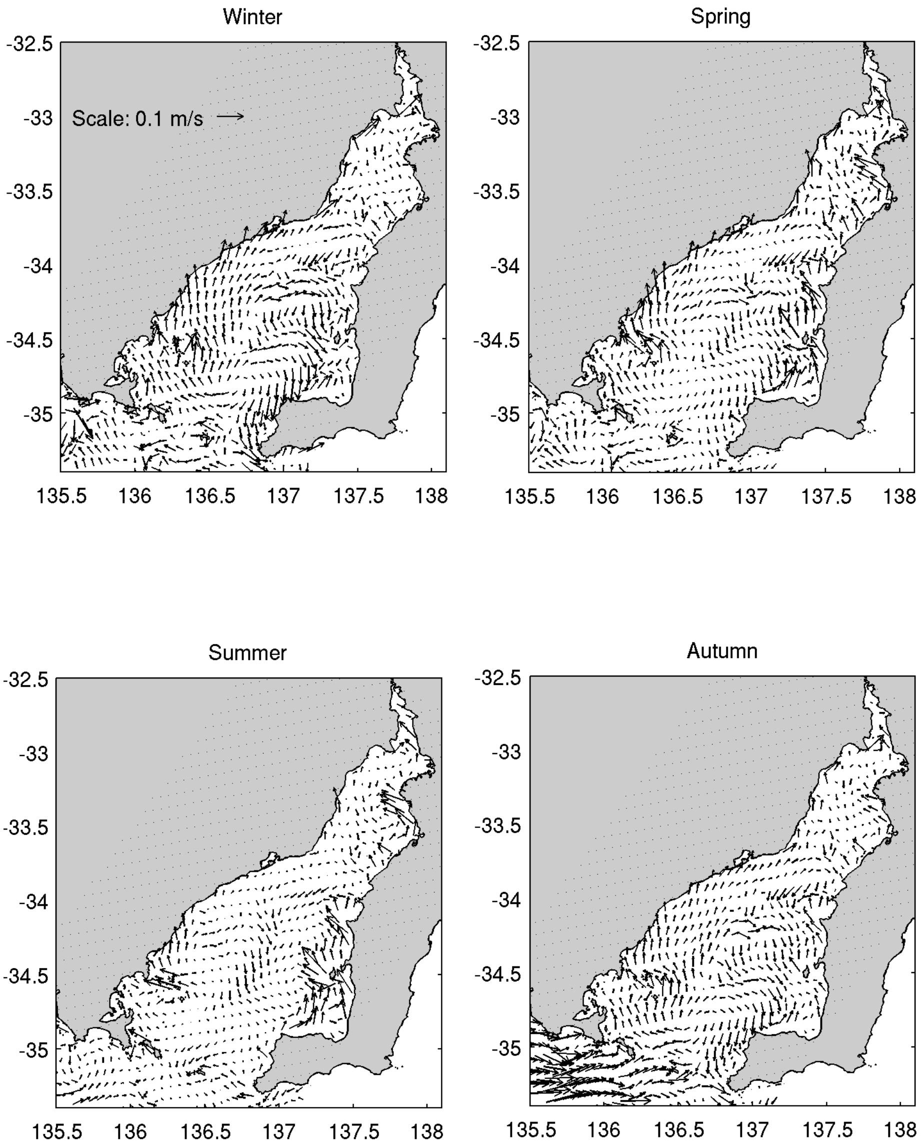


Figure 2.3 Mean of depth-averaged seasonal circulation patterns simulated in Spencer Gulf for the Winter (July-August), Spring (September-November), Summer (December-March) and Autumn (April-June).

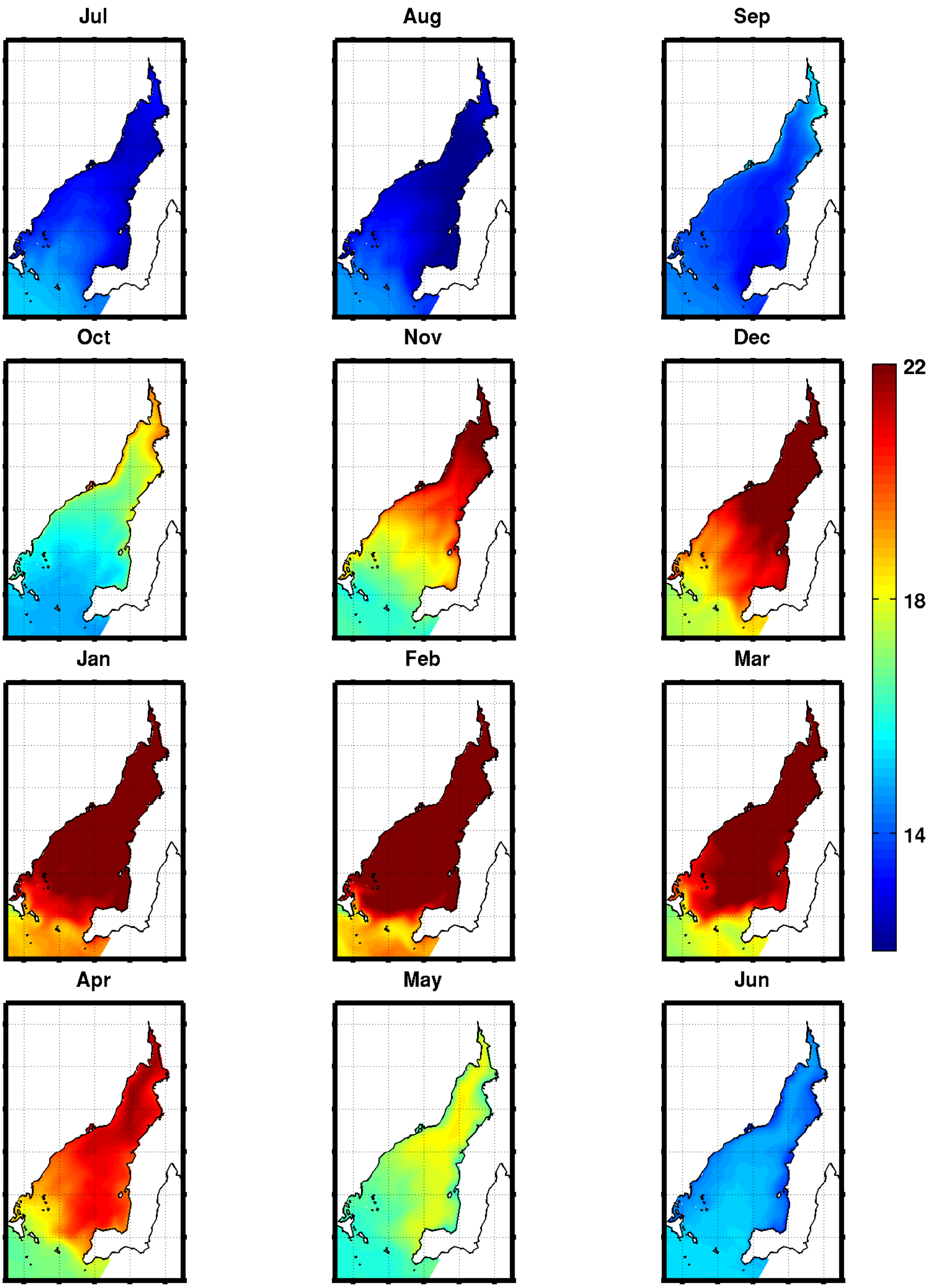


Figure 2.4 Monthly means of simulated surface temperature (^{o}C) for July 2010 to June 2011.

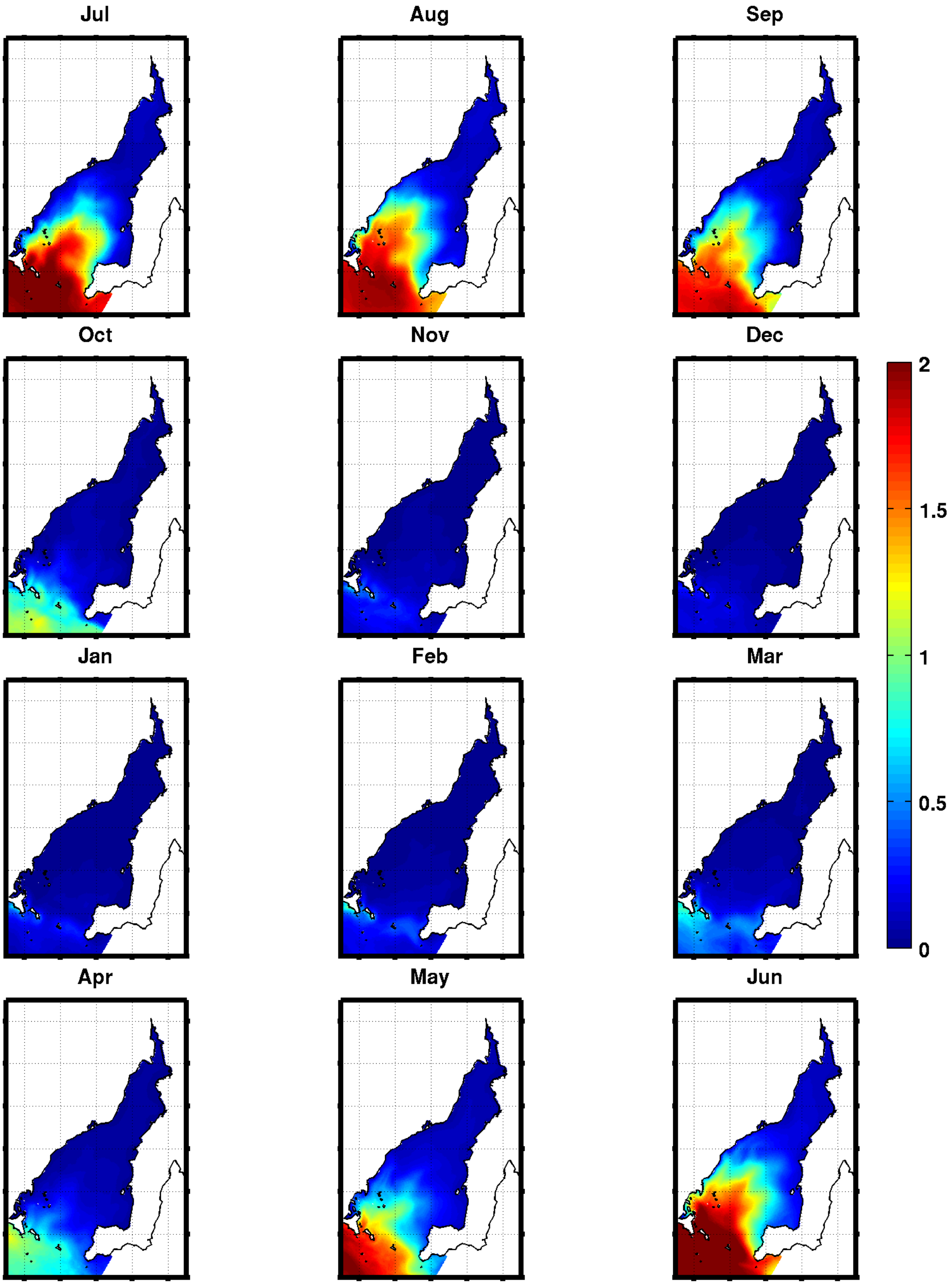


Figure 2.5 Monthly means of simulated surface nitrate (NO_3) concentrations (mmol N m^{-3}) for July 2010 to June 2011.

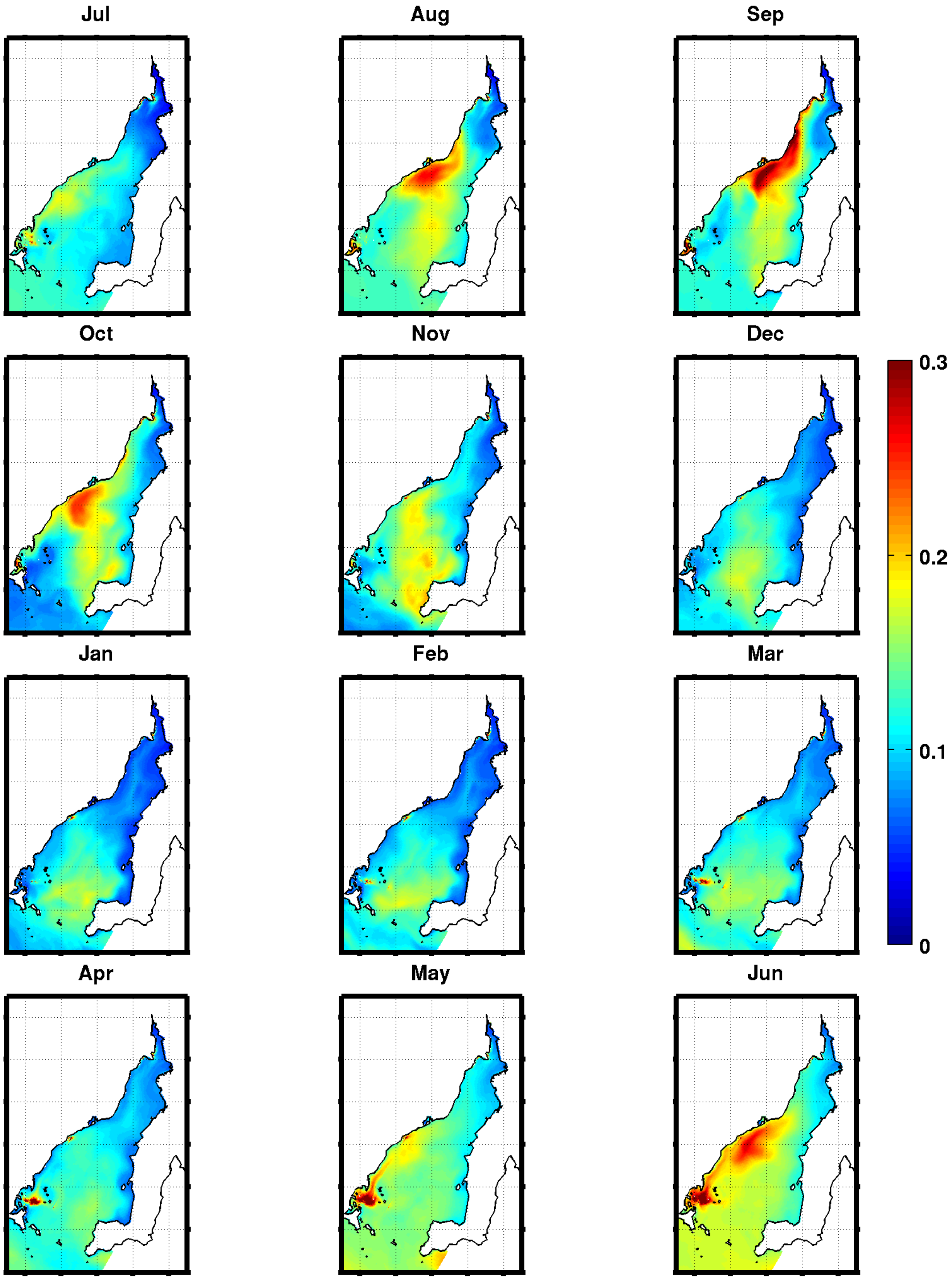


Figure 2.6 Monthly means of simulated surface ammonium (NH_4) concentrations (mmol N m^{-3}) for July 2010 to June 2011. Elevated concentrations from aquaculture loads are visible in aquaculture zones of Port Lincoln (Zone 1), Arno Bay (Zone 2) and Fitzgerald Bay (Zone 3) (see Fig. 2.1 for zone locations).

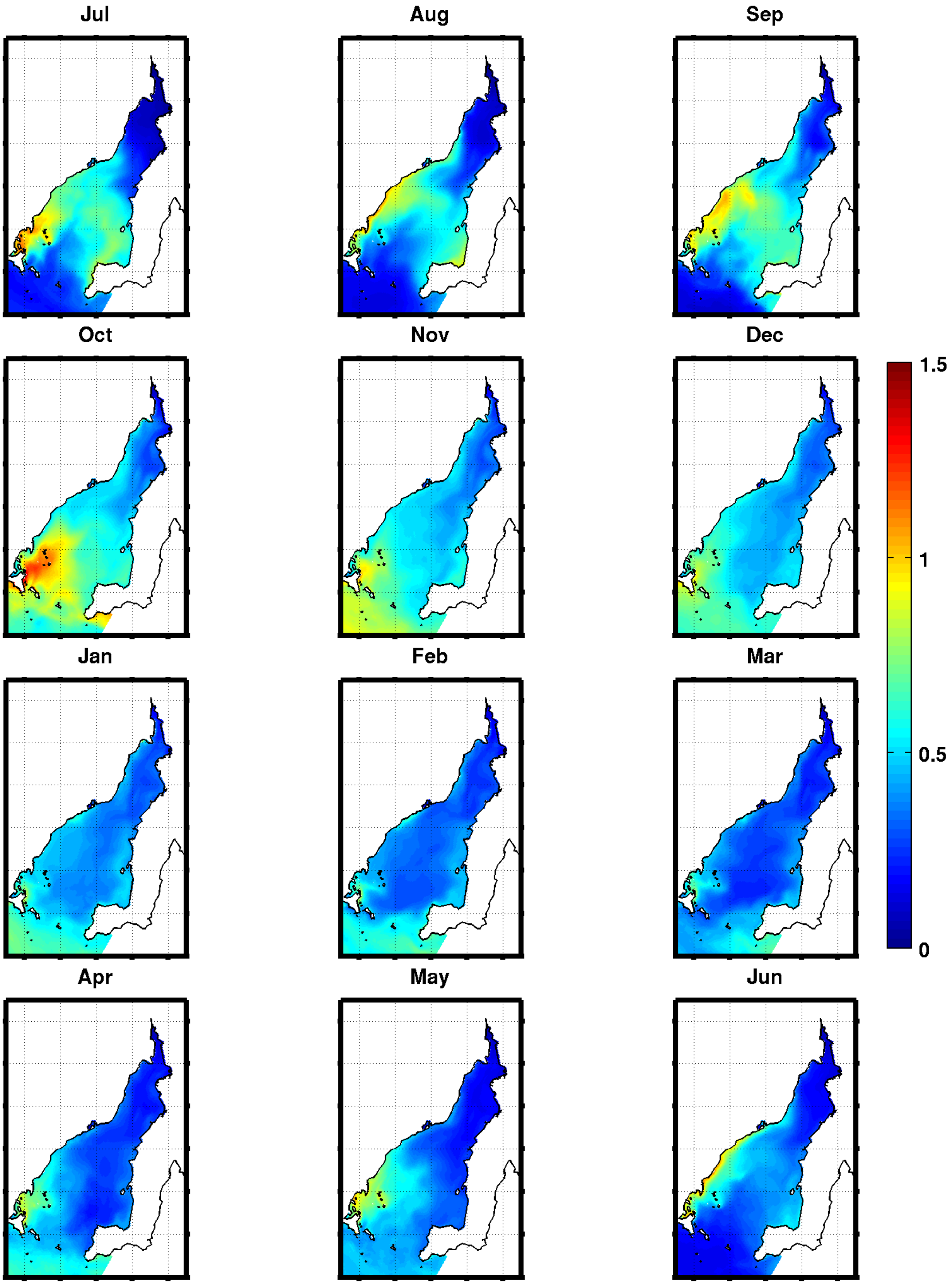


Figure 2.7 Monthly means of simulated surface phytoplankton concentrations (mmol N m^{-3}) for July 2010 to June 2011.

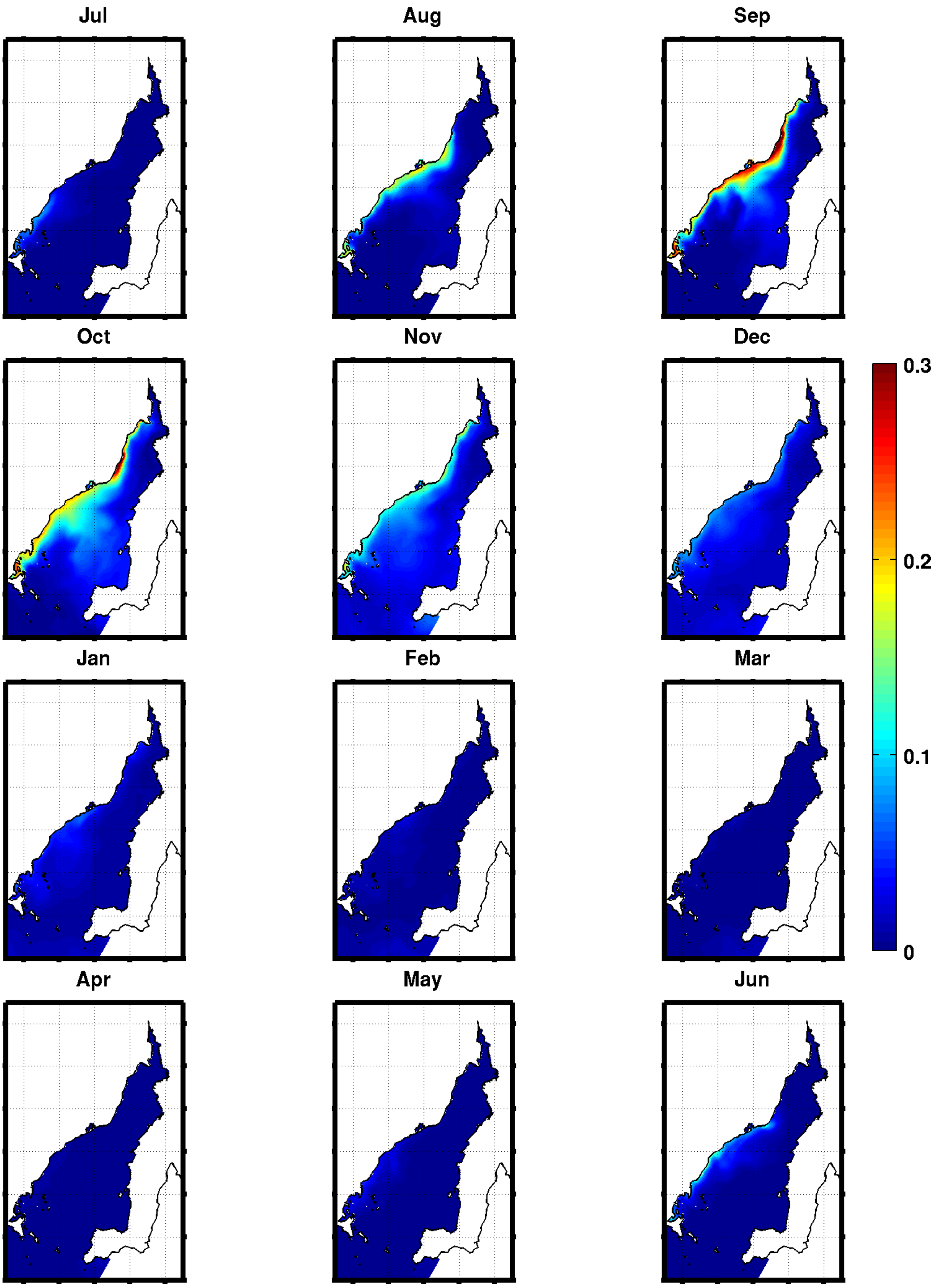


Figure 2.8 Monthly means of simulated surface zooplankton concentrations (mmol N m^{-3}) for July 2010 to June 2011.

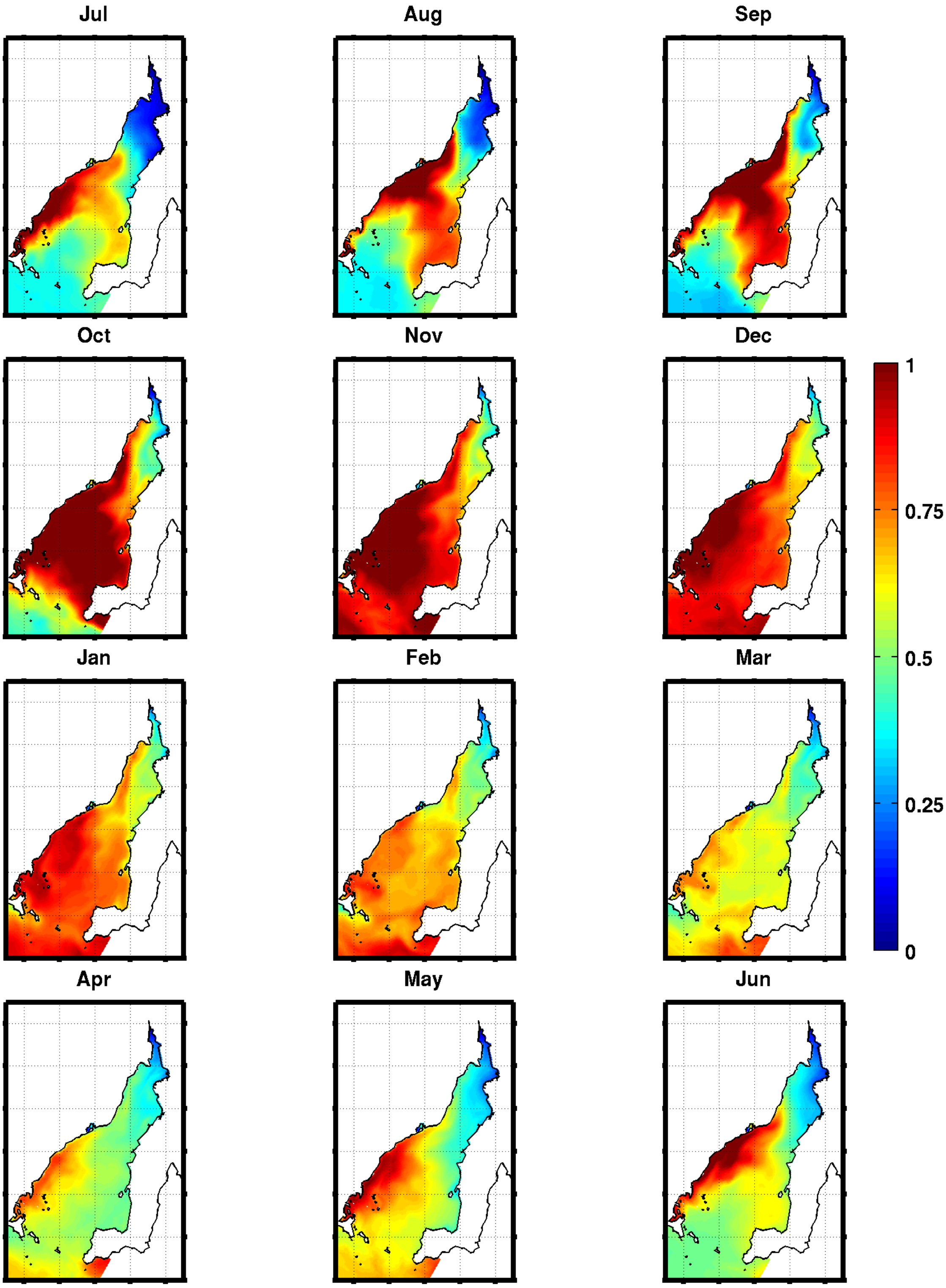


Figure 2.9 Monthly means of simulated surface detritus concentrations (mmol N m^{-3}) for July 2010 to June 2011.

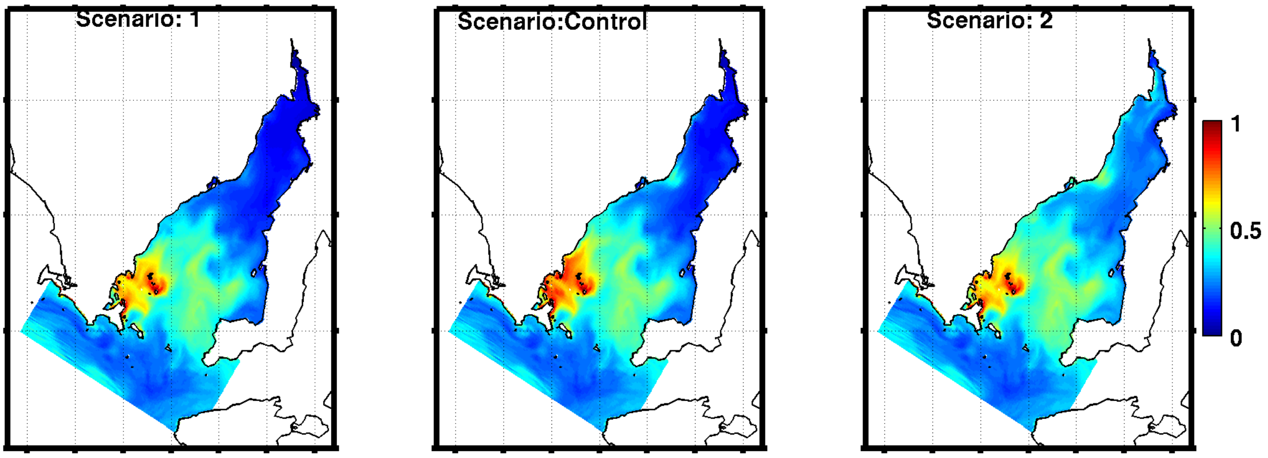


Figure 2.10 Snapshot of the daily-averaged simulated surface phytoplankton concentrations (mmol N m^{-3}) on 1 June 2011 under scenario studies with varying loads of anthropogenic nutrients. Scenario 1 (left) includes no anthropogenic nutrient sources, Control scenario (middle) includes anthropogenic nutrients from finfish aquaculture and other sources for the 2010/11 year, and Scenario 2 (right) includes an increase in the finfish aquaculture related loads. See Table 2.1 for a description of the total anthropogenic loads and their sources for each scenario study.

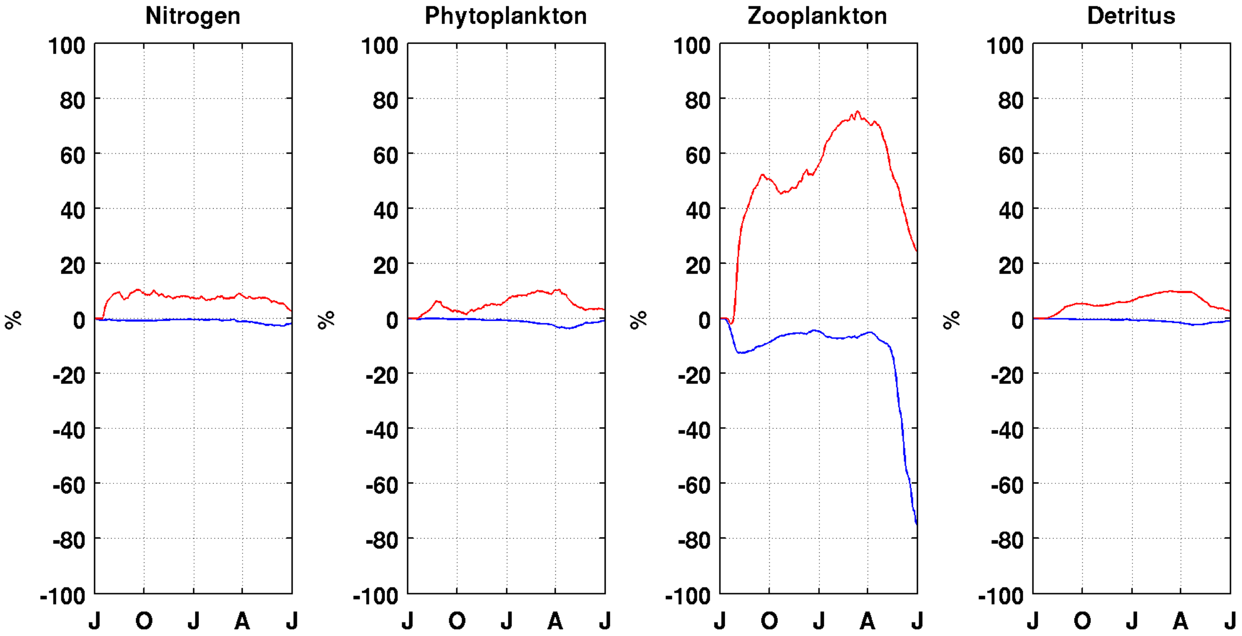


Figure 2.11 Scenario study responses, expressed as the daily percentage change, in the total model inventory of dissolved nitrogen, phytoplankton, zooplankton and detritus to a reduction (Scenario 1; blue line) and increase (Scenario 2; red line) in anthropogenic nutrients in comparison to the control simulation.

The management of marine resources in Spencer Gulf are influenced by complex interactions between physical, chemical and biological processes. While the level of human disturbances (e.g. anthropogenic nutrients) can be controlled, many environmental processes (e.g. natural flushing and associated nutrient delivery from the shelf) cannot. This chapter demonstrates how the development of a validated, advanced three-dimensional ocean-ecosystem model can be applied to enhance our understanding of the interaction of natural ecosystem processes and human activities. The demonstrated changes in anthropogenic nutrient loads from finfish aquaculture (and other sources) are clearly shown to have influence on the lower trophic ecosystem and associated water quality (e.g. nutrient and biomass concentrations) in Spencer Gulf. While previous work (Doubell et al. 2013) has shown a good level of quantitative agreement between the modelled control scenario and field observations, continual development and refinement of the model, such as the inclusion of co-limiting nutrients (van Ruth and Doubell 2013) and harmful algae (van Ruth et al. 2009), is required. Outputs for the alternative scenario studies should be considered as indicative only at this stage.

As will be demonstrated in the following chapters, outputs from this fundamental model of the base physical and biological systems can be used to drive a variety of other models to assess the potential impact of natural and anthropogenic changes on, water quality, habitat, fisheries and aquaculture, and the ecosystem as a whole. Output from the model presented here does not, yet, include two-way coupling with the other benthic and higher trophic ecosystem models presented in subsequent chapters. In particular, despite the fact that the zooplankton community provides a critical link between biochemistry and fisheries, the two-way coupling of zooplankton in linking phytoplankton and fish is ignored. Whilst, recent work has demonstrated how small changes in describing zooplankton in models may influence higher trophic dynamics (Mitra et al. in press), the future integration of biogeochemical and ecosystem models through zooplankton requires further empirical studies of zooplankton and related covariates (e.g. predator, prey, nutrients) in order to enhance our understanding and modelling of zooplankton interactions with other trophic levels (Mitra et al. in press). Such work remains a fundamental challenge to the development and provision of true ‘end-to-end’ models for quantitative ecosystem-based management.

Notwithstanding these limitations, the continued development and integration of a range of models will provide an ecologically holistic framework and decision support tool to address, with increasing confidence, natural resource management across multiple users dependent on a unique marine system

3 Modelling seagrass and macroalgae distributions within Spencer Gulf

3.1 Introduction

Benthic primary production by seagrasses and macroalgae is a key input for the development of an ecosystem model for Spencer Gulf. However, comprehensive benthic habitat mapping for Spencer Gulf is currently restricted to inshore areas where aerial surveys and field validation are more feasible than other areas (Fig. 3.1, Department for Environment and Heritage 2007). This chapter describes how, given these data limitations, estimates of habitat area and primary production for the Spencer Gulf ecosystem (SGE) area (see Fig. 1.1) were derived for seagrasses and macroalgae to inform the ecosystem modelling components.

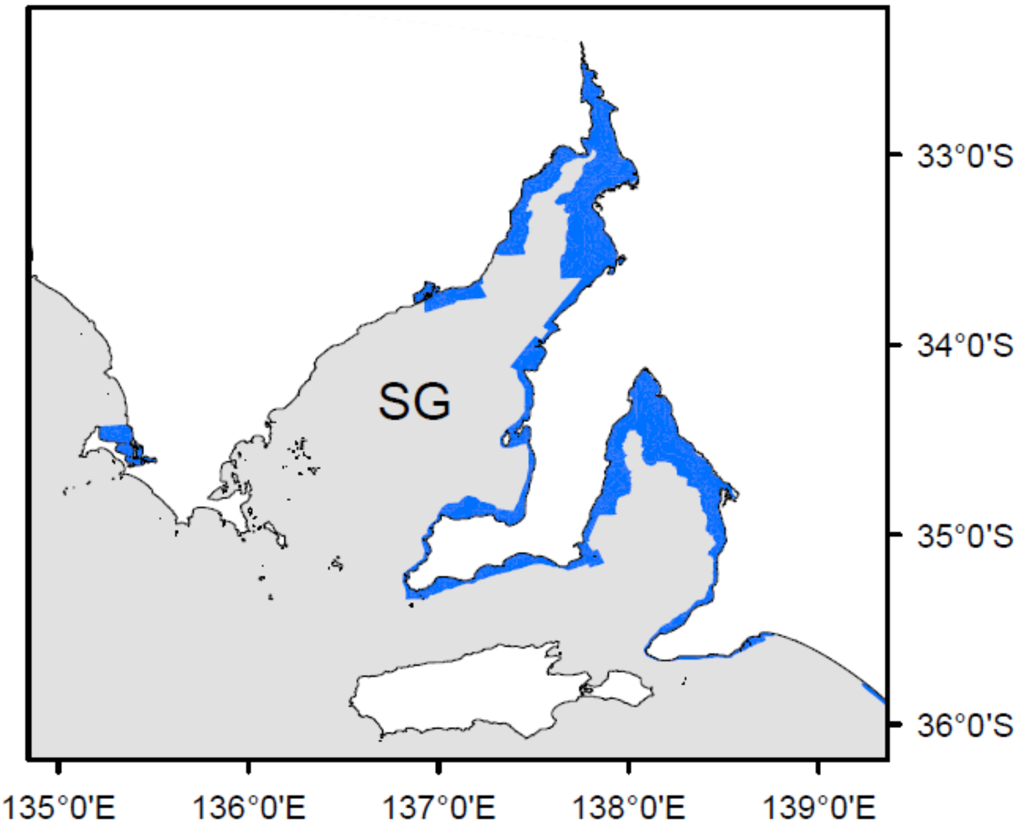


Figure 3.1 The spatial extent of comprehensive marine benthic mapping for South Australia (blue regions), illustrating that benthic habitat information is currently unavailable for much of Spencer Gulf (SG).

Statistical distribution models are one of the primary tools used to investigate the abiotic and biotic drivers of species distributions and to predict the presence or absence of species in new, unsurveyed locations. Here we employ a model selection procedure to test candidate distribution models (i.e. models including different predictor variables) and evaluate their ability to account for spatial patterns in the percentage cover of seagrasses and macroalgae within the surveyed regions of Spencer Gulf. We then use the top-performing distribution models to predict seagrass and macroalgae cover throughout the unsurveyed regions and subsequently to generate estimates of total habitat area and productivity for these groups. Finally, we discuss the key assumptions underlying these estimates and identify important knowledge gaps where additional data could be gathered to assist with refining these estimates further.

3.2 Methods

3.2.1 Habitat cover and putative environmental covariates

The percentage cover of seagrass (levels: 0, 30, 50, 70, 90, 100 %) and macroalgae occurrence (levels: absent, present) were sourced from the South Australian State Benthic Habitats layer (Fig. 3.1, Department for Environment and Heritage 2007). No attempt was made to differentiate seagrass or macroalgae species as this information was not available in the State Benthic Habitats layer, nor was species-specific information required for the ecosystem modelling. We then collated spatial data for the following suite of environmental covariates that could influence the distribution or cover of the benthic habitats and were available for the entire Spencer Gulf:

1. Seabed *depth* (m) sourced from a c. 250 m resolution digital elevation model (DEM) provided by Geoscience Australia (Webster and Petkovic 2005). This DEM lacked sufficient detailed bathymetric information for nearshore benthic areas and as such was merged with topographic data derived from navigation charts previously mapped by the Department of Water, Environment and Natural Resources (DEWNR), South Australia.
2. Seabed *temperature* (°C) and *salinity* (psu) derived from SARDI's ROMS/NPZD oceanographic model for Spencer Gulf (hereafter the SGM) (see section 2). Since north-south gradients in these variables reverse from winter to summer, monthly averages were calculated for the two most extreme months (March and August).
3. Depth-averaged *detritus content* (millimole Nitrogen m^{-3}), NO_3 (millimole m^{-3}) and NH_4 (millimole m^{-3}) derived from the SGM (see section 2). These variables might be expected to influence light penetration to benthic primary producers either directly (*detritus content*) or indirectly by stimulating phytoplankton or epiphytic growth (NH_4 and NO_3).
4. Seabed *current velocity* (m s^{-1}) and *wind-induced orbital velocity* (m s^{-1}) derived from the SGM (see section 2). These variables provide an index of the physical hydrodynamic stress experienced on the benthos.
5. The predicted occurrence of *rocky reef habitat* from a c. 250 m resolution spatial layer developed by Watts et al. (2011).

The response data (habitat cover) and these spatial covariates were collated across a common 0.0025° (c. 250 m) grid. The spatial data were prepared using the *R* computing environment (R Development Core Team 2011) and functions within the *R* package raster (Hijmans and van Etten 2012). To upscale (downscale) raster datasets, layers were first aggregated (disaggregated) to a 0.0025° grid and then resampled to a common extent using nearest neighbour assignments (discrete variables) or bilinear interpolation (continuous variables).

3.2.2 Statistical distribution models

We fit statistical distribution models as a two-stage process, and evaluated candidate model performance at each step. First, we modelled the occurrence (i.e. presence or absence) of seagrass and macroalgae using separate autologistic regressions for each habitat group. Autologistic regression extends standard logistic regression analysis by accounting for spatial autocorrelation in a binary response that can bias the estimation of regression coefficients, specifically by adding a distance-weighted function of neighbouring response values (i.e. the 'spatial autocovariate') to the logistic model (Dormann et al. 2007). Second, since detailed information on the percentage cover of seagrass was available, we evaluated models of seagrass cover conditional on seagrass presence (i.e. a hurdle modelling approach). Positive seagrass percentage cover categories (30, 50, 70, 90, 100 %) represent an ordinal, bounded response for which classical assumptions of constant variance and linearity are not justifiable. Consequently, we used proportional odds logistic regression (POLR) to model positive seagrass cover conditional on seagrass presence in this second stage.

3.2.3 Model selection

A candidate set of statistical models (incorporating interactions and polynomial terms) was evaluated for each stage detailed above. Covariates for which Pearson's correlation coefficient exceeded 0.75 were not included as predictors in the same model. We quantified the performance of candidate distribution models both in terms of model fit (i.e. the ability of models to account for variation in percentage cover for a training dataset), as well as in terms of out-of-sample validation (i.e. the ability of models to predict correctly for a hold-out validation dataset). Data frames for model training and validation were produced by extracting response and covariate data from the 0.0025° resolution spatial layers using 0.02° sampling grids (offset by 0.01°), resulting in c. 4000 sites for each dataset.

3.2.4 Habitat predictions and benthic productivity estimates for Spencer Gulf.

We used the top-performing autologistic models to classify the occurrence of seagrass and macroalgae habitats for the unsurveyed regions of the Spencer Gulf study region. For areas predicted to have seagrass cover, the actual percent cover was predicted as a second step using the selected POLR model.

3.2.5 Biomass and productivity estimates for benthic primary producers

Spencer Gulf seagrass communities are dominated by *Posidonia* species (Kirkman 1997), so the literature on *Posidonia* seagrass meadows was reviewed to derive estimates of seagrass biomass and productivity. *Posidonia* biomass varies with depth, season and species (Cambridge and Hocking 1997). However, the total (above- and below-ground) biomass of dense *Posidonia* communities has been estimated at ~1.05 (Paling and McComb 2000) and ~1.5 kg dry wt m^{-2} (Collier et al. 2007). These two estimates were averaged to give 1.275 kg dry wt m^{-2} (or 1 275 t dry wt km^{-2}) which was assumed to represent the biomass density for regions of 100 % seagrass cover in Spencer Gulf. To estimate seagrass productivity, below- and above-ground productivity for *Posidonia* species were first estimated as 2.2 (Paling and McComb 2000) and 0.9 kg dry wt m^{-2} yr^{-1} (West and Larkum 1979, Cambridge and Hocking 1997), respectively. Total seagrass productivity for 100 % seagrass cover was therefore estimated as 3.1 kg dry wt m^{-2} yr^{-1} (or 3 100 t dry wt km^{-2} yr^{-1}).

Biomass and productivity estimates for macroalgae were derived from studies of *Ecklonia radiata*, a habitat-forming macrophyte in Spencer Gulf. Using the mean depth of surveyed macroalgae habitat for the Gulf (10 m), a density of approximately 13 *E. radiata* m^{-2} was assumed for these habitats on the basis of Kirkman's (1989) depth-density relationship for the species. The mean dry weight of a single plant is 99 g (Larkum 1986), leading to an estimated biomass of 1.29 kg dry wt m^{-2} (or 1 290 t dry wt km^{-2}). A review of productivity estimates for *Ecklonia radiata* in shallow temperate waters provided an average value of 3.58 kg dry wt m^{-2} (or 3 586 t dry wt km^{-2}) (Novaczek 1984, Larkum 1986, Kirkman 1989).

Using these estimates of biomass density and productivity for seagrass and macroalgae, and assuming that dry weight for these groups equates to 10 % of wet weight (Duarte and Kirkman 2001), we derived estimates of total biomass for seagrasses and macroalgae within the SGE. Further, a ratio of annual productivity to biomass was derived for both groups to inform the ecosystem modelling.

3.3 Results and Discussion

3.3.1 Statistical distribution models

Statistical models for seagrass occurrence, and for seagrass cover conditional on seagrass presence, suggested the importance of both depth and a seabed current velocity on seagrasses within the SGE (Table 3.1). The selected autologistic model for seagrass occurrence (depth + $depth^2$ + current + $current^2$) performed well when tested against the out-of-sample validation dataset (81 % of cases were correctly classified; Table 3.1). This model was therefore used to predict the probability of seagrass occurrence across the SGE (Fig. 3.2a). For those grid cells classified as seagrass habitat in this first step, we used the selected POLR model (also depth + $depth^2$ + current + $current^2$) to predict the percentage cover of seagrass. Finally, these predictions were merged with data from the South Australian State Benthic Habitats layer to produce final estimates of seagrass occurrence and cover (Fig. 3.2b). Interestingly, we found no statistical evidence to suggest that high nutrient levels negatively impact seagrass distribution and cover (Table 3.1). In contrast, empirical evidence suggests that high nutrient loads in the ocean can suppress seagrasses by stimulating epiphytic growth. However, we note that there is little spatial variation in simulated NO_3 and NH_4 for Spencer Gulf and that currently these simulated outputs do not include nutrient inputs due to aquaculture.

Table 3.1 The performance of candidate statistical models of seagrass occurrence (a) and seagrass cover (b). In both cases, models are ordered by the Akaike's Information Criterion (AIC) for which lower numbers indicate better performing models. The predictive performance of each model was also evaluated against an out-of-sample validation dataset. These validation metrics are reported as follows: (a) for binomial occurrence models, the percentage of correctly classified cases (PCC); and (b) for the proportional odds logistic regression (POLR) models, Somer's D which measures the strength of association between two ordinal values (higher values indicate more accurate predictions). The acronym WIBOV denotes simulated wave-induced orbital velocity at the seabed.

(a) Seagrass occurrence (binomial regression)			(b) Seagrass cover where present (POLR)		
Model	**AIC**	**PCC**	**Model**	**AIC**	**Somer's D**
depth + $depth^2$ + current + $current^2$	2768	81.0	depth + $depth^2$ + current + $current^2$	7853	0.407
depth + $depth^2$ + tempMarch	3053	79.2	depth + $depth^2$ + WIBOV	7869	0.342
depth + $depth^2$ + NO_3	3062	79.7	depth + $depth^2$ + detritus	7941	0.324
depth + $depth^2$ + current	3120	79.4	depth + $depth^2$+salinityAugust	7950	0.322
depth + $depth^2$ + tempAugust	3175	79.2	depth + $depth^2$ + salinityMarch	7983	0.324
depth + $depth^2$ + detritus	3177	78.6	depth + $depth^2$+NH_4	7986	0.328
depth + $depth^2$ + salinityAugust	3181	78.8	depth + $depth^2$ + current	7997	0.319
depth + $depth^2$ + salinityMarch	3183	78.4	depth + $depth^2$ + tempMarch	8030	0.329
depth + $depth^2$ + NH_4	3227	78.8	depth + $depth^2$ +NO_3	8110	0.338
depth + $depth^2$ + WIBOV	3285	79.4	depth + $depth^2$	8112	0.320
depth + $depth^2$	3300	79.9	depth + $depth^2$ +tempAugust	8112	0.330
depth	3418	77.0	depth	8182	0.234
1	3424	77.3	1	8526	0.000

Table 3.2 Biomass and productivity estimates for seagrass and macroalgae habitats in the Spencer Gulf Ecosystem area (SGE).

	Seagrass	Macroalgae
SGE area (km^2)	28974	28974
Habitat area (km^2)	5064[a]	271
Habitat fraction	0.175	0.009
Total Biomass (wet mass, t)	64,560,000	3,500,000
Biomass per unit habitat area ($t\ km^{-2}$)	12,749	12,915
Production per unit habitat area ($t\ km^{-2}\ y^{-1}$)	31,000	35,862
P/B	2.432	2.777

[a] Area for seagrass is given as 100% seagrass cover equivalents

Attempts to identify spatial drivers of macroalgae occurrence within the SGE were unsuccessful. Evaluation of autologistic models of macroalage occurrence indicated that the null model (i.e. a model including the spatial autocovariate only) performed similarly to models that additionally included environmental covariates. Further, coefficient estimates for some covariates were not biologically realistic (e.g. a positive relationship between depth and the probability of macroalgae presence was estimated). Unfortunately, while it was clear from the South Australian State Benthic Habitats layer that all macroalgae-dominated habitat occurs on rocky reefs, there are currently no spatial data available on the occurrence of rocky reef habitat for much of the SGE. Our attempts to use predicted rocky reef occurrence (Watts et al. 2011) as a proxy for empirical data on reef occurrence were unsuccessful; in fact, of the total number of grid cells classified as reefs by the State Benthic Habitats layer, only 13% were correctly identified as reefs by the predicted reefs layer. We therefore concluded that statistical models of macroalgae occurrence could not be extrapolated to the unsurveyed regions of the SGE with any confidence at this time. Consequently, we assumed that the area of macroalgae habitat within the SGE was equal to the currently mapped area of macroalgae habitat.

3.3.2 Estimates of biomass and productivity for each habitat

The final estimates for seagrass and macroalgae biomass and productivity, which were derived from estimated seagrass and macroalgae habitat areas and empirical data for *Posidonia* species (seagrasses) and *Ecklonia radiata* (macroalgae), are detailed in Table 3.2 above.

3.3.3 Key assumptions

The estimates provided in Table 3.2 rely on a number of assumptions that should be made explicit. The statistical distribution and percent cover models for seagrasses rely on spatial seagrass data from the South Australian State Benthic Habitats layer, which includes data collected over 2005 and 2006. There is no information in this dataset regarding benthic habitat changes over seasonal or longer time scales. Further, no attempt was made to model individual seagrass species. Although some limited species-specific point location data for seagrasses are available from video surveys, the State Benthic Habitats layer provides far more comprehensive spatial coverage. Predictions of seagrass cover for the unsurveyed regions of the SGE required extrapolation of the statistical models which assumes fitted relationship are extendable to new covariate values. Further, although occurrence of seagrasses is associated with certain substratum characteristics (i.e. soft sediment), substratum type was not used when fitting or extrapolating the statistical models because it is unavailable for the unsurveyed regions. Finally, we assumed that the area of macroalgae habitat within the SGE equals the currently mapped area of this habitat, because attempts to construct reliable distribution models for macroalgae were unsuccessful. As a result, our estimate of total macroalgae habitat area is probably an underestimate.

Our estimates of biomass and productivity for seagrass and macroalgae habitats within the SGE are simplistic in that empirical data for *Posidonia* seagrass species and the macroalgae *Ecklonia radiata* are used as surrogates for entire seagrass and macroalgae communities. Further, productivity estimates are based on empirical estimates of biomass accumulation. These may suffer from negative bias because they neglect production lost to processes such as leaf erosion and herbivory (Fairhead and Cheshire 2004).

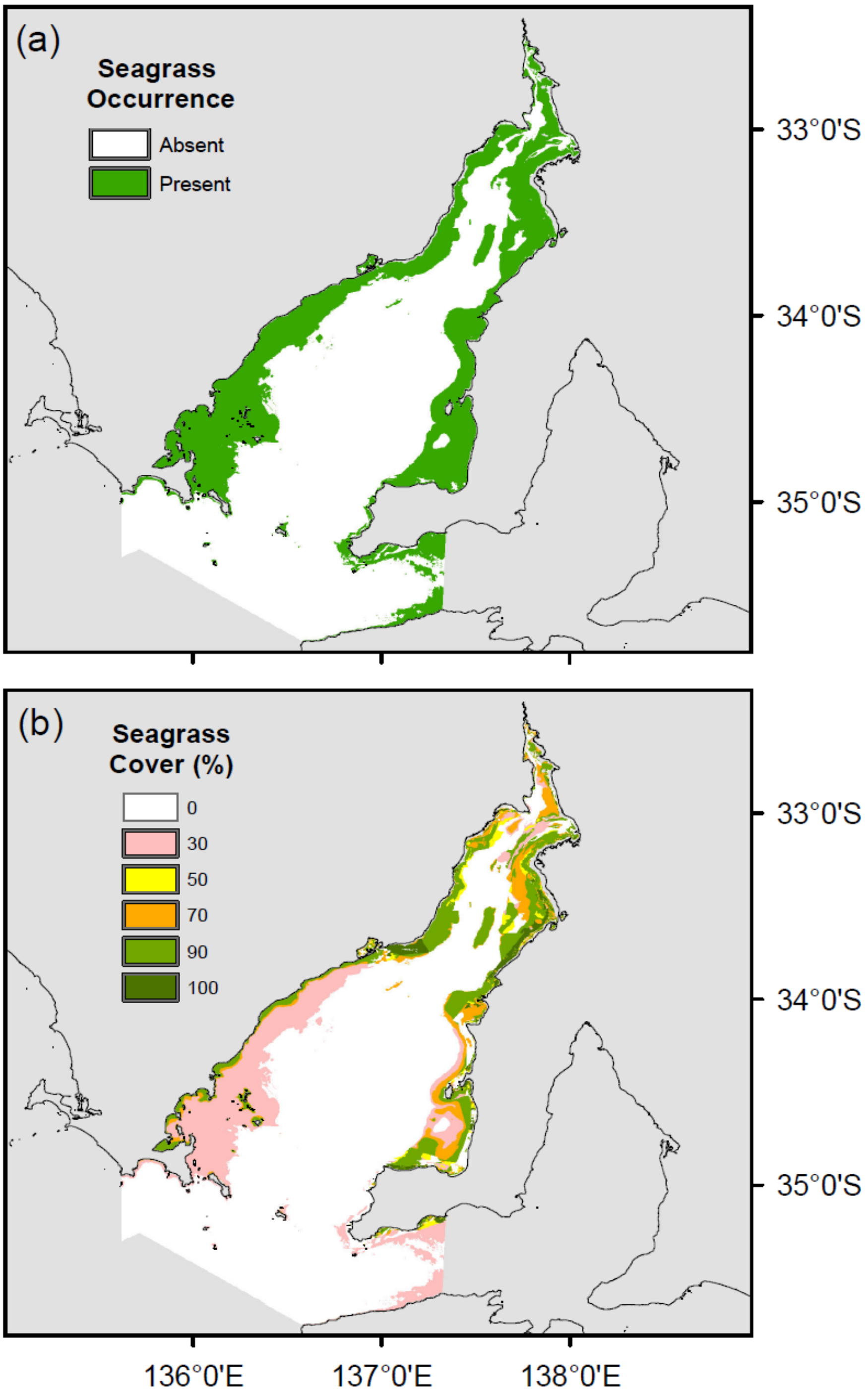


Figure 3.2 Predicted seagrass occurrence and cover within the Spencer Gulf Ecosystem area. (a) Predicted seagrass occurrence generated using the selected autologistic model. (b) Predicted percentage cover of seagrasses conditional on seagrass presence, generated using the selected proportional odds logistic regression model. In this panel, predictions have been merged with empirical data from the South Australian Marine Benthic Habitats layer. Grey regions represent land or marine regions outside the Spencer Gulf Ecosystem area.

4 Development of a Spencer Gulf ecosystem model for fisheries and aquaculture

4.1 Introduction

Spencer Gulf represents a multiple use system with important wildcatch and aquaculture fisheries, conservation and recreation areas and increasingly mining ventures and associated infrastructure, such as port developments. These activities provide fertile ground for ongoing policy and natural resource management issues. To help mitigate and manage these challenges and ensure future development is ecologically sustainable, an independent and credible decision support system to enable evidence-based assessment of development options, with full consideration of social and economic benefits and cumulative environmental implications needs to be developed. A key element to realising this objective is the development of ecological models to understand the key drivers and sensitivities in the Spencer Gulf ecosystem and to provide a means to resolve and attribute potential impacts to the ecosystem from multiple human stressors and environmental change.

The aim of this study is to develop a Spencer Gulf Ecosystem (SGE) model using the *Ecopath with Ecosim* software, incorporating 20 years of fisheries and fin-fish aquaculture data, and:

(1) Develop a range of ecosystem performance indicators on the state of the marine environment;
(2) Provide capacity to resolve complex dynamic interactions between multiple fisheries and aquaculture industries and attribute their potential impacts on each other and the marine ecosystem; and
(3) Conduct scenario testing including how changes to fisheries and aquaculture production and climate will impact the ecology of Spencer Gulf.

4.2 Methods

4.2.1 Ecopath and mass balance approach

We used the *Ecopath with Ecosim* software (www.Ecopath.org) (software version 6.4) to develop a trophic mass-balance model of the Spencer Gulf Ecosystem (SGE). *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). *Ecopath* enables description of the static state energy flow of an ecosystem at a particular point in time, whereas *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 and Walters 2004, Shannon et al. 2008, Griffiths et al. 2010, Piroddi et al. 2010). For the SGE, an *Ecopath* model was constructed for 1991. Time series data over a 20 year period (1991-2010) were used to develop the *Ecosim* model.

4.2.2 Model area and structure

The model area extends from its western extent at the 100 m contour south of Liguanea Island to Cape Border (Kangaroo Island), including parts of the Investigator Strait to Foul Bay in the east, with the northern extent at Yorkes Crossing in northern Spencer Gulf (Fig. 1.1). The model area was calculated at 28,974 km^2.

A number of functional or trophic groups were developed in the SGE model, based on species similarity in terms of diet, habitat, foraging behaviour, size, consumption and rates of production (Table 4.1). Many commercial species were modelled as separate groups to aid scenario testing/modelling. The SGE model structure was built around 78 functional groups including mammals (4), birds (6), chondrichthyans (9), teleosts (28), cephalopods (4), other invertebrates (18), microbial (2), autotrophic (3), detritus (3) and discard groups (1). A large dietary matrix was developed that included 328 prey taxa categories. Dietary information with high provenance to the SGE was used where available. Key sources of dietary data were Page et al. (2011) and Currie and Sorokin (2010). Guild structure analyses in Currie and Sorokin (2010) also provided a basis for structuring of functional/trophic groups within the model, particularly for fish species. 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 associated with aggregating.

In addition to diet information, there are four key parameters that are required by *Ecopath* to balance a model. These include biomass, production per unit of biomass (P/B, equivalent to the instantaneous rate of total mortality (Z) used by fisheries biologists), consumption per unit of biomass (Q/B) and ecotrophic efficiency (EE, the fraction of the production that is used in the system, i.e. either passed up the food web, used for biomass accumulation, migration or export, and varies between 0 and 1 and can be expected to approach 1 for groups with considerable predation pressure). Values for three of these four parameters need to be estimated, with the final parameter value estimated by the model. 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 available in Appendix B.

Table 4.1 Functional or trophic groups used in the Spencer Gulf ecosystem model. Parameter values are indicated where P/B = production/biomass; Q/B = consumption/biomass; EE = ecotrophic efficiency. Bold values are estimated by the model. DDF = deposit detritovore feeding; DOM = dissolved organic matter; POM = particulate organic matter; SAO = sand-associated omnivore; ZF = zooplankton feeding.

	Group name	Trophic level	Habitat area (fraction)	Biomass in habitat area (t/km^2)	P/B (/year)	Q/B (/year)	EE
1	Australian sea lion	4.93	1.00	0.00636	0.7920	29.4400	**0.0005**
2	New Zealand fur seals	5.00	1.00	0.00870	1.1840	49.8600	**0.0002**
3	Bottlenose dolphin	4.85	1.00	0.00354	0.0800	18.9900	**0.0148**
4	Common dolphin	5.09	1.00	0.03721	0.0900	20.5800	**0.0005**
5	Petrels	4.62	1.00	0.00293	1.0000	191.1800	**0.5087**
6	Australian gannet	5.40	1.00	0.00008	1.0000	124.0000	**0.0000**
7	Little penguin	4.87	1.00	0.00128	1.2900	85.6400	**0.6792**
8	Shags & cormorants	4.46	1.00	0.00020	1.0000	77.4000	**0.0000**
9	Terns	4.84	1.00	0.00002	1.0000	90.6500	**0.9464**
10	Gulls	3.96	1.00	0.00015	1.0000	126.1800	**0.0000**
11	White shark	5.87	1.00	**0.00001**	0.1000	1.7300	0.9500
12	Whaler sharks	5.15	1.00	**0.00397**	0.0950	2.6100	0.9500
13	Smooth hammerhead	5.63	1.00	**0.00082**	0.2100	3.1500	0.9500
14	Common thresher shark	5.00	1.00	**0.00005**	0.2000	2.7800	0.9500
15	Gummy shark	3.69	1.00	0.01849	0.5500	2.6000	**0.6775**
16	School shark	5.15	1.00	0.00595	0.8800	2.5000	**0.7392**
17	Port Jackson shark	4.22	1.00	0.09440	0.2500	1.5200	**0.4982**
18	Other demersal sharks	3.66	1.00	0.03900	0.3510	2.6000	**0.9593**
19	Rays & skates	3.65	1.00	0.35858	0.4180	1.7600	**0.2990**
20	Southern bluefin tuna	5.16	1.00	**0.00034**	0.2000	1.6000	0.9000
21	Yellowtail kingfish	5.22	1.00	**0.00035**	0.2000	2.5000	0.9000
22	Snapper	3.80	1.00	0.20826	0.4930	3.8000	**0.8007**
23	Snook	4.80	1.00	0.04980	0.4110	3.5100	**0.9850**
24	Barracouta	5.22	1.00	**0.32727**	0.4110	3.6400	0.9000

25	Skipjack trevally	3.70	1.00	0.28000	0.4800	4.1700	**0.9931**
26	Medium piscivorous fish	4.48	1.00	0.42000	0.6360	1.5800	**0.9828**
27	Medium echinoderm fish	3.33	1.00	0.01900	0.6250	2.3400	**0.9813**
28	Australian salmon	4.95	1.00	**0.69407**	0.4500	4.7000	0.9000
29	Australian herring	3.83	1.00	**0.31746**	1.6400	6.3200	0.9000
30	King George whiting	3.58	1.00	0.06269	0.5480	2.2900	**0.9723**
31	Garfish	2.94	1.00	0.11500	0.3290	4.7300	**0.9156**
32	Red mullet	3.66	1.00	0.09900	0.7900	2.3600	**0.9848**
33	Silverbelly	3.62	1.00	0.24200	1.1000	4.4000	**0.9590**
34	Medium crustacean fish	3.72	1.00	0.12100	0.5460	2.9700	**0.9933**
35	Medium mollusc fish	3.38	1.00	0.32500	0.8690	2.2600	**0.9997**
36	Small crustacean fish	3.46	1.00	0.51000	1.3150	3.3200	**0.9909**
37	Degens/Rough leatherjacket	3.10	1.00	1.21396	0.9000	2.2600	**0.9208**
38	Small polychaete fish	3.22	1.00	0.35500	0.9920	2.8200	**0.9928**
39	Syngnathids	3.63	1.00	0.00985	1.0000	4.7000	**0.9838**
40	Blue mackerel	4.14	1.00	**1.58066**	0.4900	6.4000	0.9000
41	Jack/yellowtail mackerel	4.24	1.00	**4.25640**	0.5200	5.3700	0.9000
42	Sardine	4.18	1.00	1.30000	1.6000	5.0400	**0.9869**
43	Anchovy	3.97	1.00	2.06950	0.9800	5.7600	**0.9994**
44	Sprats	3.30	1.00	**0.17930**	1.8000	5.7600	0.9000
45	Farmed SBT	2.00	0.15	0.08196	1.6200	11.8700	**0.0000**
46	Farmed yellowtail kingfish	2.00	0.11	0.00100	0.4864	1.1800	**0.0000**
47	Fish larvae	2.82	1.00	**1.88709**	4.0000	20.0000	0.9900
48	Southern calamary	5.05	1.00	0.11000	1.8300	18.2500	**0.9647**
49	Giant cuttlefish	3.72	1.00	0.06500	2.3700	5.8000	**0.9968**
50	Other squids	4.51	1.00	0.24500	1.8000	17.5000	**0.9416**
51	Octopus	3.74	1.00	0.07100	2.3700	7.9000	**0.9419**
52	Rock lobster	2.87	0.50	**0.05122**	0.7300	12.4100	0.9000
53	Western king prawn	2.38	1.00	0.57055	7.5700	37.9000	**0.9986**
54	Blue swimmer crab	2.99	1.00	0.68530	2.8000	8.5000	**0.6753**
55	Sand crab	3.06	1.00	**1.01629**	2.8000	8.5000	0.8000
56	Other large crabs/bugs	2.01	1.00	**23.62436**	2.8000	8.5000	0.8000
57	SAO crustaceans	2.50	1.00	**25.33019**	0.7900	11.3000	0.9000
58	Hebivorous macrobenthos	2.32	1.00	**31.86060**	2.8000	14.0000	0.9000
59	Sand-zoobenthos feeders	2.27	1.00	**105.26040**	0.6500	7.5000	0.9000
60	Greenlip abalone	2.00	1.00	**0.01553**	0.7300	12.4100	0.9000
61	Blacklip abalone	2.00	1.00	**0.00819**	0.7300	12.4100	0.9000
62	Small mobile DDF crustaceans	2.51	1.00	**0.97572**	7.0100	27.1400	0.9000
63	Small mobile ZF crustaceans	3.48	1.00	**39.94473**	1.1200	9.5000	0.9500
64	Polychaetes DDF	2.62	1.00	**7.95070**	1.6000	6.0000	0.9000
65	Sessile epifauna	2.47	1.00	**0.38903**	2.8000	11.8000	0.9000
66	Gelatinous zooplankton	3.38	1.00	0.20000	16.5000	80.0000	**0.1682**
67	Large carn zooplankton	2.95	1.00	**57.86993**	5.0000	32.0000	0.8000
68	Small herb zooplankton	2.03	1.00	**65.83572**	29.5000	55.0000	0.8000
69	Meiofauna	2.56	1.00	**0.82531**	35.0000	125.0000	0.9900
70	Benthic microflora	1.65	1.00	0.50000	3300.0000	16000.0000	**0.8818**
71	Planktonic microflora	1.62	1.00	**3.01186**	571.0000	1142.0000	0.9900
72	Macroalgae	1.00	0.06	12915.1	2.7800		**0.1603**
73	Seagrass	1.00	0.18	12748.8	2.4300		**0.0048**
74	Phytoplankton	1.00	1.00	29.95000	132.4000		**0.8244**
75	Detritus DOM water column	1.00	1.00	20.40000			**0.9941**
76	Detritus POM sediment	1.00	1.00	18.50000			**0.7086**
77	Fish farm feed	1.00	0.22	1.91000			**0.0746**
78	Discards	1.00	1.00	0.44186			N/A

Fishery data on landings, discards and effort were obtained for the SGE region and broken down into 27 fishing fleets (Table 4.2). These all fall within six main fishery management units: the South Australian (SA) sardine (1 fleet), SA Marine Scalefish (19), Spencer Gulf prawn fishery (1), blue crab (2), abalone (2) and rock lobster (1). Annual fishery landings and effort data were obtained for all fleets between 1991 and 2010 (logbook data obtained from 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 or effort (Currie et al. 2009, Fowler et al. 2009, Roberts and Steer 2010). All landed and discarded species were assigned their functional group, and biomasses summed at the functional group level (t km^{-2}). Time series of annual catch and catch per unit-effort (CPUE) were calculated for functional groups, and biomass and fishing mortality (F) estimates were used where available.

Table 4.2 Details of the 27 different fishing fleets examined in the SGE model. PS = purse seine; HN = haul net; DN = dab net; LL = long line; HL = hand line. Other refers to minor fisheries (ocean jackets, octopus, poles/rods, troll line, trot line and cockle rake).

No.	Fleet name	Fishery Management Unit
1	Sardine	SA Sardine Fishery
2	Prawn	Spencer Gulf prawn fishery
3	Aust salmon PS	SA Marine Scalefish Fishery
4	Aust salmon HN	SA Marine Scalefish Fishery
5	Aust herring HN	SA Marine Scalefish Fishery
6	Garfish HN	SA Marine Scalefish Fishery
7	KGW HN	SA Marine Scalefish Fishery
8	Snook HN	SA Marine Scalefish Fishery
9	Other HN	SA Marine Scalefish Fishery
10	Garfish DN	SA Marine Scalefish Fishery
11	Snapper LL	SA Marine Scalefish Fishery
12	Snapper HL	SA Marine Scalefish Fishery
13	KGW HL	SA Marine Scalefish Fishery
14	Other HL	SA Marine Scalefish Fishery
15	Shark gillnet	SA Marine Scalefish Fishery
16	Shark LL	SA Marine Scalefish Fishery
17	Dropline	SA Marine Scalefish Fishery
18	Calamary jig	SA Marine Scalefish Fishery
19	Cuttlefish jig	SA Marine Scalefish Fishery
20	Rock lobster	SA Southern rock lobster Fishery
21	Blue crab trap	Blue crab fishery
22	Sand crab net	Blue crab fishery
23	Ocean jacket trap	SA Marine Scalefish Fishery
24	Small mesh gillnet	SA Marine Scalefish Fishery
25	Greenlip abalone	Abalone fishery
26	Blacklip abalone	Abalone fishery
27	Other	Minor fisheries (see caption)

The SGE model also included two finfish aquaculture industries (southern bluefin tuna (SBT) and farmed yellowtail kingfish (YTK)). These were modelled differently to most other functional groups because the fish supplied for the cages are sourced from outside of the SGE model area, and during their grow-out period are sustained on feed (whole fish or pellets). Although some of this feed (sardines fed to SBT) is derived from within the SGE, a varying amount is derived from imported feed from outside the SGE. To simplify we kept the sardine harvest and finfish aquaculture feed separate, and designated all aquaculture feed as imported into the SGE model.

4.2.3 Model fitting

Dynamic simulations were run in *Ecosim* using the time-series estimates (1991-2010) of fishing effort, biomass or relative biomass (CPUE) and fishing mortality (F) 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 (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 examine the potential influence of environmental forcing and if incorporating such forcing functions (FF) improved the fit of the model to the time-series data. We used monthly average sea surface temperatures from January 1991 to December 2010 centred on southern Spencer Gulf as a FF. This was used to drive primary production dynamics of all primary producer groups.

4.2.4 Ecosystem indicators

After the model fitting procedure in *Ecosim*, we examined four variables 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 and Taylor 1976, Ainsworth and Pitcher 2006); 3) the mean trophic level of the catch (mTLC) which is calculated as the weighted average of the trophic level (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 mean trophic level of the catch is matched by an ecologically appropriate increase in catch, and conversely for increasing trophic level (Pauly and 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 and Palomares 2005).

4.2.5 Ecospace model

Ecospace is a spatially explicit model that builds on *Ecopath with Ecosim*. It is essentially a spatial version of *Ecosim*, and uses a cell-based format to describe in two dimensions, the spatial distribution of species groups under the influence of biotic and abiotic factors. Inputs can include the movement rates of fauna, vulnerability settings (top-down vs. bottom-up control as per *Ecosim*), habitat preference, spatial distribution of fishing effort and the vulnerability of predators to various habitats (Walters et al. 1999). We developed a preliminary model using bathymetry and habitat data available for Spencer Gulf (see section 3). A 34 x 75 5km cell block grid angled at 27 degrees, with left corner centred on 135.62° E, -32.47 ° S was developed for the Spencer Gulf ecosystem.

4.2.6 Scenario testing

Three main scenarios were examined to provide examples of the kinds of simulations and 'what-if' questions that can be informed by trophodynamic modeling of the SGE. These focused around changes to production in the finfish aquaculture industry, and changes in catches and fishing effort in the two largest volume fisheries in Spencer Gulf, the sardine and western king prawn fisheries. For the finfish (SBT and YTK) aquaculture scenarios, monthly and spatially averaged biomass phytoplankton, zooplankton and detritus (PZD) time-series estimated for Spencer Gulf from the coupled hydrodynamic-biogeochemical model developed to explore different aquaculture loadings (see section 2) (Middleton et al. 2013), were used as forcing functions (FF) in the SGE *Ecosim* model. Three alternate aquaculture loading scenarios were examined: scenario 1 (S1) – no aquaculture nutrient inputs; control scenario (CS) – current (2010/2011) aquaculture nutrient loadings (~1.5 kT/Year); and scenario 2 (S2) – maximum aquaculture loadings (~6.1 kT/year) (see section 2) (Middleton et al. 2013). Mean estimated monthly PZD biomasses under each of these scenarios were used to develop a 60 year time-series where the annual pattern is repetitive (extending to 2050), scaled to the January S1 (no aquaculture) scenario. The PZD time series were then used as a FF for all primary producer groups (excluding seagrass and macroalgae because there were no time series data available for these primary producers); zooplankton groups and detritus groups (including fish farm feed, DOM and POM, but excluding fishery discards), respectively. The SGE *Ecosim* model was run between 1991 and 2050 using the three alternate aquaculture PZD biomass scenarios, and the relative changes in biomass of all modelled groups were examined.

Fishing catch and effort scenarios were modelled by creating times series between 2010 and 2050 and then running these in the SGE *Ecosim* model. For the sardine and Spencer Gulf prawn fisheries, three scenarios (SCS1, SCS2, SCS3) representing a 10%, 20% and 30% increase in target catch from 2010 levels were developed. For the Spencer Gulf prawn fishery, an additional three scenarios (PS1, PS2, PS3) representing a 10%, 20% and 30% increase in fishing effort from 2010 levels were also examined. The effort scenarios (PES1, PES2, PES3, prawn effort scenarios) examined the impact of increased discards, which are significant for the prawn fishery. For all scenarios the relative changes in biomass of all modelled groups were examined.

4.3 Results and Discussion

4.3.1 Trophic structure and flow

The basic parameters used to inform the 78 functional groups within the *Ecopath* model are presented in Table 4.1, 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. There were many adjustments that were required to balance the model. Some of these could be achieved by slight changes to dietary proportions for some functional groups, others required changes to estimated biomass, P/B and Q/B estimates. White sharks required a biomass accumulation rate term to be included in order to balance the model.

The trophic flows between the functional groups in the SGE ecosystem estimated by *Ecopath* are summarised in Figure 4.1. The trophic level of the functional groups ranged from 1 to 5.87, with the highest values for white shark, smooth hammerhead, Australian gannet, barracouta, yellowtail kingfish, southern bluefin tuna, whaler sharks, school shark, common dolphin, New Zealand fur seals and southern calamari (TL >5). Common thresher shark, Australian salmon, Australian sea lion, little penguin, bottlenose dolphin, terns, snook, petrels and other squids had TL ≥ 4.5. (Table 4.1, Fig. 4.1). Shags and cormorants, gulls, demersal sharks, rays and skates, most medium sized fish, small pelagic fishes group (sardines, anchovy, blue mackerel, jack/yellowtail mackerel) and other cephalopods had trophic levels ranging between 3.5 and 4.5. In terms of biomass, the lower part of the Spencer Gulf food web is dominated by crustacean groups, seagrass and macroalgae (Fig. 4.1).

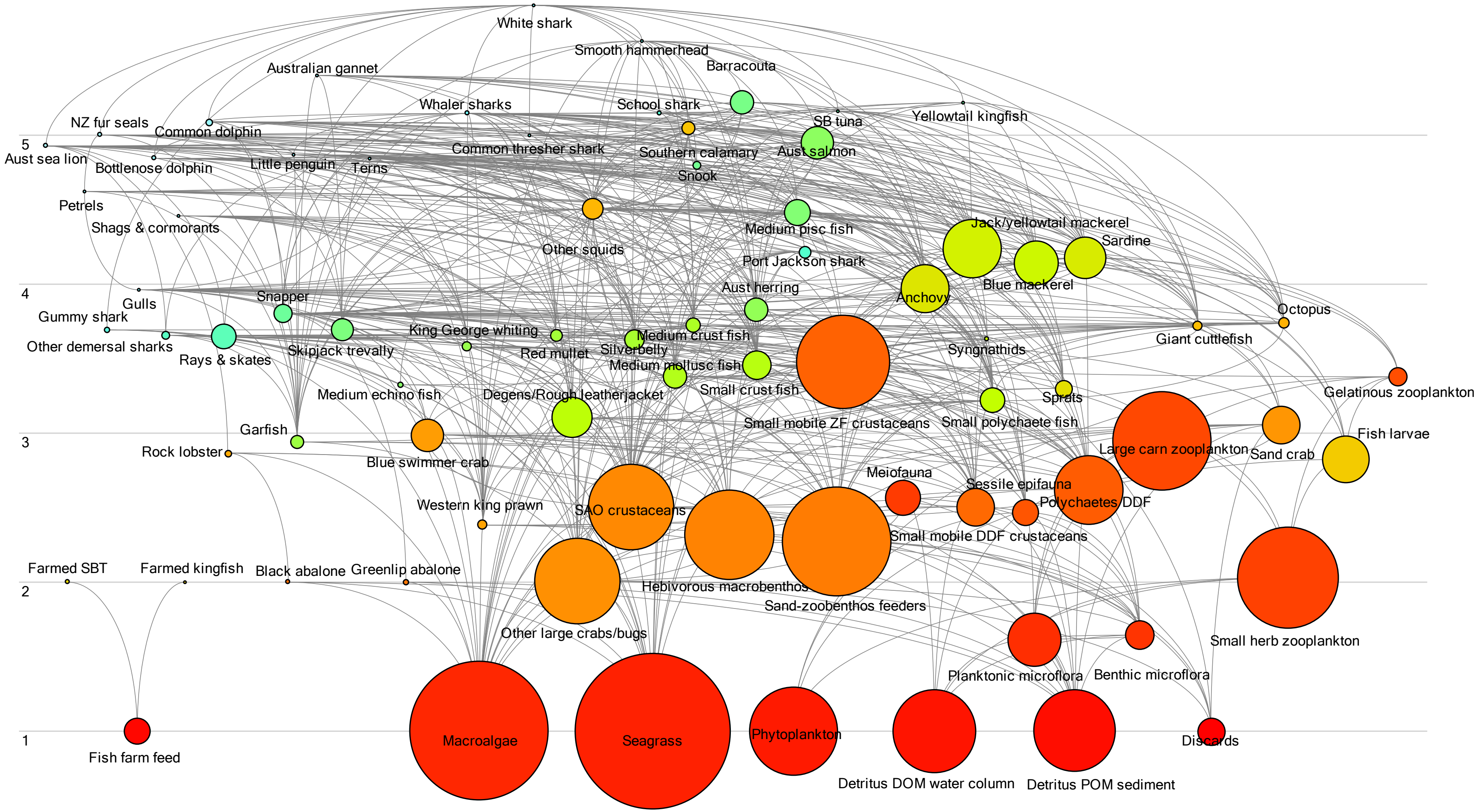


Figure 4.1 Flow diagram expression of trophic flows and trophic levels in the Spencer Gulf Ecosystem. Functional groups are represented by a circle; the size of the circle is proportional to its biomass.

The mixed trophic impacts routine in the network analysis tools within *Ecopath* was used to evaluate critical trophic interactions between groups in the ecosystem (Fig. 4.2). The Leontif matrix visually represents the effects of increasing biomass of one functional group or fishery fleet onto groups and fisheries, and provides a form of sensitivity analyses (Fig. 4.2). Positive impacts are open boxes that extend upwards, and negative impacts are filled boxes extending downwards. The routine is based on the method developed by Ulanowicz and Puccia (1990), and allows the computation of direct and indirect impacts that 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. Cephalopod groups negatively affected a number of fish groups including medium echinoderm specialists, Degens/rough leatherjacket, red mullet, skipjack trevally, barracouta, snapper and snook (Fig. 4.2). Calamary negatively affected sprats, medium piscivorous fish and Australian salmon, while positively affecting medium molluscan fish and silverbelly. Australian herring negatively affected sprats, while Australian salmon negatively affected Australian herring, sliverbelly, medium molluscan fish, blue mackerel and anchovy, and positively affecting sprats. Medium piscivorous fish affected small polychaete fish negatively. Barracouta affected Australasian gannet, smooth hammerhead, snapper, skipjack trevally, medium piscivorous fish, medium echinoderm specialist fish, red mullet and Degens/rough leatherjacket positively; while sardine and other squid groups negatively. Snapper impacted syngnathids, snook, King George whiting, blue swimmer crabs and greenlip abalone negatively. Gummy and school shark impacted whaler sharks, smooth hammerhead and thresher sharks negatively, and bottlenose and common dolphins, southern bluefin tuna and yellowtail kingfish positively. In contrast, whaler sharks impacted bottlenose and common dolphins, southern bluefin tuna and yellowtail kingfish negatively and skipjack trevally positively. White shark impacted fur seals and sea lions negatively, and petrels and little penguins positively. Gulls impacted negatively on terns; common dolphins impacted negatively on southern calamari and skipjack trevally; and New Zealand fur seals impacted negatively on little penguins, petrels, other squid and medium crustacean specialists (Fig. 4.2). Australian sea lions impacted negatively on octopus and medium crustacean specialists. Groups containing commercially targeted species influenced their respective fishing fleets positively, and most fishing fleets negatively impacted their target species (Fig. 4.2). Of the fisheries, the shark gillnet fishery also negatively affected whaler, thresher and smooth hammerhead sharks, and positively affected common and bottlenose dolphin, SBT and yellowtail kingfish. The western king prawn fishery also negatively affected Port Jackson sharks, other demersal sharks and skates and rays. All these sensitivities assume a steady-state system, and do not take into account the changing abundances or diets of groups. Such dynamics are explored in *Ecosim* scenarios below.

4.3.2 Food-web control and model fitting

Following balancing of the SGE Ecopath model, the time series Ecosim model was developed for the period from 1991 to 2010. A key element is the key fishery catch and effort time series data, as well as biomass data available for some predator species (fur seals and sea lions). A total of 110 time series were loaded into the *Ecosim* model (see Appendix C). The model fit to time series of modelled biomass and estimated catch to observed trends for targeted fished species is presented in Figure 4.3. Modelled estimates of biomass tracked those observed through CPUE data reasonably well for most groups (Fig. 4.3). Similarly, estimated catches also tracked actual catches well, although catches of Australian salmon and whaler shark were underestimated by the model, and rock lobster catch was overestimated (Fig. 4.3). Initial Ecosim runs failed to model actual catch of sardines, because catch exceeded biomass in the SGE model area. Including a net immigration rate as other production in the Ecopath model was required in order to balance catch with the available biomass. Similarly, time series biomass and fishing mortality rate data on garfish could not match catch time series data in the model (i.e. not enough biomass to account for catch). This issue was solved by deselecting the biomass and fishing mortality time series and enabling *Ecosim* to estimate these parameters.

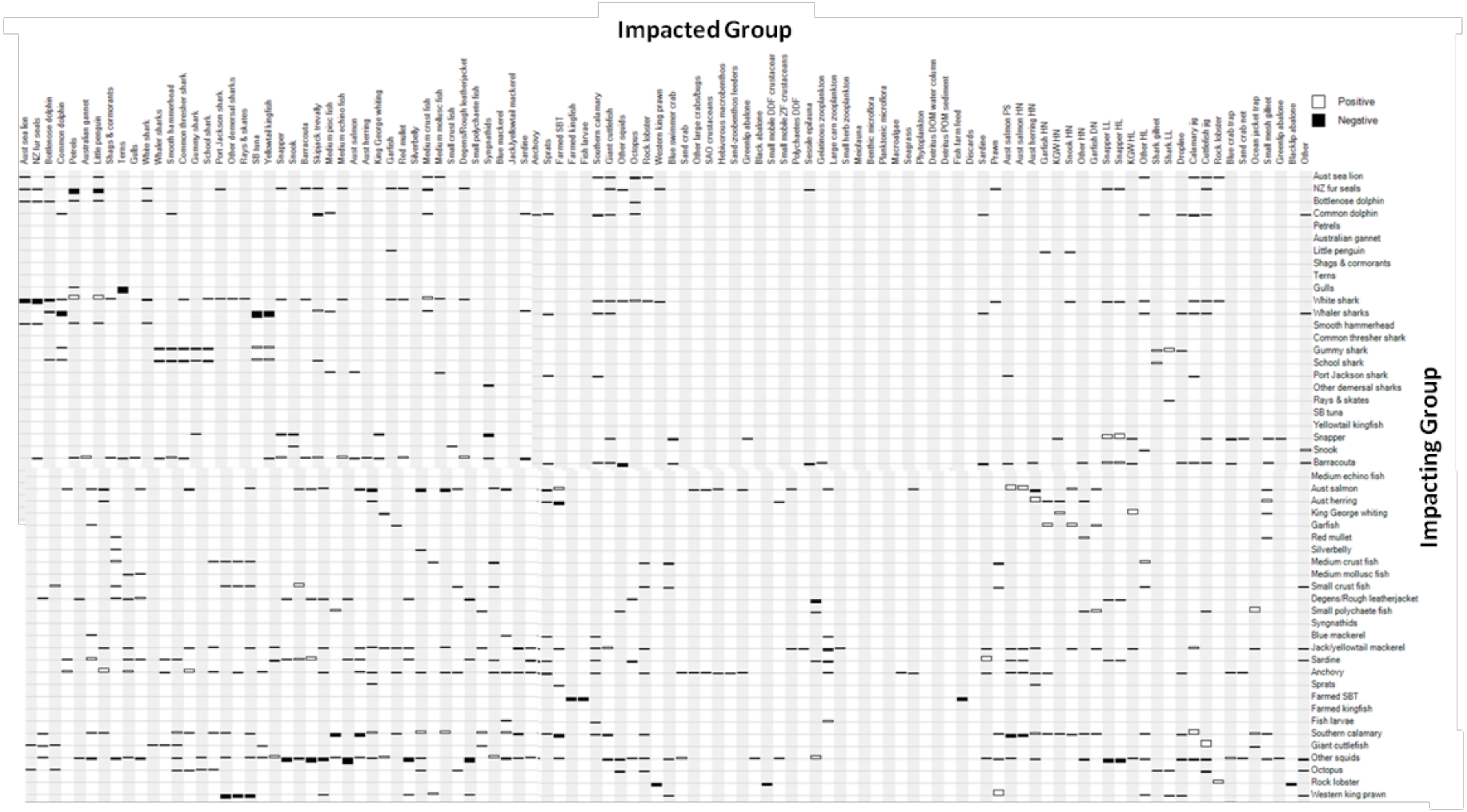


Figure 4.2 Leontif sensitivity matrix showing impacts of increasing abundance of groups on the y-axis on groups on the x-axis. Impacts are expressed as relative % changes, not all impacts are discernible on this figure.

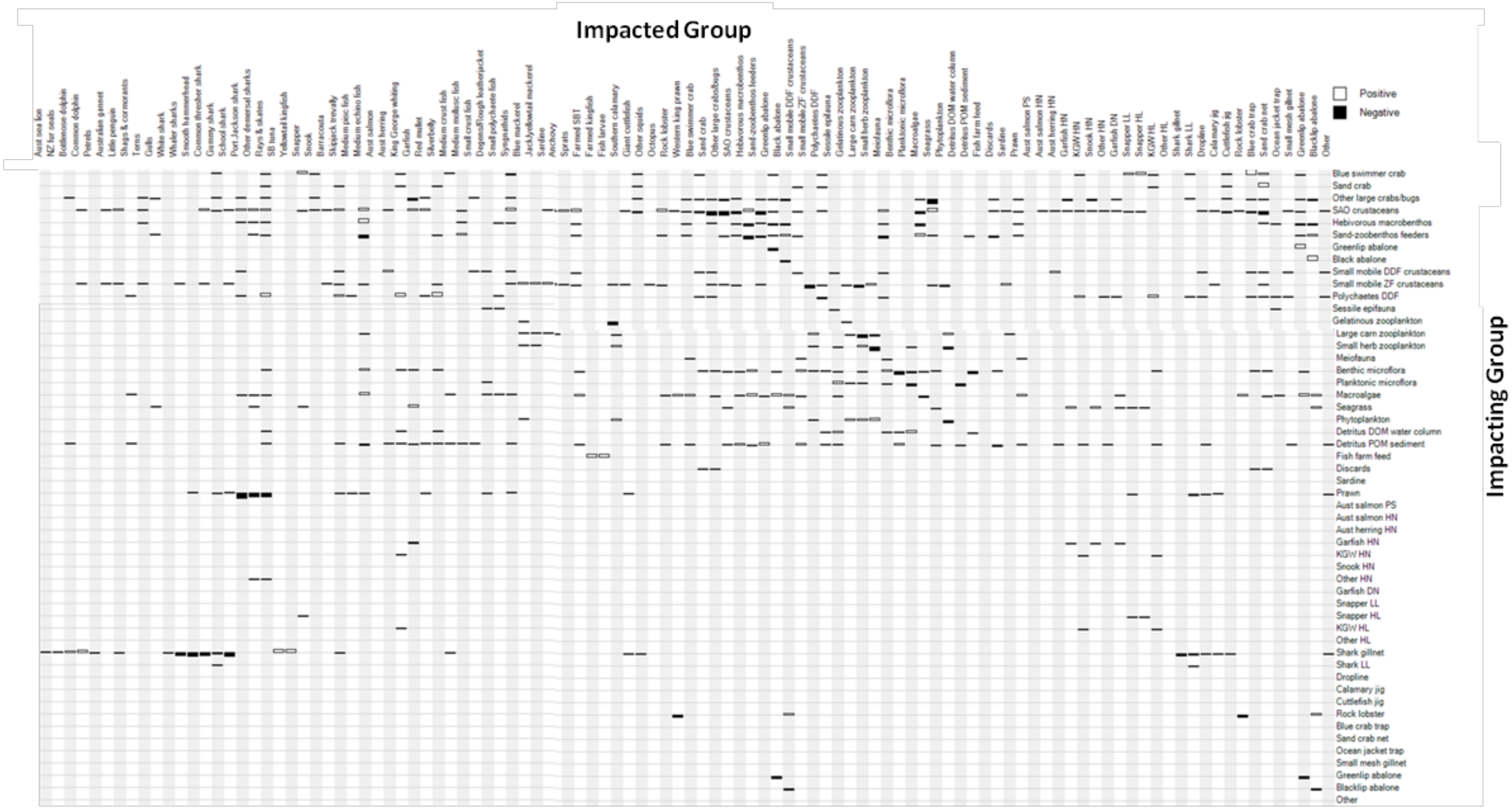
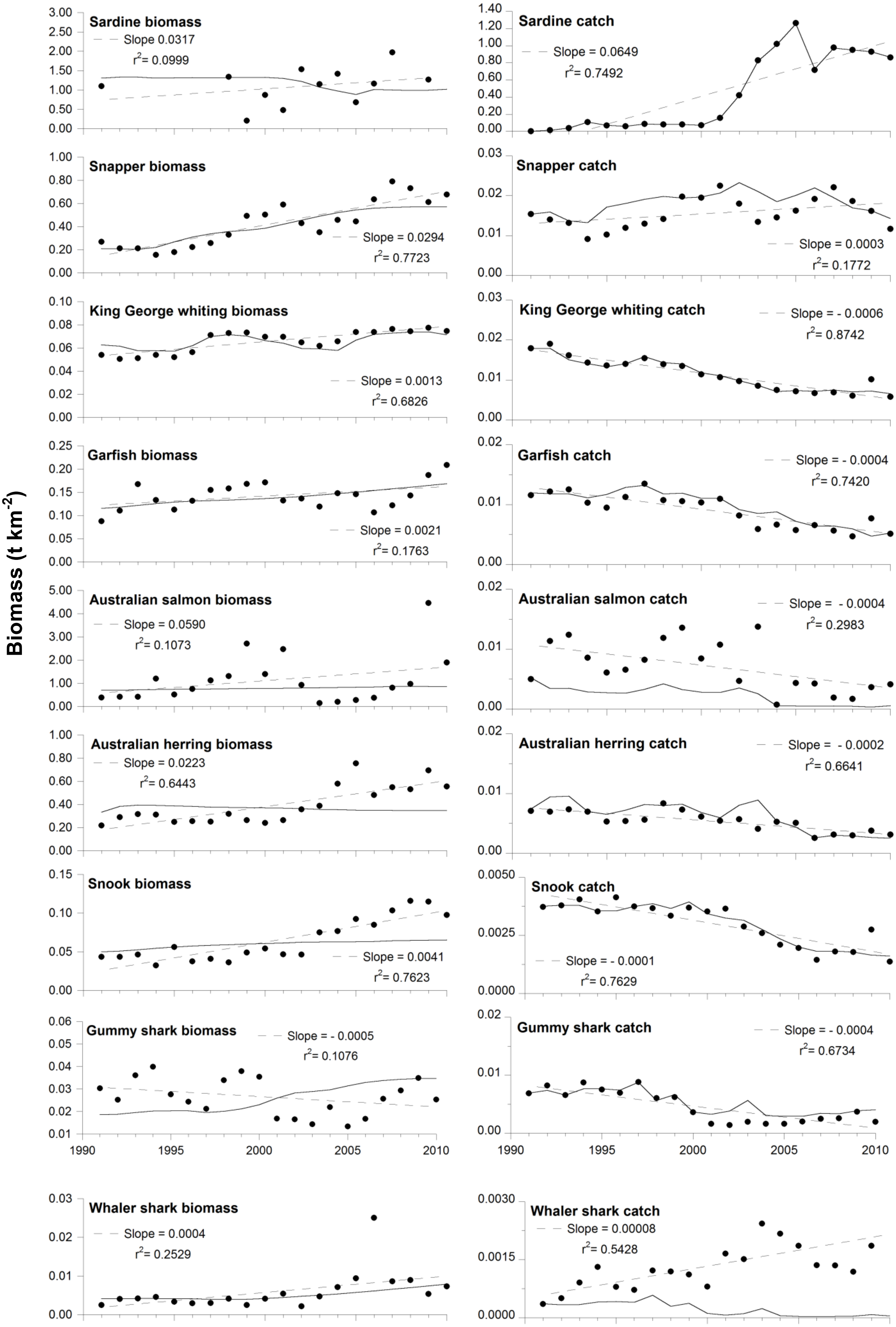


Figure 4.2 continued Leontif sensitivity matrix showing impacts of increasing abundance of groups on the y-axis on groups on the x-axis. Impacts are expressed as relative % changes, not all impacts are discernible on this figure.


Biomass (t km^{-2})
Sardine biomass
Slope 0.0317
r^2= 0.0999
Sardine catch
Slope = 0.0649
r^2= 0.7492
Snapper biomass
Slope = 0.0294
r^2= 0.7723
Snapper catch
Slope = 0.0003
r^2= 0.1772
King George whiting biomass
Slope = 0.0013
r^2= 0.6826
King George whiting catch
Slope = - 0.0006
r^2= 0.8742
Garfish biomass
Slope = 0.0021
r^2= 0.1763
Garfish catch
Slope = - 0.0004
r^2= 0.7420
Australian salmon biomass
Slope = 0.0590
r^2= 0.1073
Australian salmon catch
Slope = - 0.0004
r^2= 0.2983
Australian herring biomass
Slope = 0.0223
r^2= 0.6443
Australian herring catch
Slope = - 0.0002
r^2= 0.6641
Snook biomass
Slope = 0.0041
r^2= 0.7623
Snook catch
Slope = - 0.0001
r^2= 0.7629
Gummy shark biomass
Slope = - 0.0005
r^2= 0.1076
Gummy shark catch
Slope = - 0.0004
r^2= 0.6734
Whaler shark biomass
Slope = 0.0004
r^2= 0.2529
Whaler shark catch
Slope = 0.00008
r^2= 0.5428
1990
1995
2000
2005
2010

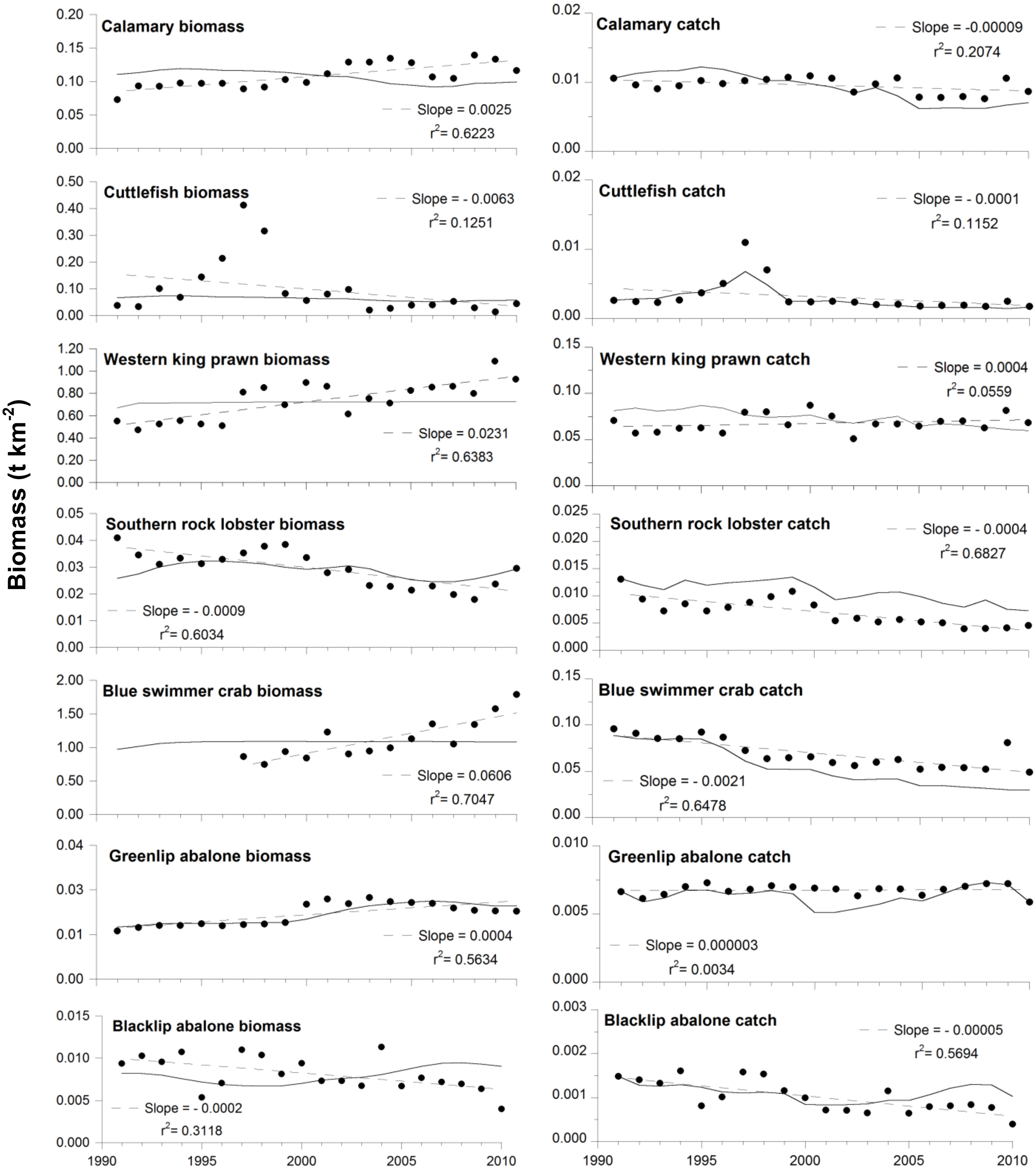


Figure 4.3 Time series fits of the Spencer Gulf *Ecosim* model (thin line) to observed biomass (CPUE) and catch data (dots) for 16 functional groups between 1991 and 2010. The trend lines (dashed) fitted to observed biomass (CPUE) and catch data (dots) are provided, together with values of the slope and coefficient of variation (r^2).

The fit to time series optimisation procedure in *Ecosim* identified 50 predator-prey interactions, and their vulnerability values as the optimal model for minimising the model sum of squares (SS) and producing the best fit to the time series data. Vulnerability parameters describe the nature of the flow control between predator and prey, with values of 2 representing mixed-flow control, values closer to 1 represent bottom-up control while values >2 represent top-down control. The majority (62%) of sensitive interactions were bottom-up, largely represented by key prey taxa exercising bottom-up control on apex predator species (Table 4.3). This included New Zealand fur seals being bottom-up controlled by small crustacean specialist fish; little penguins bottom-up controlled by anchovy; whaler sharks bottom-up controlled by yellowtail kingfish; and smooth hammerhead sharks bottom-up controlled by calamari, giant cuttlefish, other squids, Australian herring, skipjack trevally and snapper (Table 4.3). Common thresher sharks were bottom-up controlled by sardine, anchovy and Australian herring; gummy sharks bottom-up controlled by octopus, western king prawns, blue swimmer crabs and other large crabs/bugs; and school shark bottom-up controlled by medium crustacean specialist fish, jack/yellowtail mackerel, anchovy, southern calamari and octopus (Table 3.3). Snapper were bottom-up controlled by snook, Australian salmon, syngnathids and greenlip abalone; sardine bottom-up controlled by sand associated omnivore (SAO) crustaceans and small mobile zooplankton feeding crustaceans; and anchovy bottom-up controlled by small mobile zooplankton feeding (ZF) crustaceans. SAO crustacean were in-turn bottom-up controlled by macroalgae and particulate organic matter (POM); and sand-zoobenthos feeder bottom-up controlled by POM (Table 4.3). Overall, medium to large fish as prey to apex predators accounted for 10 (32%) of bottom-up controlled interactions; with most other bottom-up control interactions accounted for by small pelagic fish (5, 16%); cephalopods (6, 19%), and large crustaceans (4, 13%) (Table 4.3).

Most of the 19 top-down control interactions (38%) were exercised by seals (8) and snapper (3) (Table 4.3). Australian sea lions exercised top-down control on cephalopods (southern calamari, giant cuttlefish and octopus) and rock lobster. New Zealand fur seals exercised top-down control on petrels, little penguins, barracouta and medium crustacean fish. Snapper exercised top-down control of giant cuttlefish, octopus and blue swimmer crab; Port Jackson shark exercised top-down control of southern calamari (Table 4.3). Australian salmon had top-down control on anchovy (Table 4.3). Blue mackerel had top down control on small mobile zooplankton feeding crustaceans; anchovy had top-down control of SAO crustaceans; southern calamari had top-down control of Australian salmon; other squids has top-down control of barracouta; rock lobster had top-down control of blacklip abalone; and benthic microflora had top-down control of POM (Table 4.3). Bulman et al. (2011) investigated the dynamics of food-web control in the Great Australian Bight (GAB) and Eastern Bass Strait (EBS) ecosystems, based on EwE models developed by Goldsworthy et al. (2011) and Bulman et al. (2006), respectively. Similar to the SGE, they found that these ecosystems were largely bottom-up forced. Both the GAB and EBS models highlighted the importance of fur seals, despite the relatively low biomasses in the ecosystems. In the GAB ecosystem, most interactions between fur seals and their prey were bottom-up controlled (i.e where an increase in fur seal biomass does not particularly influence the biomass of their prey); in contrast, fur seal interactions in the EBS ecosystem were mostly top-down controlled (i.e. where fur seal biomass has a stronger influence on the biomass of their prey) (Bulman et al. 2011). Although most seal interactions in the GAB ecosystem were bottom-up forced, those in the adjacent SGE mostly where top-down controlled (by both New Zealand fur seals and Australian sea lions) (Table 4.3). There is some evidence for wasp-waist control in the SGE, where small pelagic fish dominate their trophic level by biomass, forming a central point between which energy flows from lower to higher trophic levels. In the SGE, 22% of the most sensitive interactions involved small pelagic fish (sardines, anchovy, jack/yellowtail and blue mackerel), although this is at the lower end of the typical 25-50% of interactions for wasp-waist species in classic upwelling systems such as the Benguela Current (Shannon et al. 2008).

Attempts were made to fit a 20-year, monthly mean sea surface temperature (SST) record for southern Spencer Gulf to the *Ecosim* model. This SST-based forcing function (FF) was applied singularly and collectively to all primary producer groups in the model, but all model runs produced poorer fits to the time-series data than when the forcing function was omitted (higher residual sum of squares), and it was not included in the final balanced model. Spencer Gulf is characterised by marked gradients in temperature and salinity, and productivity. Therefore, it is not surprising that fixing a single point in a forcing function as representative for this system did not result in improved model fit. Assessing and applying appropriate forcing functions will be important in the next generation of Spencer Gulf models.

Table 4.3 The 50 most vulnerable predator-prey interactions for the Spencer Gulf Ecosystem model. Pink boxes indicate top-down control and blue boxes bottom-up control. POM = particulate organic matter; SAO = sand-associated omnivore; ZF = zooplankton feeding.

	Predators																			
Prey	Australian sea lion	NZ fur seals	Little penguin	Whaler sharks	Smooth hammerhead	Common thresher shark	Gummy shark	School shark	Port Jackson shark	Snapper	Australian salmon	Blue mackerel	Sardine	Anchovy	Southern calamari	Other squids	Rock lobster	SAO crustaceans	Sand-zoobenthos feeders	Benthic microflora
Petrels		🟥																		
Little penguin		🟥																		
Yellowtail kingfish				🟦																
Snapper					🟦															
Snook										🟦										
Barracouta		🟥			🟦											🟥				
Australian salmon										🟦					🟥					
Australian herring					🟦	🟦														
Medium crustacean fish								🟦												
Small crustacean fish		🟦																		
Syngnathids										🟦										
Jack/yellowtail mackerel								🟦												
Sardine						🟦														
Anchovy			🟦			🟦		🟦			🟥									
Southern calamari	🟥				🟦			🟦	🟥											
Giant cuttlefish	🟥				🟦					🟥										
Other squids					🟦															
Octopus	🟥						🟦	🟦												
Rock lobster	🟥									🟥										
Western king prawn							🟦													
Blue swimmer crab							🟦			🟥										
Other large crabs/bugs							🟦													
SAO crustaceans													🟦	🟥						
Greenlip abalone										🟦							🟥			
Black lipped abalone																	🟥			
Small mobile ZF crustaceans												🟥	🟦	🟦						
Macroalgae																		🟦		
Detritus POM sediment																		🟦	🟦	🟥
No. top-down interactions	4	3	0	0	0	0	0	0	1	3	1	1	0	1	1	1	2	0	0	1
No. bottom-up interactions	0	1	1	1	6	3	4	5	0	4	0	0	2	1	0	0	0	2	1	0

4.3.3 Temporal changes in group biomass

Trends in catches of the 14 main targeted species in fisheries in the Spencer Gulf ecosystem between 1991 and 2010 are presented in Figure 4.4. Although overall catches have increased by ~380% over this period, changes in catches are driven by two main contrasting factors. First, the significant growth of the sardine fishery between 1991 and 2010, has seen it increase from 0% to >80% of the total landed catch in Spencer Gulf, with the major growth period occurring since 2000. Catches in Spencer Gulf peaked at ~36,000 t in 2005 (Fig. 4.4). In contrast, the remaining combined fisheries catches (excluding sardine) have declined over the same period by about 37%. This includes 50-81% declines in catches of King George whiting, garfish, snook, Australian herring, gummy and school shark, southern rock lobster, blue swimmer crab and blacklip abalone (Fig. 4.4). The greatest decline in catches of gummy and school sharks (81%) occurred when this fishery was transferred from South Australian to Commonwealth management in 2000, resulting in major closures in State waters including Spencer Gulf. The only fisheries that have maintained similar catches to 1990s levels are western king prawn, calamari and greenlip abalone (Fig. 4.4).

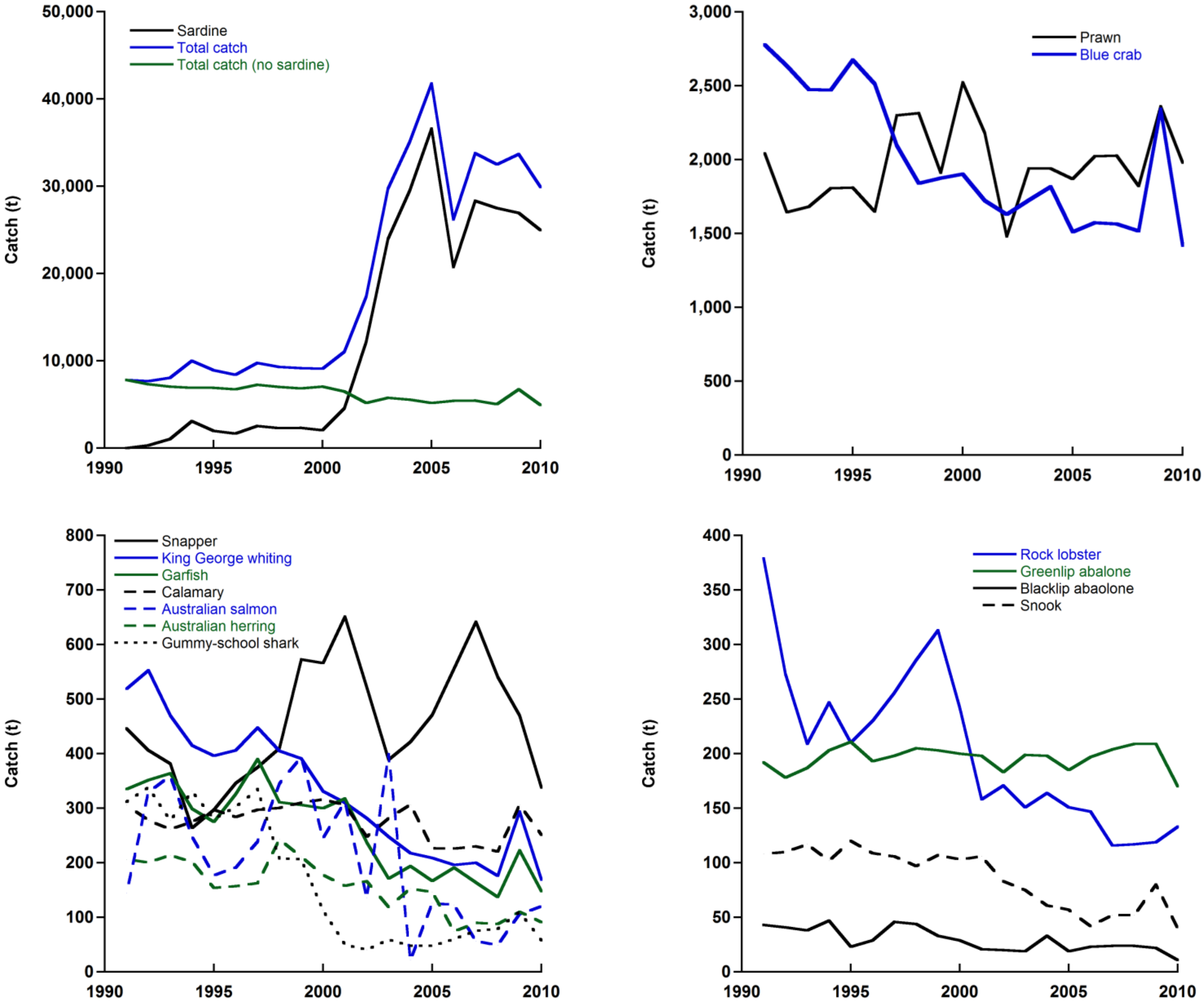


Figure 4.4 Trends in the total catch (t y^{-1}) from 14 main targets species in fisheries in the Spencer Gulf ecosystem between 1991 and 2010.

Based on outputs of the SGE EwE model, there have been significant changes in the biomass of some taxa groups between 1991 and 2010 (Fig. 4.5). For marine mammals, New Zealand fur seals have shown strong recovery since the early 1980s, with populations having more than trebled in South Australia over the last three decades (Shaughnessy et al. 2014). For Australian sea lions, biomass in southern Spencer Gulf increased in the early and mid-2000s, and then declined (Fig. 4.5). In the SGE EwE model, changes in seal biomass were fixed to times series data available for both species (see Appendix B). Common dolphin biomass was estimated to have declined since 1991, and model outputs suggest this occurred in response to increasing mortality rates from bycatch in the sardine fishery. Although mitigation strategies have been introduced since 2004 (Hamer et al. 2008), the model assumes a closed population of common dolphins, and hence the magnitude of the impact is likely to be positively biased in the model, as recent population genetics data indicate some level of movement between common dolphin populations (Bilgmann et al. 2014). Bottlenose dolphin biomass was estimated to have remained relatively stable between 1991 and 2010 (Fig. 4.5).

Notable projected changes in seabird biomass include declines in petrels and little penguins, and increases in gulls (Fig. 4.5). Initial declines in petrels and little penguins in the early to mid-1990s appear to be driven by higher rates of predation mortality by New Zealand fur seals, however, the model predicts these had declined markedly and stabilised by the late 1990. The biomass of shags, cormorants and terns were projected to have changed little over the model period (Fig. 4.5). For elasmobranchs, combined groups of pelagic sharks (white, whaler, smooth hammerhead and common thresher sharks), demersal sharks (gummy, school, Port Jackson and other demersal sharks) and rays and skates all were projected to have increased in biomass over the modelled period, with the greatest increases in rays and skates and demersal sharks. Most increases in the biomass of these groups are attributed to marked reductions in fishing mortality that have occurred principally through declines in fishing effort in the marine scalefish net and the demersal shark fisheries in the SGE (see above and Goldsworthy et al. 2011, Goldsworthy et al. 2013).

Projected changes in the biomass of seven key commercially fished species between 1991 and 2010 in the SGE are also presented in Figure 4.5. These mirror individual plots presented in Figure 4.3. The projected decline in the biomass of sardines within the SGE essentially reflect the marked increase in fishing mortality as a consequence of the rapid expansion in the fishery throughout the early to mid-2000s. However, it is also an artefact of the ‘other production’ (net immigration) that was imposed to reflect a net movement of sardine biomass into the SGE from outside the model domain area. Without this net movement of sardine biomass into the SGE, the EwE model could not reconcile historic sardine catches with the available biomass of sardines within the SGE model domain (see section 4.3.1). Increases in prawn and snapper biomass appear largely to reflect declining fishing mortality on these species over the model period (Fig. 4.5). Fishing mortality for blue swimmer crabs has also declined over the model period, but the projected biomass for this species has declined (Fig. 4.5). Model results suggest this biomass decline is a consequence of increased predation mortality from snapper as a consequence of that species projected recovery (Fig. 4.5). The model output projects little change in calamari, King George whiting or garfish biomass over the model period, despite reductions in fishing mortality across these species (Fig. 4.5).

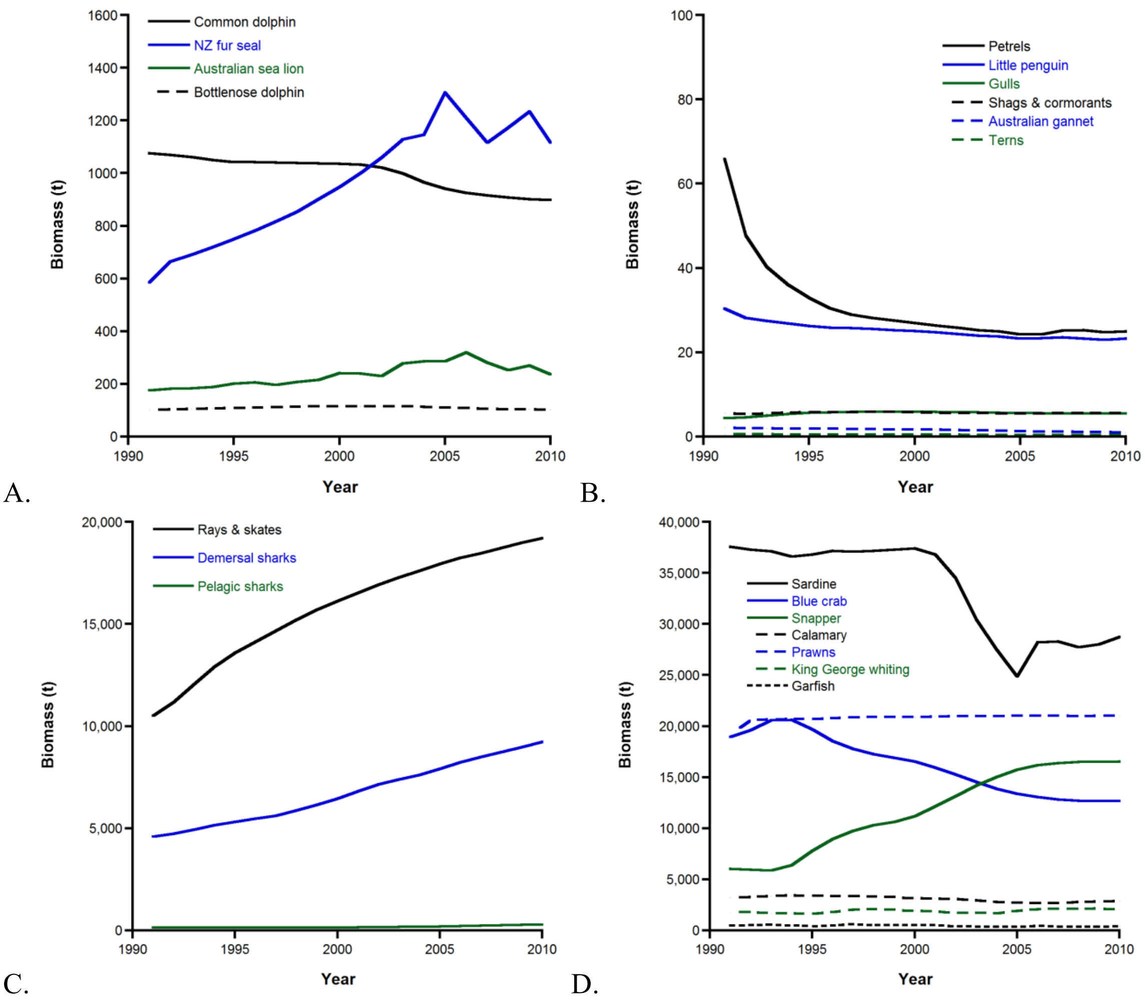


Figure 4.5 Estimated changes in the biomass of *Ecosim* SGE model groups between 1991 and 2010: (A) marine mammals (biomass change in New Zealand fur seals and Australian sea lions is forced in the model as these estimates are based on empirical data for these species in the SGE); (B) seabirds (C) chondricthyians; and (D) major commercially fished species.

4.3.4 Ecosystem indicators

The ecosystem indicators identified significant changes in the Spencer Gulf ecosystem between 1991 and 2010 (Fig. 4.6 a-d). As indicated above, total catch of the combined fisheries showed a 3.8 fold increase over the 20 year period, attributed to growth in the sardine fishery (Fig. 4.6a). With the marked increase in the growth of the sardine fishery (TL=4.2) and the concomitant decrease in the relative contribution of western king prawns (TL=2.4) there has been a consistent increase in the mean trophic level of the catch between 1991 and 2010 from around 3.2 to 3.9 (Fig. 4.6b). There was a general increasing trend in the fishing in balance (FIB) index between 1991 and 2010, however for most of the 1990s it was very low, and close to zero up until 2002 (Fig. 4.6c). An FIB index of 0 indicates high production at lower trophic levels with fishing in balance; and an FIB index greater than 0 indicates an expansion of fishing and/or where bottom-up effects are occurring, resulting in more catch than expected (Coll et al. 2009). Both of these responses reflect the shift from prawns being the dominant fishery in Spencer Gulf in the early 1990s to the sardine fishery being dominant by the mid-2000s. Kempton's Q biodiversity index has generally been stable throughout the modelled period. The Kempton's Q index usually decreases with increased fishing impacts (and loss of species with a trophic level >3) and increases with growing biomass of high trophic level species. Results suggest that overall there has not been a marked change in biomass of species with a trophic level >3, nor has there been an increase in fishing impacts over the model period. There is some evidence from the ecosystem indicator that the ecosystem was under stress during the 1990s, but had recovered significantly during the 2000s. The reduction in fishing effort for many of the Marine Scalefish Fishery species may have been responsible for this recovery.

4.3.5 Ecospace

A preliminary *Ecospace* model was built for the SGE (Fig. 4.7). Maps representing bathymetry and habitat were developed, the habitat model included seagrass, microalgae and an 'other' category. There is large uncertainty in the habitat types and their distribution in the deeper (central) parts of Spencer Gulf, and more detailed habitat maps are needed for spatial ecosystem modelling. At present, no habitat maps have been developed/estimated for the functional groups, nor have maps of the spatial distribution of fishing/aquaculture effort and aquaculture and the affinity of predators to various habitats. As a consequence all scenarios were modelled in *Ecosim* because only a preliminary version of the spatial model was developed.

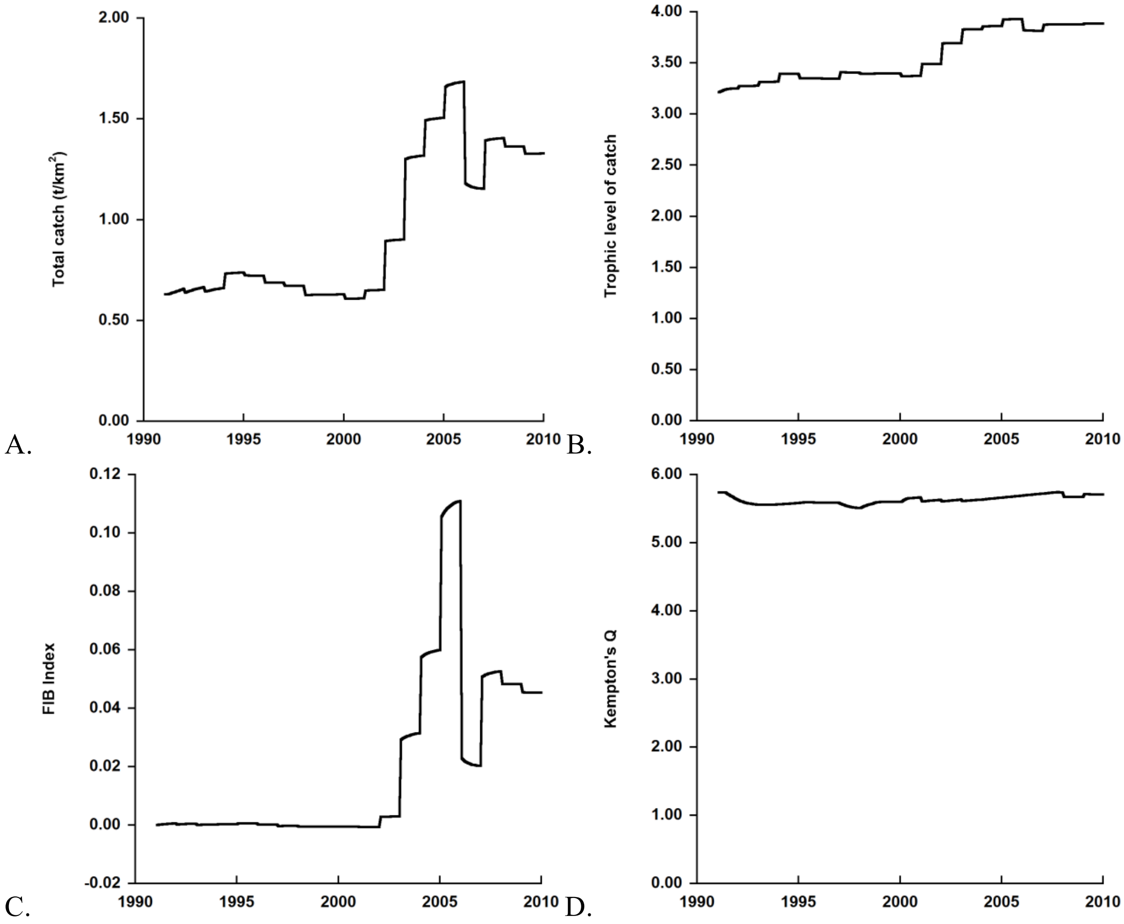


Figure 4.6 Ecosystem indicators calculated from the Spencer Gulf Ecosystem (*Ecopath with Ecosim*) model for the period 1991 to 2010. A. Changes in the landings of all fleets (total catch), B. Mean trophic level of the catch, C. Fishing In Balance (FIB) index, and D. Kempton's Q biomass diversity index.

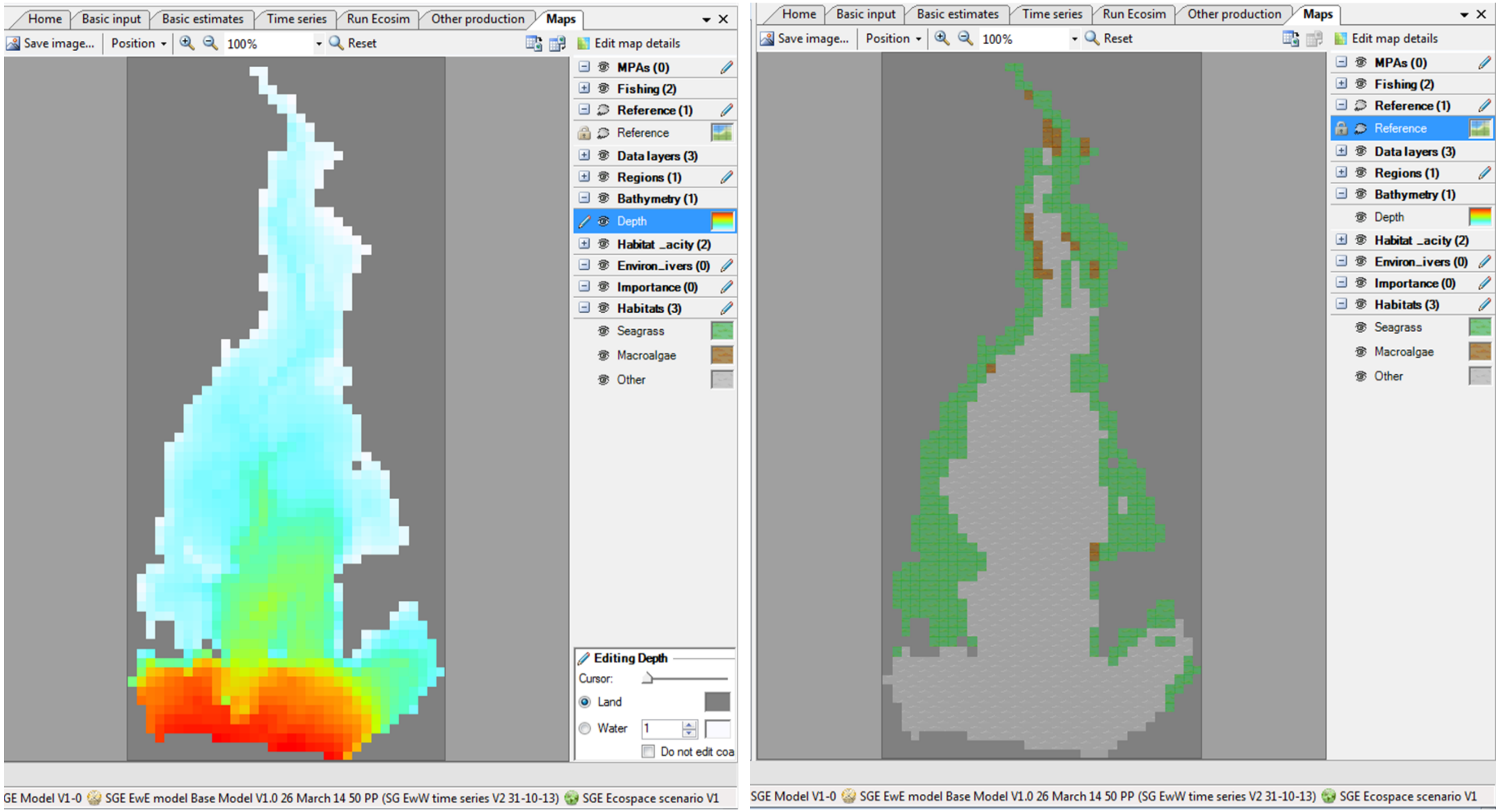


Figure 4.7 Habitat base map defined for the *Ecospace* model for the Spencer Gulf Ecosystem for the period 1991 to 2010. The bathymetry layer is presented on the left (heat map of depth), and the habitat layer on the right (green = seagrass, brown = macroalgae, grey = other).

4.3.6 Model scenario results

Below the various scenarios including increased aquaculture, increased sardine catch and changes in catch and fishing effort of western king prawns are examined.

4.3.6.1 Increased aquaculture scenarios

The effects of changing nutrient loading as a consequence of different finfish aquaculture scenarios on the SGE model are presented in Figure 4.8. Results show the relative change in biomass of trophic groups from the reduced nutrient loading from aquaculture (S1), compared to current (2010/11, ~1.5 kT/Year of N; CS) and maximum estimated aquaculture nutrient loading (~6.1 kT/Year of N, S2) (see section 2) (Middleton et al. 2013). Results suggest that increased nutrient loading from aquaculture leads to increases in phytoplankton production, and increases in the biomass of detritus groups, however, the resulting increases in zooplankton production appear to increase the biomass of gelatinous zooplankton to the extent that they exert greater predation pressure on other zooplankton groups resulting in a net reduction in their biomass (Fig. 4.8). Under the maximum aquaculture scenario (S2), the reduced biomass of large and small zooplankton, benthic and planktonic microflora and meiofauna leads to trophic cascades through the benthic ecosystem. In the benthic system, a reduction in production results in general declines in the biomass of benthic crustaceans (except large crabs and bugs), molluscs, and fish, ultimately causing declines in the biomass of benthic foraging apex predators including benthic sharks and rays, birds (shags and cormorants) and marine mammals (Australian sea lions and bottlenose dolphin) (Fig. 4.8). In contrast, under the maximum aquaculture scenario, the pelagic system shows increases in phytoplankton production have cascading effects throughout the pelagic component of the food web. These result in increases in the biomass of most small pelagic fish and their predators, notably large piscivorous fish (barracouta, yellowtail kingfish), pelagic sharks (white, whaler, thresher and smooth hammerhead) and marine mammals (New Zealand fur seal, common dolphin) (Fig. 4.8).

4.3.6.2 Increased sardine catch scenarios

Scenarios (SCS1, SCS2 and SCS3) examining the impact of increasing catches of sardine on the SGE model are presented in Figure 4.9. Predictably, increased exploitation rates impact most on the target species (sardine), and their key predators (Fig. 4.9). Note, however, that the relative magnitude reduction in sardine biomass under successive scenarios is less than expected in a closed system, because of the level of immigration of sardine biomass imposed on the model to ensure that observed catches would not exceed the available biomass within the SGE model domain (see section 4.3.1). Some of the more sensitive predator groups include barracouta, SBT, Australasian gannet, terns and common dolphins, but also snook, snapper and many of the pelagic sharks (smooth hammerhead, whaler, and white) (Fig. 4.9). The main trophic groups that appear to benefit from higher exploitation rates of sardine include other squid, anchovy, and blue mackerel; increases in these groups appear to benefit yellowtail kingfish, little penguins, bottlenose dolphins and New Zealand fur seals (Fig. 4.9).

4.3.6.3 Increased prawn catch and effort scenarios

Scenarios (PCS1, PCS2, PCS3) examining the impact of increasing catches of western king prawn in the Spencer Gulf prawn fishery on the SGE model are presented in Figure 4.10. Increased exploitation rates impacted most on the target species (western king prawns), and their key predators (Fig. 4.10). Declines in biomass were most directed towards medium-sized crustacean specialist fish, and medium-sized piscivorous fish, and declines in these groups appear to drive most declines in biomass observed in apex predators, including demersal sharks (Port Jackson, gummy and school), some pelagic sharks (smooth hammerhead, whaler and white), shags and cormorants and marine mammals (dolphins and seals) (Fig. 4.10). Small increases in biomass occurred in southern calamari, anchovy, medium molluscan fishes and small polychaete specialist fishes, silverbelly, barracouta, SBT, thresher shark, little penguin and Australasian gannet (Fig. 4.10).

The previous scenarios assume only the target species is caught, however, the bycatch of non-targets species is significant in the Spencer Gulf prawn fishery, so the implications of increased catches of western king prawns are better explored through scenarios of increased fishing effort (PES1, PES2, PES3). These scenarios where fishing effort was increased by 10%, 20% and 30% of 2010 levels, are presented in Figure 4.11. Increases in fishing effort resulted in decreases in the biomass of western king prawns, rock lobster, cephalopods (other squids, giant cuttlefish), and a range of small and medium fish groups including sprats, jack mackerel, Degens/rough leatherjacket, red mullet, Australian salmon, medium echinoderm specialist fish, medium piscivorous fish, silver trevally, and a range of demersal sharks (rays and skates, Port Jackson shark, gummy shark). Declines were also observed in whaler sharks, gulls, shags and cormorants and New Zealand fur seals (Fig. 4.11). Increases in biomass were observed in blacklip abalone, southern calamari, anchovy, blue mackerel and syngnathids, with minor increases observed in medium molluscan fish, silverbelly, Australian herring, barracouta, SBT, thresher shark and smooth hammerhead, little penguins and Australasian gannets (Fig. 4.11). Importantly, the magnitude of biomass declines among trophic groups under increased fishery effort scenarios were an order of magnitude greater that those modelled under scenarios of increased catch (Fig. 4.11).

These three model scenarios provide a good spectrum of the potential impacts and stresses on the SGE. Finfish aquaculture provides an example of how bottom-up changes through additional nutrient loading can affect both benthic and pelagic systems; changes in sardine catch in the South Australian sardine fishery provide an opportunity to examine the potential impacts of removing biomass of a key small pelagic fish species; whereas changes in catch and fishing effort of western king prawns in the Spencer Gulf prawn fishery provide an opportunity to examine the impacts of fishing on benthic communities. These examples provide an indication of the types of impacts that can be modelled, but there are many more possible scenarios from other fisheries that could be examined. The examples are by no means definitive, outputs should be viewed as indicative, and they should be not seen as quantitative. Development of the ecosystem model is just the first step in a longer process of model improvement. More work is required to ensure that the information underpinning the trophic network, biomass and consumptions rates are robust, and that data gaps are addressed. Improving the environmental time series data sets and identifying those with biological relevance that can be used to predict ecological response in the model, will be essential in developing capacity to assess the potential impacts of environmental variability and climate change on fisheries and aquaculture production, and the Spencer Gulf ecosystem. Additional work is needed to fully develop the *Ecospace* model, particularly to improve habitat layers, ascribe key habitats to trophic groups, and to spatially allocate fishing effort and aquaculture. Incorporating other key values such as ports and marine protected areas will also improve our ability to assess the ecological consequence in a more spatially refined way. All these model improvements will ultimately increase confidence and value in the model outputs and its utility as a decision support tool for complex natural resource management issues.

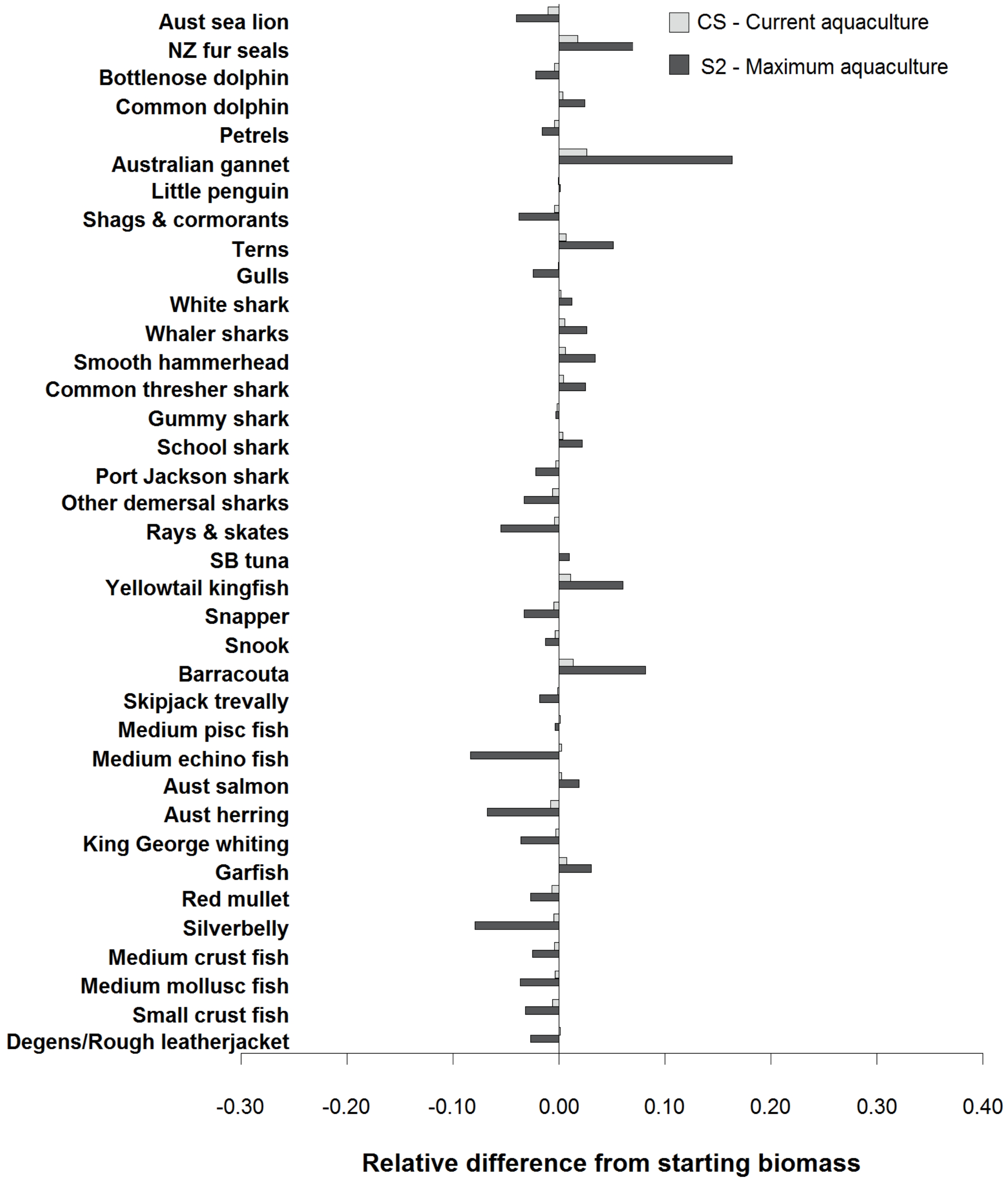


Figure 4.8 Predicted change in the biomass of functional groups in the Spencer Gulf Ecosystem model from 1991 to 2050 for scenarios where finfish aquaculture (SBT and YTK) production and nutrient loading is maintained at current (2010/11) levels (CS, ~1.5 kT/Year) and where it is increased to a maximum (S2, ~6.1 kT/year). Biomass change at 2050 is plotted relative to the starting biomass in 2010 under the base scenario of no aquaculture nutrient loading (S1).

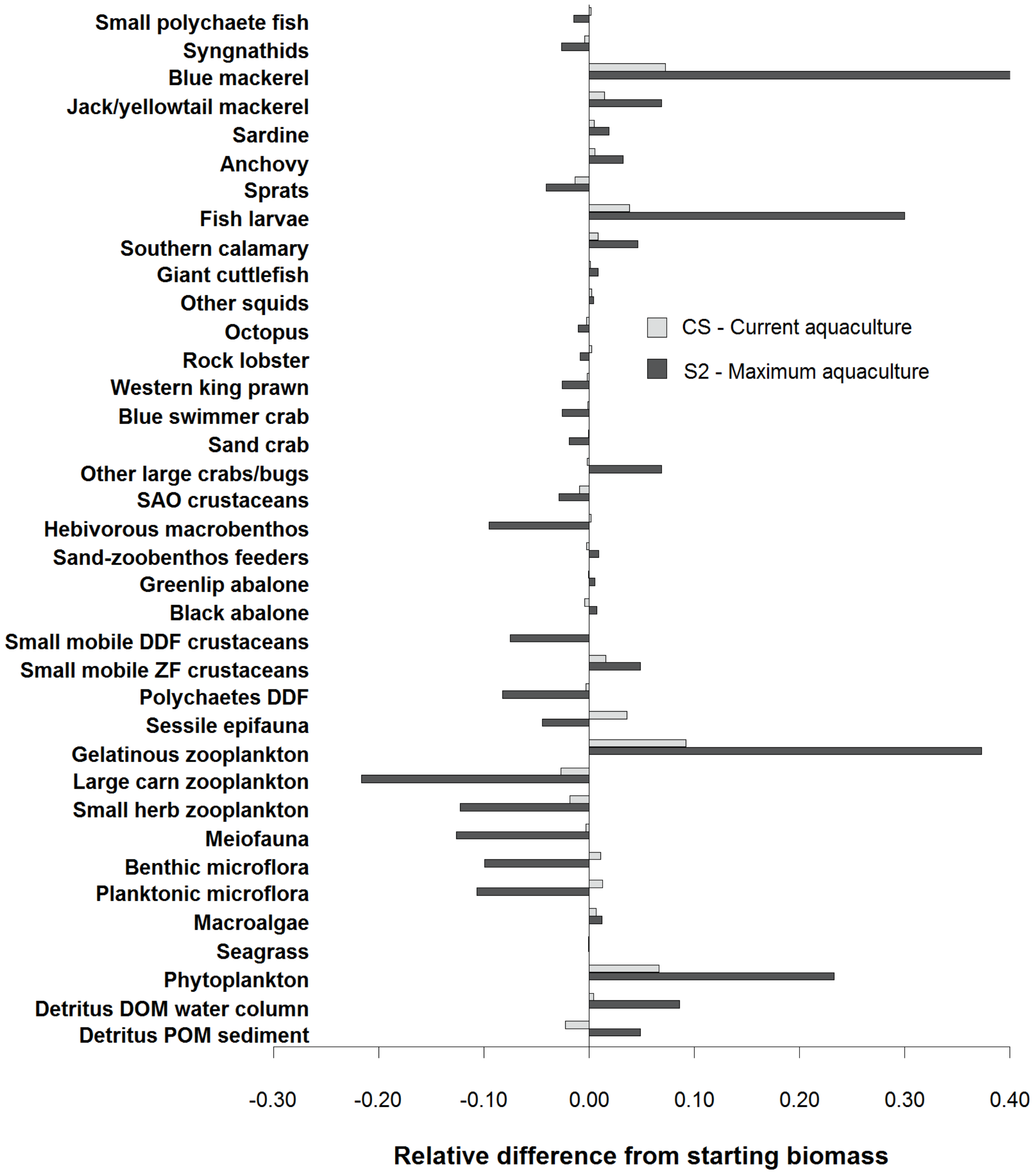


Figure 4.8 continued. Predicted change in the biomass of functional groups in the Spencer Gulf Ecosystem model from 1991 to 2050 for scenarios where finfish aquaculture (SBT and yellowtail kingfish) production and nutrient loading is maintained at current (2010/11) levels (CS, ~1.5 kT/Year) and where it is increased to a maximum (S2, ~6.1 kT/year). Biomass change at 2050 is plotted relative to the starting biomass in 2010 under the base scenario of no aquaculture nutrient loading (S1).

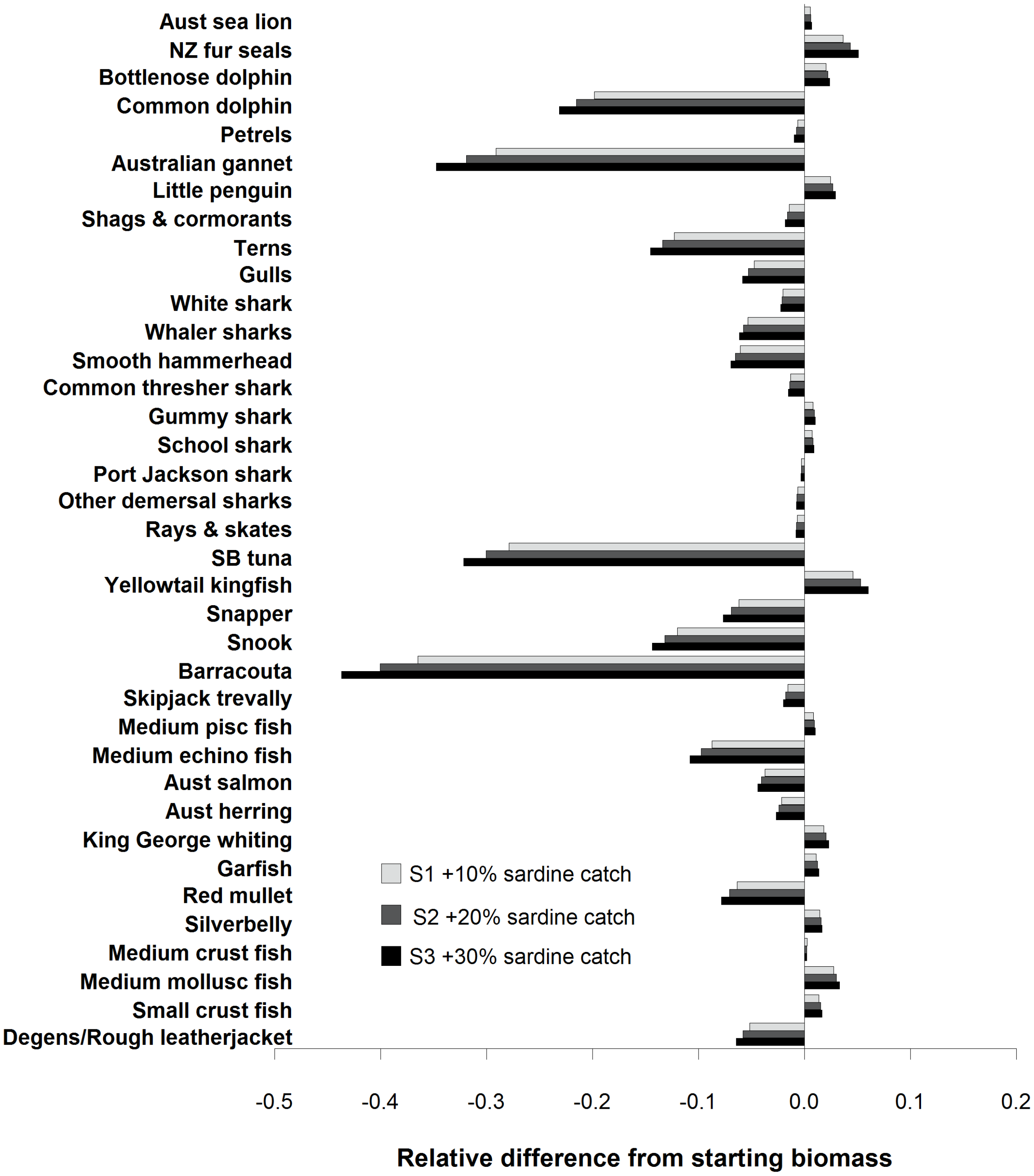


Figure 4.9 Predicted change in the biomass of functional groups in the Spencer Gulf Ecosystem model from 1991 to 2050 for scenarios where sardine catch is increased by 10% (SCS1 labelled S1 on figure), 20% (SCS2, labelled S2) and 30% (SCS3, labelled S3). Biomass change at 2050 is plotted relative to the starting biomass in 2010 under the base scenario of 2010 sardine catch levels.

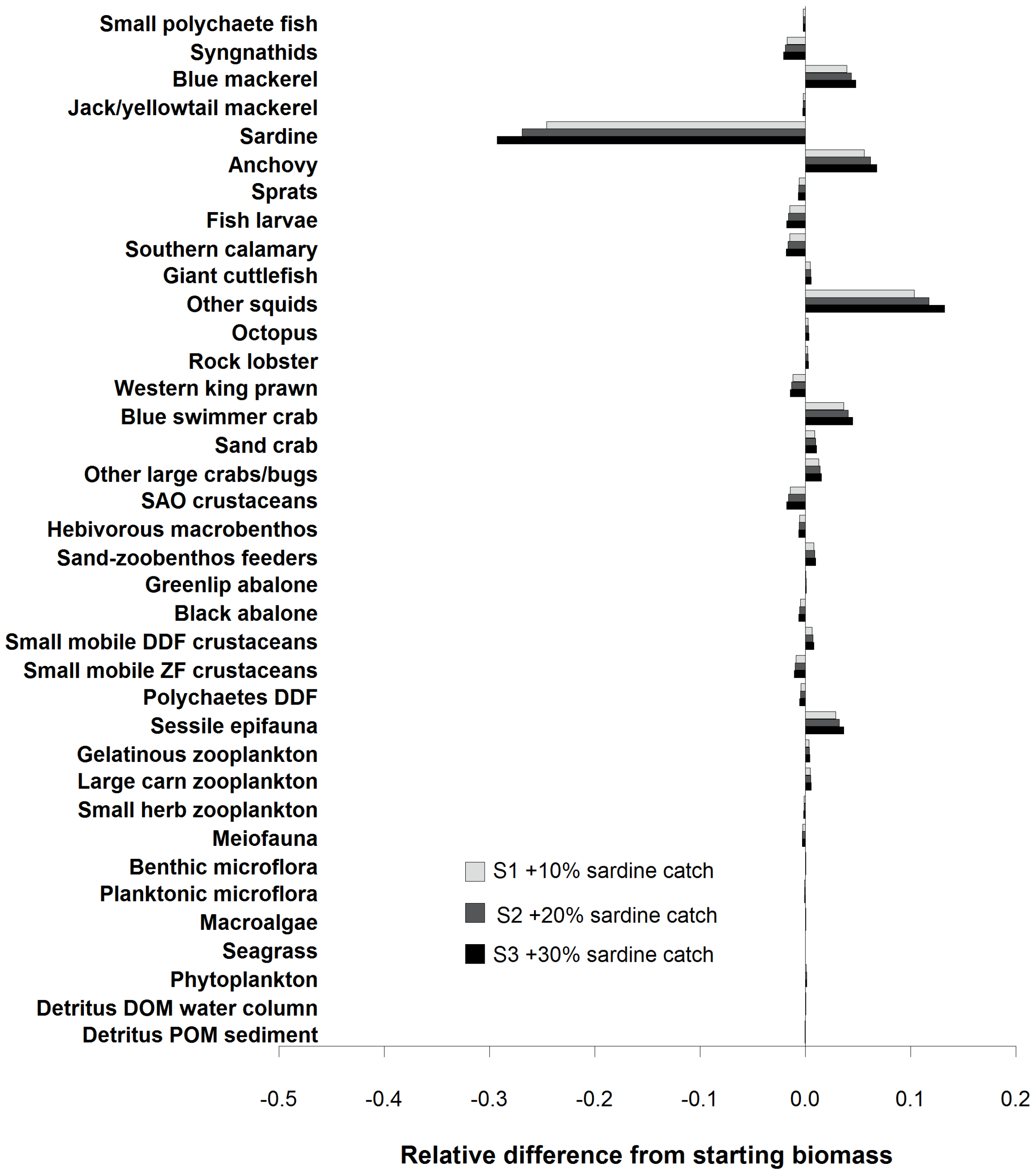


Figure 4.9 continued. Predicted change in the biomass of functional groups in the Spencer Gulf Ecosystem model from 1991 to 2050 for scenarios where sardine catch (SCS) is increased by 10% (S1), 20% (S2) and 30% (S3). Biomass change at 2050 is plotted relative to the starting biomass in 2010 under the base scenario of 2010 sardine catch levels.

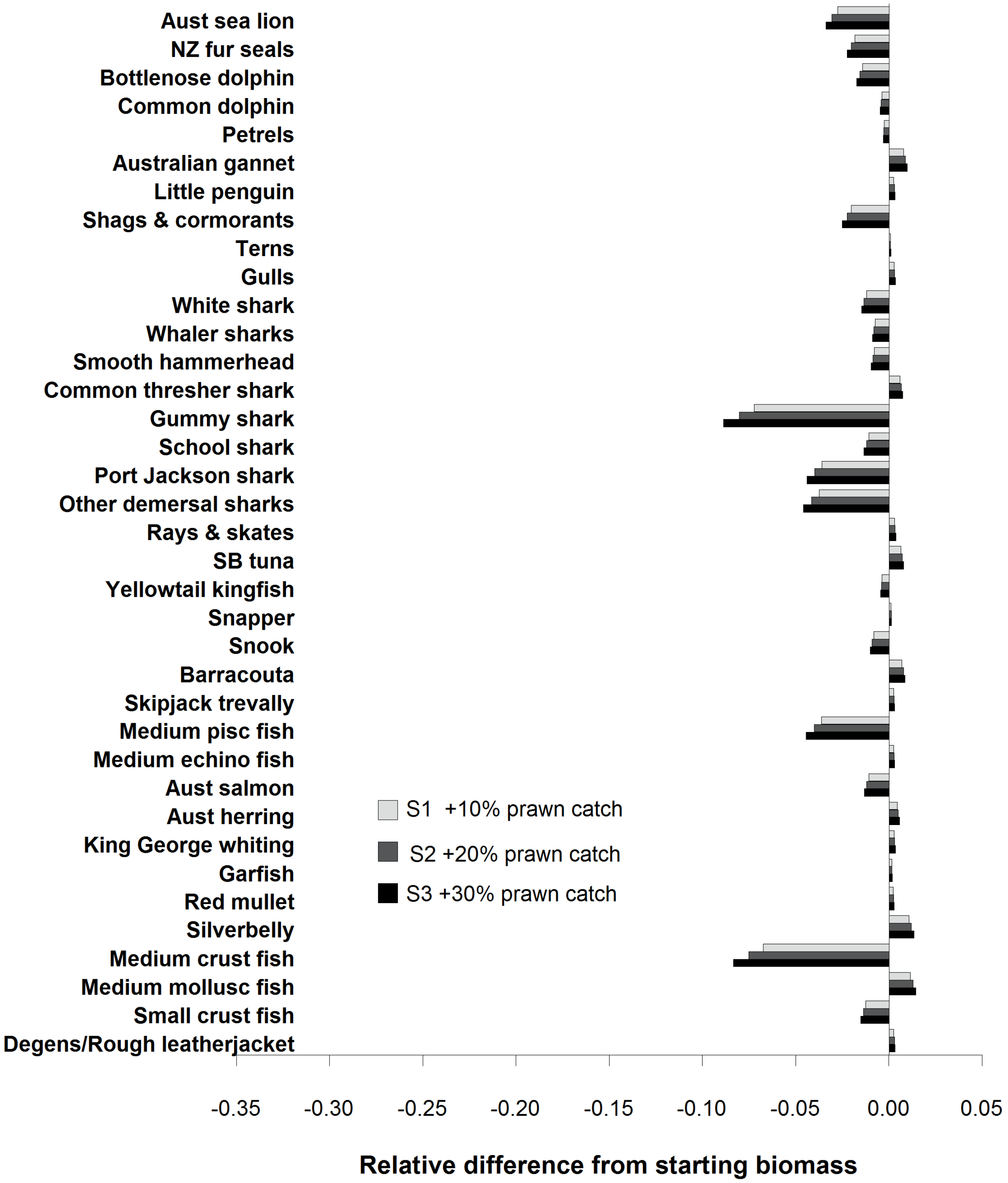


Figure 4.10 Predicted change in the biomass of functional groups in the Spencer Gulf Ecosystem model from 1991 to 2050 for scenarios where western king prawn catch is increased by 10% (PCS1 labelled S1 in figure), 20% (PCS2 labelled S2) and 30% (PCS3 labelled S3). Biomass change at 2050 is plotted relative to the starting biomass in 2010 under the base scenario of 2010 western king prawn catch levels.

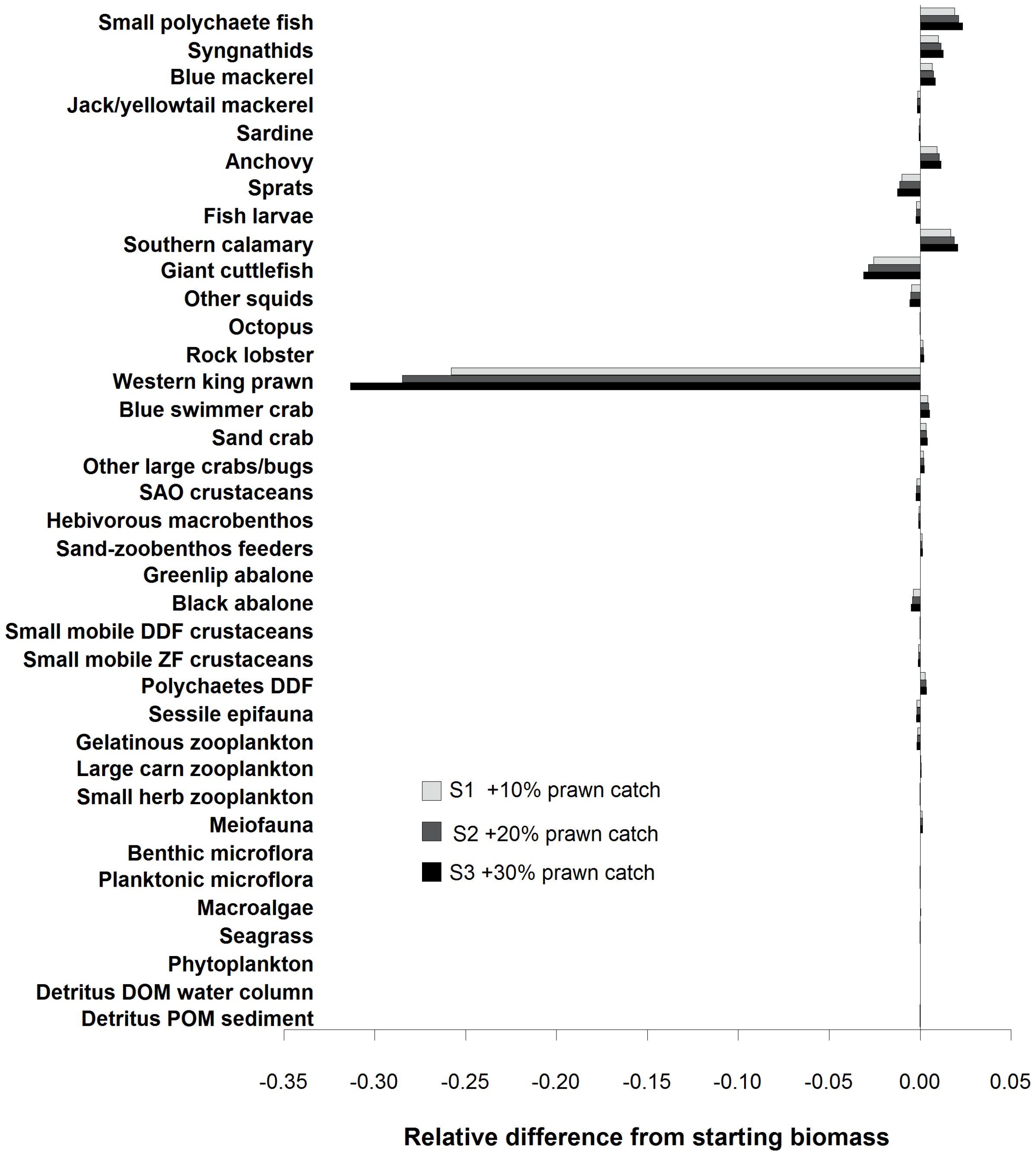


Figure 4.10 continued. Predicted change in the biomass of functional groups in the Spencer Gulf Ecosystem model from 1991 to 2050 for scenarios where western king prawn catch (PCS) is increased by 10% (S1), 20% (S2) and 30% (S3). Biomass change at 2050 is plotted relative to the starting biomass in 2010 under the base scenario of 2010 western king prawn catch levels.

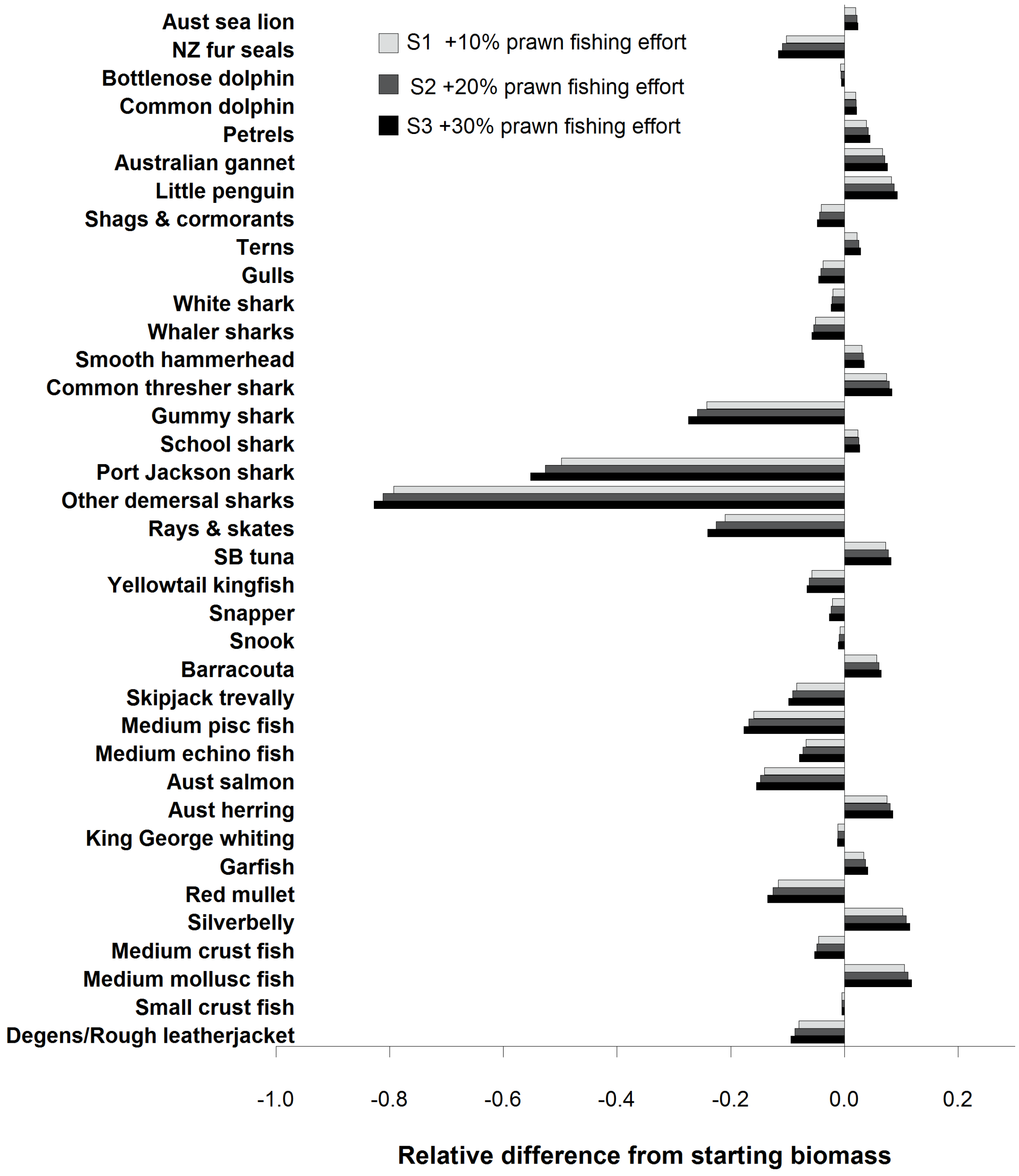


Figure 4.11 Predicted change in the biomass of functional groups in the Spencer Gulf Ecosystem model from 1991 to 2050 for scenarios where western king prawn fishing effort is increased by 10% (PES1 labelled S1 in figure), 20% (PES2 labelled S2) and 30% (PES3 labelled S3). Biomass change at 2050 is plotted relative to the starting biomass in 2010 under the base scenario of 2010 western king prawn catch levels.

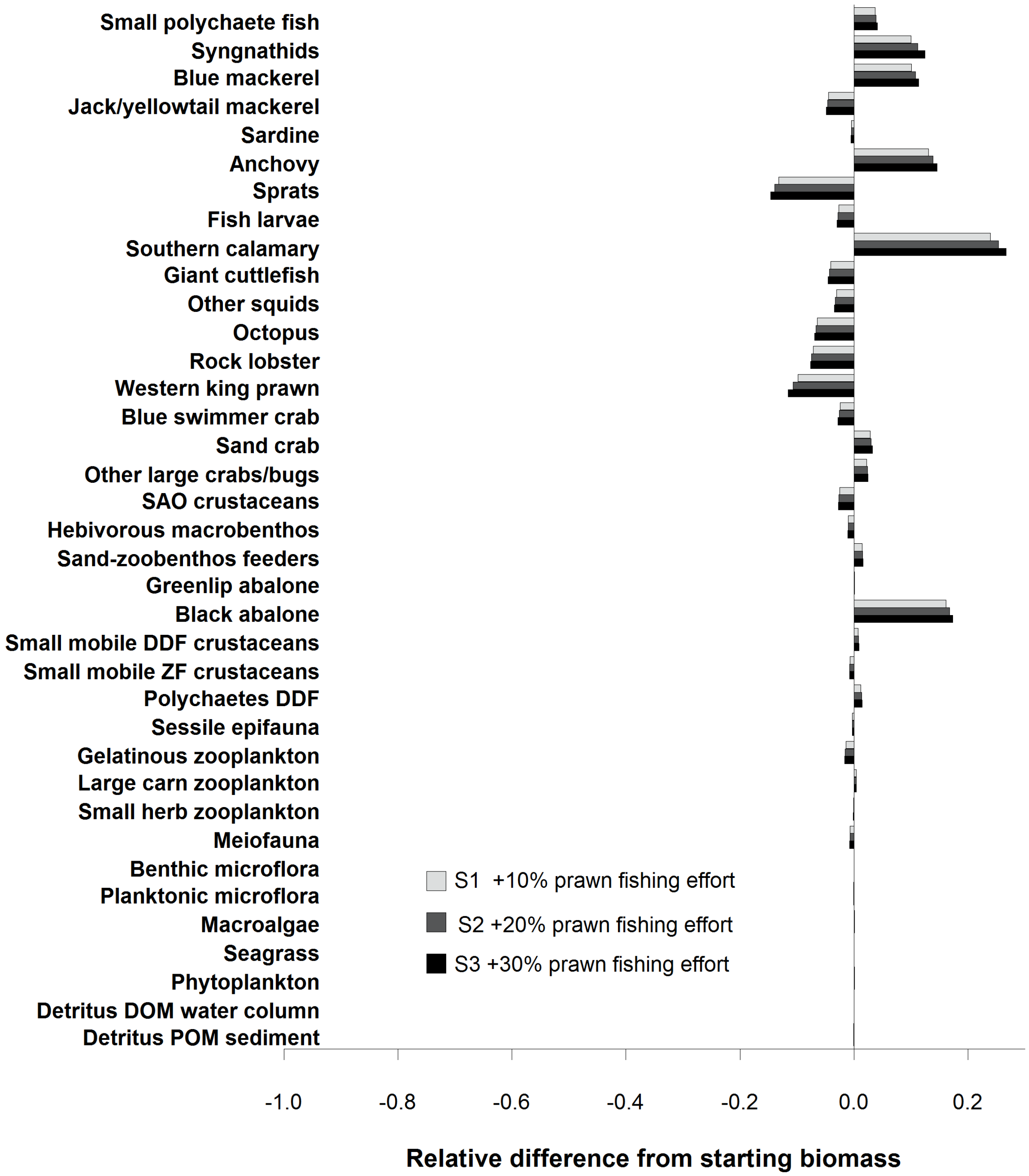


Figure 4.11 continued. Predicted change in the biomass of functional groups in the Spencer Gulf Ecosystem model from 1991 to 2050 for scenarios where western king prawn fishing effort (PES) is increased by 10% (S1), 20% (S2) and 30% (S3). Biomass change at 2050 is plotted relative to the starting biomass in 2010 under the base scenario of 2010 western king prawn catch levels.

5 Conclusion

Spencer Gulf is an important area of the State for both wildcatch fisheries and aquaculture, but there are also a range of other activities that effect Gulf waters. In addition, the region is important for the expanding mining industry and there is likely to be an increase in associated infrastructure (e.g. port development). This project provided a whole of ecosystem model that is capable of addressing ‘what if’ scenarios for the fisheries and aquaculture sectors, but the model has not been used for other industries.

The validated, three-dimensional coupled hydrodynamic-biogeochemical model for Spencer Gulf allowed an understanding of how natural ecosystem processes and human activities can interact. Anthropogenic nutrient loads from finfish aquaculture significantly influenced the lower trophic ecosystem and associated water quality parameters in Spencer Gulf. Thus, this model enhanced our understanding of interactions between natural processes and human activities and their effects on water quality.

A habitat model was also developed to predict the occurrence and cover of seagrass in Spencer Gulf. Information from both these models was used in the Spencer Gulf ecosystem model, a trophic mass-balance model, that allowed dynamic simulations thereby enabling forecasting of ecosystem response to environmental perturbations. The Spencer Gulf ecosystem model allowed performance indictors for ecosystem health (objective 1) and the ecological performance of the seafood industry (objective 2) to be assessed. Ongoing discussion with stakeholders is also important.

The trophic mass balance model (or Spencer Gulf ecosystem model) clearly demonstrated the importance of primary producers, especially seagrass and macroalgae, in the Spencer Gulf ecosystem (see Fig. 4.1). Significant changes in ecosystem indicators for Spencer Gulf were identified largely related to marked increases in catch in the sardine fishery and decreases in marine scalefish catch and effort through time. Thus, several measures of the marine ecosystem increased slightly over time (mean trophic level of catch, fishing in balance index), but in general biodiversity measures were slightly lower in the 1990s, and improved in 2000s. These ecosystem indicators provide information on which to assess the health of the Spencer Gulf ecosystem into the future.

Model scenarios provided an indication of potential impacts on the Spencer Gulf ecosystem. Finfish aquaculture demonstrated how bottom-up changes through additional nutrient loading may affect lower trophic groups, benthic and pelagic ecosystems. Thus, increased nutrient loading from aquaculture resulted in increases in phytoplankton and detrital groups, however there was also potential for an increase in gelatinous zooplankton. Changes in catch in the sardine fishery demonstrated potential impacts from removal of a key small pelagic fish species and changes in catch and fishing effort of western king prawns allowed impacts of fishing on benthic communities to be investigated. Increased exploitation rates of sardines and western king prawns impact most on the target species and their prey, but in the case of the prawn fishery do not include bycatch of non-target species. Scenarios associated with increasing fishing effort may more accurately reflect implications of increased catches of western king prawns. Biomass declines for a number of trophic groups were observed and these were an order of magnitude greater than those modelled under scenarios of increased catch. The model allows additional scenarios primarily in relation to nutrients, aquaculture and fisheries to be investigated in future.

6 Implications

The Spencer Gulf ecosystem model provides a preliminary tool to assess the impacts and interactions of fisheries and aquaculture. Several illustrative examples of the sorts of scenarios that can be investigated were provided. Ecosystem indicators were generally lower in the 1990s, and tended to increase in the 2000s indicating that the ecosystem was potentially under stress during the 1990s, but had undergone some recovery during the 2000s. The drivers of these changes warrant further investigation. The model provides a capacity to understand all fisheries and aquaculture that operate within Spencer Gulf.

The model strongly suggests that catches of sardines from Spencer Gulf are sustained by relatively high levels of migration into the region. The SGE model provides PIRSA fisheries managers and the South Australian Sardine Industry Association with an important tool to address potential scenarios associated with changing sardine catch and interactions with other fisheries in Spencer Gulf. The SGE model could also be used to examine the levels of migration required to sustain sardine catches in Spencer Gulf, and how different scenarios of sardine immigration may impact the catch availability, fishery sustainability and ecosystem impacts in Spencer Gulf. The previous ecosystem model, the eastern Great Australian Bight (eGAB) model, for sardines did not focus on regional impacts at the scale of the fishery, and included all continental shelf waters to 200 m depth between 132°E and 139.7°E, the Investigator Strait and the mid-lower portions of Gulf St Vincent and Spencer Gulf where the sardine fishery is centred (Goldsworthy et al. 2013). The SGE model built onto the earlier sardine ecosystem model and is a better tool for examining fishery sustainability and ecosystem impacts at the regional scale (Goldsworthy et al. 2011, Goldsworthy et al. 2013). However, lack of information on the rate of immigration of sardine into Spencer Gulf directly affects our estimates of the sensitivity of predator groups to increased sardine fishing.

Scenario testing suggested that the ecological impacts of increased bycatch were more significant than targeted catch in the western king prawn fishery and therefore efforts to reduce bycatch should be continued to reduce the ecological impacts of this fishery on the SGE (e.g. FRDC TRF 2013/052 and proposed follow-up research). The SGE model provides a good tool to explore the ecological benefits of a range of bycatch reduction devices and strategies in the prawn fishery. The Spencer Gulf prawn fishery is certified by the Marine Stewardship Council and as part of the ecosystem based management of the fishery is required to investigate potential impacts on other species. The SGE model therefore provides PIRSA fisheries managers and the Spencer Gulf and West Coast Prawn Fishermen's Association with the ability to examine scenarios around catch, effort and bycatch reduction in the prawn fishery and investigate how these may affect other important species including giant Australian cuttlefish.

If the Ecospace component of the SGE model is further developed then the potential impacts of spatial closures on fisheries catch in Spencer Gulf could be investigated. This would be of use not only to the range of fisheries operating in the region but also to PIRSA fisheries managers and to the Department of Environment, Water and Natural Resources who manage marine parks. A comparison of the advantages and disadvantages of different spatial ecosystem models should be undertaken prior to further development of the Ecospace model to determine which model is most suitable for the range of sectors in Spencer Gulf. Spatial models will also help understand spatial drivers and aspects of system dynamics in Spencer Gulf.

This study is a first step towards developing tools to assist with ecosystem based management and/or integrated ocean management. Expansion of the model to consider other sectors would be beneficial for all of government and the broader community. The Ecopath with Ecosim model may necessarily be the best model to use for other sectors and a review of available models would be beneficial. A multiple-model approach using different models (e.g. conceptual, toy, focused, system) is likely to be beneficial as each model has its own strengths and weaknesses. In addition, congruence between outputs of different modelling approaches will increase confidence in their utility and ability to assist with decision support.

7 Recommendations

This project has provided an ecosystem model that uses information from multiple model types to develop an understanding of how different uses of an ecosystem can interact and influence each other and the ecosystem itself. Further development and refinement of all models (hydrodynamic-biogeochemical, habitat distribution, trophic mass-balance) and data (habitat mapping, dietary) incorporated into the Spencer Gulf ecosystem model is now required (see Further development below) to provide tools that can be used by managers from all fisheries and aquaculture sectors operating in Spencer Gulf to fully consider the ecosystem impacts of their activities. Importantly, the suite of models also provides tools to enable interactions between different fisheries and aquaculture activities to be examined, and the resultant cumulative impacts on the environment to be assessed. As development pressure increases, it is these cumulative impacts that have the most potential to provide unwelcome surprises, as the current single-issue management approach traditionally employed pays little or no attention to them.

Being a relatively discrete ecosystem with traditional industries that are well understood (fishing and aquaculture), and a range of emerging industries, Spencer Gulf provides a good opportunity to develop a world best practise approach to ecosystem management across the full range of industries (and other uses) present. This has the potential to significantly reduce the impact of adverse environmental impacts, while at the same time allowing for better planning of resource allocation across different sectors, and reducing the overall costs of environmental planning and management. Currently, resource allocation tends to occur on either a first-come first-served basis, or is based on political influence, and there is no framework available to consider trade-offs between different sectors, and how to maximise returns, whether they be economic, environmental and/or social. In addition, as each new proposed development effectively starts from scratch when it comes to environmental assessment, much work is effectively duplicated, or is put into examining options that are simply not the best available. By having an integrated series of models such as this with which to undertake preliminary environmental assessments, options that are likely to be favourable can be determined early on in the planning process, and any on-ground work can be effectively targeted to fill in the data gaps. The *Ecopath with Ecosim* model presented here is one type of model for ecosystem based assessment and has clear applications for the fisheries and aquaculture sectors. It is recommended that a review of the available models is undertaken to determine which suite of models may be most appropriate for informing multiple use management of Spencer Gulf and for an integrated ocean management approach (see also further development).

8 Further development

Further development and refinement of the three-dimensional hydrodynamic-biogeochemical model is required. For example, co-limiting nutrients and harmful algae could be incorporated into the model. At present the outputs of the alternative scenarios for this model are considered indicative only. If additional nutrient sources are added to Spencer Gulf (e.g. through new aquaculture leases or industrial and waste water sites) then further refinement and validation of the model will be required. The hydrodynamic-biogeochemical model is also based on data from the 2010/2011 period and additional data could be collected from other years to build further confidence in the model. Further data collection is required for refinement and validation of the hydrodynamic-biogeochemical model. In addition, there is a need for a Spencer Gulf integrated marine observing system to capture the physical, chemical and biological environment, as well as provide information at local scales for proposed developments. At least three observing sites in Spencer Gulf are required, but the locations need to consider the broader Spencer Gulf research program as well as the needs of the South Australian government agencies.

Although benthic primary production by seagrasses and macroalgae are important components for the Spencer Gulf ecosystem model, the majority of benthic habitats within the Gulf remain unsurveyed (see Fig. 3.1). There is consequently considerable uncertainty regarding benthic primary production, particularly throughout deeper regions for which aerial surveys are ineffective. Statistical models of seagrass occurrence and cover performed well when judged against training and hold-out cross-validation datasets; consequently, these models were extrapolated to unsurveyed regions to generate predictions for these areas. Ideally, however, additional benthic surveys should be used to validate and improve these model predictions. Additional surveys could also permit the development of species-specific models and predictions for seagrasses, as well as to investigate changes in seagrass cover and productivity over seasonal and longer time-scales. Further, the lack of reliable information on the presence of rocky reef habitat throughout much of Spencer Gulf currently hampers our ability to develop useful predictions of macroalgae occurrence and additional benthic surveys could address this knowledge gap. Given the suggested importance of macroalage in the ecosystem model (Fig. 4.1) there is a critical need to obtain information on rocky reef habitats in Spencer Gulf. Maps are also required of the spatial distribution of fishing and aquaculture effort – while these data are largely available, it was beyond the scope of the current project to collate it and incorporate it into the model.

The Spencer Gulf ecosystem model is a first step in providing a model to address ‘what if’ scenarios. Further work is required to ensure that the information underpinning the model (e.g. trophic network, biomass and consumption rates) is robust. Data from Spencer Gulf in relation to dietary, biomass and consumption rates are required as these are currently key data gaps for many species. For addressing potential impacts of environmental variability and climate change on the Spencer Gulf ecosystem, as well as fisheries and aquaculture production, it is important that improved environmental time series are incorporated. Sensitivity analyses should also be undertaken to better understand which parameters are “critical” to the model. Improved knowledge of rates of sardine immigration into Spencer Gulf is needed to assess impacts of increased sardine catches on populations of predatory species.

A preliminary spatially resolved model was obtained, and additional work is required to fully develop the *Ecospace* model. For example, improved data layers around key habitats (see above), ascribing key habitats to trophic groups, spatial allocation of fishing and aquaculture effort, and incorporating other spatial layers in relation to the full range of activities in Spencer Gulf is required. The spatial component is critical for use as a decision support tool that can evaluate alternative management scenarios and should be dynamic in nature. A graphical user interface for all models could be developed which would facilitate greater uptake by managers, industry and the broader community (e.g. CarCap 1.0, James et al. 2013). Spatial layers for the full range of activities in Spencer Gulf will also allow areas of individual and cumulative stressors to be identified such that potential interactions between the fisheries and aquaculture industries and other activities could be identified.

Unfortunately, the module of the *Ecopath with Ecosim* software that allowed economic performance of the seafood industry to be assessed under the various scenarios could not be utilised. Once this module is widely available it is recommended that this component be incorporated in the Spencer Gulf ecosystem model. This would allow the economic performance of the seafood industry to be investigated in relation to changing scenarios and may provide further guidance on potential tradeoffs.

Our focus in this project was on using the *Ecopath with Ecosim* model, however there are a range of models that could be used. The *Ecopath with Ecosim* model provides a suitable ecosystem model to consider fisheries and aquaculture effects on other species, but there are many other activities within Spencer Gulf. It is not clear how some of these activities could be incorporated into the existing ecosystem model. For example, it is unclear how the full effects of shipping could be incorporated especially in a spatial context. Whole-of-system models are increasingly being used to inform management of natural resources, but marine systems remain a challenge given the environmental and anthropogenic drivers, biophysical, economic and social components (Fulton 2010). These models all have their own focus, resolution, strengths and weaknesses (Fulton 2010). It is beyond the scope of this report to review such models but a key point consistently made is that multiple models should be considered as there is no one “right” model (Fulton 2010). The focus has traditionally been on the biophysical component with less explicit consideration of social and economic components. Whole-of-system models are also complex and challenging to use and are best utilised to consider ‘what-if’ management questions or scenarios. Both qualitative and quantitative ecosystem models can be used in decision-making. Generally, different modelling approaches can be used to complement each other to get a more robust understanding of the system. A review of available models and their suitability for multiple use management in Spencer Gulf should be considered.

An integrated approach to marine management is required that considers (1) multiple objectives, (2) conflicts and synergies of marine users, (3) the risk of cumulative impacts of various activities, (4) existing spatial zoning or management options and (5) scenario testing (Stelzenmueller et al. 2013). A range of models and tools are required to address questions regarding risk assessment, forecasting and modelling, as well as simulation models to address “what if” scenarios in relation to planning options. The spatial component suggests benefits from implementation of a Geographic Information System (GIS) framework. Value trees may help identify various objectives for which measurable attributes are then identified which allow decision alternatives to be evaluated (Stelzenmueller et al. 2013). A comprehensive approach to ranking human activities and assessing cumulative impacts is important. A recent review suggested that decision support tools for use in ecosystem-based marine assessment were currently being developed, although many of the models were technically complex and could only be used by scientists or programmers despite a need to engage stakeholders and decision makers (Stelzenmueller et al. 2013).

9 Extension and Adoption

This FRDC project forms one component of the first phase of the broader Spencer Gulf Ecosystem Development Initiative (SGEDI), a MISA initiative led by the Environment Institute, University of Adelaide, but also involving SARDI and Flinders University. SGEDI has held a series of stakeholder workshops based around specific sectors (e.g. fisheries and aquaculture; recreation and conservation; mining, manufacturing and other industries) and in regional centres (e.g. Port Lincoln, Whyalla, Port Augusta, Port Pirie, Wallaroo) where key concerns of stakeholders, what they saw were important information gaps, and aspirations for the region were discussed. These were held in late 2012. As part of these workshops the current project was discussed. Additional stakeholder workshops were held in early 2014 in regional centres and Adelaide, and included a whole of government workshop. Future stakeholder workshops are also planned to communicate results of the project – these workshops will extend and communicate the project to the full range of industries operating in Spencer Gulf as well as the broader community. The Board of the SGEDI has also been updated regularly on the project.

The project has been communicated to other researchers (including managers) through a poster presentation at the Australian Society for Fish Biology conference in Adelaide (2012), Marine Innovations Southern Australia (MISA) presentations in Adelaide and Port Lincoln (2012), the NRM Science Conference in Adelaide (presentations by Gillanders and Goldsworthy; 2014) and the National Estuaries Network meeting in Adelaide (2014). Presentations were also made to the Upper Spencer Gulf Common Purpose Group and the Port Pirie Regional Sustainability Forum.

We have discussed the project with PIRSA fisheries and aquaculture managers, and will present to the relevant fisheries and aquaculture management groups including to the Australian Southern Bluefin Tuna Industry Association, Marine Fishers Association, South Australian Sardine Industry Association, Spencer Gulf and West Coast Prawn Fishermen's Association and the South Australian Blue Crab Pot Fishers Association.

10 References

Ainsworth, C. H. and T. J. Pitcher. 2006. Modifying Kempton's species diversity index for use with ecosystem simulation models Ecological Indicators **6**:623-630.

Avnimelech, Y. 1999. Carbon nitrogen ratio as a control element in aquaculture systems. Aquaculture **176**:227-235.

Bianucci, L., K. Fennel, and K. L. Denman. 2012. Role of sediment denitrification in water column oxygen dynamics: comparison of the North American east and west coasts. Biogeoscience **9**:2673-2682.

Bilgmann, K., G. J. Parra, N. Zanardo, L. B. Beheregaray, and L. M. Moeller. 2014. Multiple management units of short-beaked common dolphins subject to fisheries bycatch off southern and southeastern Australia. Marine Ecology Progress Series **500**:265-U300.

Bulman, C., S. Condie, D. Furlani, M. Cahill, N. Klaer, S. Goldsworthy, and I. Knuckey. 2006. Trophic dynamics of the eastern shelf and slope of the South East Fishery: impacts of an on the fishery. Final Report to the Fisheries Research and Development Corporation (FRDC 2002/028). CSIRO Marine and Atmospheric Research, Hobart, Tasmania.

Bulman, C. M., S. A. Condie, F. J. Neira, S. D. Goldsworthy, and E. A. Fulton. 2011. The trophodynamics of small pelagic fishes in the southern Australian ecosystem and the implications for ecosystem modelling of southern temperate fisheries. FRDC Project 2008/023 Final Report. CSIRO, Hobart.

Cambridge, M. L. and P. J. Hocking. 1997. Annual primary production and nutrient dynamics of the seagrasses *Posidonia sinuosa* and *Posidonia australis* in south-western Australia. Aquatic Botany **59**:277-295.

Christensen, V. and D. Pauly. 1992. A guide to Ecopath II software system (version 2.1). ICLARM, Malyasia.

Christensen, V. and C. J. Walters. 2004. Ecopath with Ecosim: methods, capabilities and limitations. Ecological Modelling **172**:109-139.

Christensen, V., C. J. Walters, D. Pauly, and R. Forrest. 2008. Ecopath with Ecosim version 6 User Guide, November 2008. Lenfest Ocean Futures Project 2008.

Coll, M., A. Santojanni, I. Palomera, and E. Arneri. 2009. Food-web changes in the Adriatic Sea over the last three decades. Marine Ecology Progress Series **318**:17-37.

Collier, C. J., P. S. Lavery, R. J. Masini, and P. J. Ralph. 2007. Morphological, growth and meadow characteristics of the seagrass *Posidonia sinuosa* along a depth-related gradient of light availability. Marine Ecology Progress Series **337**:103-115.

Currie, D. R., D. C. D., S. D. Roberts, G. E. Hooper, S. J. Sorokin, and T. M. Ward. 2009. Fishery-independent by-catch survey to inform risk assessment of the Spencer Gulf Prawn Trawl Fishery. Report to PIRSA Fisheries. South Australian Research and Development Institue (Aquatic Sciences), Adelaide. SARDI Publication No. F2009/000369-1. SARDI Research Report Series No. 390. Publication No. F2009/000369-1, Research Report Series No. 390, South Autralian Research and Development Institute (Aquatic Sciences), Adelaide.

Currie, D. R. and S. J. Sorokin. 2010. A preliminary evaluation of the distribution and trophodynamics of demersal fish from Spencer Gulf. Report to the South Australian Department for the Environment and Heritage. South Australian Research and Development Institue (Aquatic Sciences), Adelaide. SARDI Publication No. F2010/000088-1. SARDI Research Report Series No. 424.

Department for Environment and Heritage. 2007. Nearshore marine benthic habitat mapping: information for NRM planning – South Australia. Department for Environment and Heritage, South Australia, Adelaide.

Dormann, C. F., J. M. McPherson, M. B. Araujo, R. Bivand, J. Bolliger, G. Carl, R. G. Davies, A. Hirzel, W. Jetz, W. D. Kissling, I. Kuhn, R. Ohlemuller, P. R. Peres-Neto, B. Reineking, B. Schroder, F. M. Schurr, and R. Wilson. 2007. Methods to account for spatial autocorrelation in the analysis of species distributional data: a review. Ecography **30**:609-628.

Doubell, M. J., C. E. James, P. D. van Ruth, J. Luick, and J. F. Middleton. 2013. Chapter 5. Modelling biogeochemical cycles in Spencer Gulf: Development of a nitrogen-based ecosystem model and implications for aquaculture.*in* J. Middleton, M. Doubell, C. James, J. Luick, and P. Van Ruth, editors. PIRSA Initiative II: carrying capacity of Spencer Gulf: hydrodynamic and biogeochemical measurement, modelling and monitoring. Final report for the Fisheries Research and Development Corporation. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2013/000311-1. SARDI Research Report Series No 705. 97 pp. . South Australian Research & Development Institute (Aquatic Sciences), Adelaide.

Duarte, C. M. and H. Kirkman. 2001. Methods for the measurement of seagrass abundance and depth distribution.*in* F. T. Short and R. G. Coles, editors. Global seagrass research methods. . Elsevier, Amsterdam, The Netherlands.

Econsearch. 2012. The economic impact of aquaculture on the South Australian state and regional economies, 2010/11. A report prepared for PIRSA Fisheries and Aquaculture. South Australia.

Fairhead, V. A. and A. C. Cheshire. 2004. Rates of primary productivity and growth in *Ecklonia radiata* measured at different depths, over an annual cycle, at West Island, South Australia. Marine Biology **145**:41-50.

Fennel, K., J. Wilkin, J. Levin, M. Moisan, J. O'Reilly, and D. Haidvogel. 2006. Nitrogen cycling in the Middle Atlantic Bight: Results from a three-dimensional model and implications for the North Atlantic nitrogen budget. Global Biogeochemical Cycles **20**: GB03007.

Fernandes, M., M. Angove, T. Sedawie, and A. Cheshire. 2007. Dissolved nutrient release from solid wastes of southern bluefin tuna (*Thunnus maccoyii*, Castelnau) aquaculture. Aquaculture Research **38**:388-397.

Fernandes, M. and J. E. Tanner. 2008. Modelling of nitrogen loads from the farming of yellowtail kingfish *Seriola lalandi* (Valenciennes, 1833). Aquaculture Research **39**:1328–1338.

Fowler, A. J., M. T. Lloyd, and D. Schmarr. 2009. A preliminary consideration of by-catch in the Marine Scalefish fishery of South Australia. SARDI Publication Number F2009/000097-1, SARDI Research Report Series No. 365.

Fulton, E. A. 2010. Approaches to end-to-end ecosystem models. Journal of Marine Systems **81**:171-183.

Gaylard, S. 2014. Marine pollution.*in* S. A. Shepherd, S. Madigan, B. M. Gillanders, S. Murray-Jones, and D. Wiltshire, editors. Natural history of Spencer Gulf. Royal Society of South Australia, Adelaide.

Goldsworthy, S. D., B. Page, P. Rogers, and T. Ward. 2011. Establishing ecosystem-based management for the South Australian Sardine Fishery: developing ecological performance indicators and reference points to assess the need for ecological allocations. FRDC Project 2005/031 Final Report. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2010/000863-1. SARDI Research Report Series No. 529. 173 pp.

Goldsworthy, S. D., B. Page, P. J. Rogers, C. Bulmand, A. Wiebkin, L. J. McLeay, L. Einoder, A. M. M. Baylis, M. Braley, R. Caines, K. Daly, C. Huveneers, K. Peters, A. D. Lowther, and T. M. Ward. 2013. Trophodynamics of the eastern Great Australian Bight ecosystem: Ecological change associated with the growth of Australia's largest fishery. Ecological Modelling **255**:38-57.

Griffiths, S. P., J. W. Young, M. J. Lansdell, R. A. Campbell, J. Hamption, S. D. Hoyle, A. Langley, D. Bromhead, and M. G. Hinton. 2010. Ecological effects of longline fishing and climate change on the pelagic ecosystem off eastern Australia. Reviews in Fish Biology and Fisheries **20**:239-272.

Hamer, D. J., T. M. Ward, and R. McGarvey. 2008. Measurement, management and mitigation of operational interactions between the South Australian Sardine Fishery and short-beaked common dolphins (*Delphinus delphis*). Biological Conservation **141**:2865-2878.

Hijmans, R. J. and J. van Etten. 2012. raster: Geographic analysis and modeling with raster data. R package version 2.0-08. http://CRAN.R-project.org/package=raster.

Hobbie, J. 1988. A comparison of the ecology of planktonic bacteria in fresh and salt water. Limnology and Oceanography **33**:750-764.

Howarth, R. W. and R. Marino. 2006. Nitrogen as the limiting nutrient for eutrophication in coastal marine systems: evolving views over three decades. Limnology and Oceanography **51**:364-376.

James, C., J. Middleton, J. Luick, and M. Doubell. 2013. Chapter 6. A graphical user interface: a tool for management of carrying capacity.*in* J. Middleton, M. Doubell, C. James, J. Luick, and P. Van Ruth, editors. PIRSA Initiative II: carrying capacity of Spencer Gulf: hydrodynamic and biogeochemical measurement, modelling and monitoring. Final report for the Fisheries Research and Development Corporation. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2013/000311-1. SARDI Research Report Series No 705. 97 pp. .

Kempton, R. A. and L. R. Taylor. 1976. Models and statistics for species diversity. Nature **262**:818-820.

Kirkman, H. 1989. Growth, density and biomass of *Ecklonia radiata* at different depths and growth under artificial shading off Perth, Western Australia. Australian Journal of Marine and Freshwater Research **40**:169-177.

Kirkman, H. 1997. Seagrasses of Australia. State of the Environment Technical Paper Series (Estuaries and the Sea). Department of the Environment, Canberra.

Larkum, A. W. D. 1986. A study of growth and primary production in *Ecklonia radiata* (C. Ag) J. Agardh (Laminariales) at a sheltered site in Port Jackson, New South Wales. Journal of Experimental Marine Biology and Ecology **96**:177-190.

Luick, J. L., M. Doubell, and J. F. Middleton. 2013. Chapter 1. A validated hydrodynamic model for Spencer Gulf and connectivity studies.*in* J. Middleton, M. Doubell, C. James, J. Luick, and P. Van Ruth, editors. PIRSA Initiative II: carrying capacity of Spencer Gulf: hydrodynamic and biogeochemical measurement, modelling and monitoring. Final report for the Fisheries Research and Development Corporation. South Australian Research and Development Institute (Aquatic Sciences), Adelaide.

SARDI Publication No. F2013/000311-1. SARDI Research Report Series No 705. 97 pp. . South Australian Research & Development Institute (Aquatic Sciences), Adelaide.

Messer, L. F., M. J. Doubell, T. C. Jeffries, M. V. Brown, and J. R. Seymour. 2014. Prokaryotic and diazotrophic population dynmaics within a large inverse estuary. Aquatic Microbial Ecology.

Middleton, J. F., M. Doubell, C. James, J. Luick, and P. Van Ruth. 2013. PIRSA Initiative II: carrying capacity of Spencer Gulf: hydrodynamic and biogeochemical measurement, modelling and monitoring. Final report for the Fisheries Research and Development Corporation. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2013/000311-1. SARDI Research Report Series No 705. 97 pp. . South Australian Research & Development Institute (Aquatic Sciences), Adelaide.

Mitra, A., C. Castellani, W. C. Gentleman, S. H. Jónasdóttir, K. J. Flynn, A. Bode, C. Halsband, P. Kuhn, P. Licandro, M. D. Agersted, A. Calbet, P. K. Lindeque, R. Koppelmann, E. F. Møller, A. Gislason, T. G. Nielsen, and M. St. John. in press. Bridging the gap between marine biogeochemical and fisheries sciences; configuring the zooplankton link. Progress in Oceanography.

Nixon, S. W. 1995. Coastal marine eutrophication - a definition, social causes, and future concerns. Ophelia **41**:199-219.

Nixon, S. W. and M. E. Q. Pilson. 1983. Nitrogen in estuarine and coastal marine ecosystems.*in* E. J. Carpenter and D. G. Capone, editors. Nitrogen in the Marine Environment. Elsevier, New York.

Novaczek, I. 1984. Development and phenology of *Ecklonia radiata* at 2 depths in Goat Island Bay, New Zealand. Marine Biology **81**:189-197.

Nunes, R. A. and G. W. Lennon. 1986. Physical property distribution and seasonal trends in Spencer Gulf, South Australia: an inverse estuary. Australian Journal of Marine and Freshwater Research **37**:39-53.

Nunes Vaz, R., G. Lennon, and D. G. Bowers. 1990. Physical behaviour of a large negative or inverse estuary. Continental Shelf Research **10**:277-304.

Page, B., S. D. Goldsworthy, L. McLeay, A. Wiebkin, K. Peters, L. Einoder, P. Rogers, M. Braley, S. Gibbs, J. McKenzie, C. Huveneers, R. Caines, K. Daly, S. Harrison, A. Baylis, M. Morrice, P. Gill, R. McIntosh, N. Bool, and T. Ward. 2011. The diets of marine predators in southern Australia: assessing the need for ecosystem-based management of the South Australian sardine fishery. Pages 31-60 in S. D. Goldsworthy, B. Page, P. Rogers, and T. M. Ward, editors. 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, PN 2005/031.SARDI Publication No. F2010/000863-1. SARDI Research Report Series No. 529. South Australian Research and Development Institute.

Paling, E. I. and A. J. McComb. 2000. Autumn biomass, below-ground productivity, rhizome growth at bed edge and nitrogen content in seagrasses from Western Australia. Aquatic Botany **67**:207-219.

Pauly, D., V. Christensen, J. Dalsgaard, R. Froese, and F. J. Torres. 1998. Fishing down marine food webs. Science **279**:860-863.

Pauly, D. and M. L. Palomares. 2005. Fishing down marine food webs: it is far more pervasive than we thought. Bulletin of Marine Science **76**:197-211.

Pikitch, E. K., C. Santora, E. A. Babcock, A. Bakun, R. Bonfil, D. O. Conover, P. Dayton, P. Doukakis, D. Fluharty, B. Heneman, E. D. Houde, J. Link, P. A. Livingston, M. Mangel, M. K. McAllister, J. Pope, and K. J. Sainsbury. 2004. Ecosystem-based fishery management. Science **305**:346-347.

Piroddi, C., G. Bearzi, and V. Christensen. 2010. Effects of local fisheries and ocean productivity on the northeastern Ionian Sea ecosystem. Ecological Modelling **221**:1526-1544.

Polovina, J. J. 1984. Model of a coral reef ecosystem. The ECOPATH model and its application to French Frigate Shoals. Coral Reefs **3**:1-11.

R Development Core Team. 2011. R: a language and environment for statistical computing. R Foundation for Statistical Computing., Vienna, Austria.

Roberts, S. D. and M. A. Steer. 2010. By-product assessment in the Spencer Gulf Prawn Fishery with an emphasis on developing management option for Balmain bugs. South Australian Research and Development Institue (Aquatic Sciences), Adelaide. SARDI Publication No. F2010/000165-1. SARDI Research Report Series No. 439.

Schendel, E. K., S. E. Nordstrom, and L. L. M. 2004. Floc and sediment properties and their environmental distribution from a marine fish farm Aquaculture Research **35**:483-493.

Schwarz, M. P. 2003. Explanatory notes for the LINCOLN 1:250 000 Geological Map. Primary Industries and Resources South Australia, Geological Survey of South Australia, Adelaide.

Shannon, L. J., S. Neira, and M. Taylor. 2008. Comparing internal and external drivers in the southern Benguela and the southern and northern Humboldt upwelling ecosystems. African Journal of Marine Science **30**:63-84.

Shaughnessy, P. D., S. D. Goldsworthy, and A. I. Mackay. 2014. Status and trends in abundance of New Zealand fur seal populations in South Australia. Final report to the Australian Marine Mammal Centre. SARDI Research Report Series No. 781. SARDI Aquatic Sciences, Adelaide.

Smith, S. and M. Atkinson. 1983. Mass balance of carbon and phosphorus in Shark Bay, Western Australia. Limnology and Oceanography **28**:625-639.

Smith, S. V. and H. H. Veeh. 1989. Mass balance of biogeochemically active materials (C, N, P) in a hypersaline gulf. Estuarine, Coastal and Shelf Science **29**:195-215.

Stelzenmueller, V., J. Lee, A. South, J. Foden, and S. I. Rogers. 2013. Practical tools to support marine spatial planning: A review and some prototype tools. Marine Policy **38**:214-227.

Ulanowicz, R. E. and C. J. Puccia. 1990. Mixed trophic impacts in ecosystems. Coenoses **5**:7-16.

van Ruth, P., P. Thompson, S. Blackburn, and P. Bonham. 2009. Temporal and spatial variability in phytoplankton abundance and community composition, and pelagic biogeochemistry in the tuna farming zone. Page 287 *in* J. E. Tanner and J. Volkman, editors. AquaFin CRC – Southern Bluefin Tuna aquaculture subprogram: Risk & response – Understanding the tuna farming environment. Technical report, Aquafin CRC Project 4.6, FRDC Project 2005/059. Aquafin CRC, Fisheries Research & Development Corporation and South Australian Research & Development Institute (Aquatic Sciences), Adelaide.

van Ruth, P. D. and M. J. Doubell. 2013. Chapter 4: Spatial and temporal variation in primary and secondary productivity and lower trophic ecosystem function in Spencer Gulf, South Australia.*in* J. Middleton, M. Doubell, C. James, J. Luick, and P. Van Ruth, editors. PIRSA Initiative II: carrying capacity of Spencer Gulf: hydrodynamic and biogeochemical measurement, modelling and monitoring. Final report for the Fisheries Research and Development Corporation. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2013/000311-1. SARDI Research Report Series No 705. 97 pp. . South Australian Research & Development Institute (Aquatic Sciences), Adelaide.

Walters, C., D. Pauly, and V. Christensen. 1999. Ecospace: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. Ecosystems **2**:539-554.

Walters, C. J., V. Christensen, and D. Pauly. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Reviews in Fish Biology and Fisheries **7**:139-172.

Watts, M. J., Y. X. Li, B. D. Russell, C. Mellin, S. D. Connell, and D. A. Fordham. 2011. A novel method for mapping reefs and subtidal rocky habitats using artificial neural networks. Ecological Modelling **222**:2606-2614.

Webster, M. A. and P. Petkovic. 2005. Australian bathymetry and topography grid, June 2005., Geoscience Australia Record (No 2005/12).

West, R. J. and A. W. D. Larkum. 1979. Leaf productivity of the seagrass, *Posidonia australis*, in eastern Australian waters. Aquatic Botany **7**:57-65.

11 Appendix A

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- Intellectual Property

This report will be made freely available and can be copied and distributed provided attribution of the work is made. The Spencer Gulf ecosystem model will also be available for modelling additional scenarios.

12 Appendix B

Description of functional groups, data sources, methods and assumptions in estimating parameters used in the Spencer Gulf Ecosystem model

Pinnipeds

Australian sea lion (ASL)

Biomass and consumption: Australian sea lions (*Neophoca cinerea*) are endemic to Australia and restricted to South and Western Australia, with over 85% of the species breeding in South Australia (Shaughnessy et al. 2011). About one third of the species' total population occurs within the Spencer Gulf Ecosystem (SGE) model area, within which there are eight breeding colonies (Dangerous Reef, English Island, Peaked Rock, North Islet, Albatross Island, Lewis Island, East Island and South Neptune Island) (Goldsworthy et al. 2010, Shaughnessy et al. 2011). There is little trend data available for most of the SGE colonies, with the exception of Dangerous Reef. Trend data from this site suggest that pup production was between 300-400 pups per breeding season during the 1990s; it then increased significantly through the early to mid -2000s peaking at ~830 in 2006/07, before declining somewhat thereafter (Goldsworthy et al. 2012). By combining surveys from other sites within the model area we have estimated total pup production in 1993 to be 627, peaking at 1,148 in 2006/07 and stabilising at 883 in 2013.

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 18 months (Shaughnessy et al. 2006), survival was calculated for every 1.5 years. Age-mass relationships for females and males followed those developed for the species by McIntosh (2007) and were used to estimate total biomass in 1993 at 184 tonnes, or a biomass density within the SGE of $B = 0.00636$ t km^{-2}.

A mass-based regression equation of field metabolic rate (FMR) based on seven otariid species developed by 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-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 and Gales 2000, Winship et al. 2002), with daily energy requirements at-sea being about 1.8 times greater than those on-shore ($ER_{on-shore}$) (Costa and 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 t y^{-1}) of age/sex classes as:

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

Using this approach prey consumption for the Australian sea lion population in the SGE areas was estimated (Q = 5,426.9 t/yr); with Q/B = 29.445 and P/B = 0.792. Production (P) per Biomass estimates (P/B) were estimated as: ((current biomass live + dead)/(previous year annual biomass alive)).

Diet: No definitive diet study of Australian sea lion in Spencer Gulf has been undertaken. Data were pooled from three main sources; Page et al. (2011) which drew heavily on the study of McIntosh et al. (2006); from unpublished faecal prey DNA studies (K. Peters unpublished data) and from observations of prey capture of Australian sea lion females from southern Spencer Gulf fitted with cameras (Goldsworthy unpublished data).

New Zealand fur seal

Biomass and consumption: New Zealand fur seals (*Arctocephalus forsteri*) are native to southern Australia and New Zealand. South Australia has most (>80%) of Australia's population (Goldsworthy and Page 2007), with the largest colonies occurring in the southern areas of the SGE model areas at North and South Neptune and Liguanea Islands (Goldsworthy and Page 2007). Estimates for the abundance of New Zealand fur seals in the SGE model area were based on pup production estimates obtained from these three sites in the 1989/90, 1992/93 and 1999/2000 breeding seasons (Shaughnessy et al. 1994, Shaughnessy et al. 1996, Shaughnessy and McKeown 2002), and over four consecutive breeding seasons between 2004/05 and 2007/08 (Goldsworthy and Page 2007, Goldworthy unpublished data). These data were used to provide estimates of annual changes in pup production between 1993 and 2013. Pup production across the three main colonies was estimated to be 5,592 in the 1993/94 year and based on an average annual increase of about 3.8%, 11,634 in 2012/13.

Adult female fur seals were not considered to forage within the SGE model area based on satellite tracking studies undertaken at both the Neptune and Liguanea Islands, indicating that almost all foraging is directed towards oceanic areas of the sub-tropical front between 400 and 1,100 km to the south west (Baylis et al. 2008, Baylis et al. 2012). Weaned pups and yearlings forage in oceanic waters (B. Page, A. Baylis and S. Goldsworthy unpublished data, Page et al. 2006) while adult males once reaching reproductive age (first male tenure average 9 years, McKenzie et al. 2007b) forage in continental slope waters (Page et al. 2006). In contrast, satellite tracking studies of juvenile and subadult males tracked from southern Spencer Gulf indicate that most of their foraging occurs within the Gulf and inner shelf regions, although animals tracked ranged extensively westward to the Nuyts Archipelago and eastward to Gulf St Vincent (B. Page and S. Goldsworthy unpublished data). For the purposes of the SGE model, we considered juvenile and sub-adult males aged between 2 and 8 years and juvenile females between 2 and 4 years (mean age of first reproduction in females is 5, McKenzie et al. 2007b) foraged within the model area.

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 $S = 0.627\text{-}0.073a + 0.003a^2\text{-}(5.91 \text{ x } 10^{-5})a^3$; males $S = 0.627\text{-}0.097a + 0.006a^2\text{-}(0.140 \text{ x } 10^{-3})a^3$), 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. 2007a). Age-mass relationships for females and males followed those developed for the species by McKenzie et al. (2007a), and were used to estimate the biomass (B) of juveniles and subadult males at 252 t (0.00870 t km^{-2}) in 1993/94 and 524 t (0.01810 t km^{-2}) in 2012/13.

Consumption and production estimates followed the methods described above for Australian sea lions. Prey consumption by juvenile and sub-adult male New Zealand fur seal population from breeding colonies within SGE areas was estimated (Q = 12,566 t yr^{-1}); with Q/B = 49.861 and P/B = 1.184. Production (P) per Biomass estimates (P/B) were estimated from the entire population as: ((current biomass live + dead)/(previous year annual biomass alive)).

Diet: Diet data for the New Zealand fur seal were based on 333 scat samples collected from sites containing juveniles and subadult males (B. Page unpublished data) from five sites within the SGE model area (Donnington Rock, Sibsey, Althorpe, Thistle and Liguanea Islands). Recovered prey hard parts (fish otoliths, cephalopod beaks, feathers) were identified and biomass reconstructed following the methods of Page et al. (2005). Given the high mobility of juvenile and sub-adult male New Zealand fur seals, it is unlikely that all of the foraging undertaken by animals originating from the Neptune and Liguanea Islands is restricted to the SGE model area, and it is also possible that some animals originating from other key population centres outside the SGE model areas (e.g. Kangaroo Island) also spend part of their time foraging within the region. We have estimated the amount of dietary input from regions outside of the SGE model area as 40%.

Cetaceans

Within Spencer Gulf, dolphins are the most common toothed cetacean species. The two main groups are the short-beaked common dolphin (*Delphis delphis*) and members of the bottlenose dolphin genus (*Tursiops* spp.). At least two species have potentially been documented from the region, the coastal Indo-Pacific bottlenose dolphin (*T. aduncus*) and common bottlenose dolphins (*T. truncatus*), which are predominantly distributed in shelf and oceanic waters further offshore than the Indo-Pacific bottlenose dolphin (Kemper and Ling 1991, Kemper 2004, Gibbs et al. 2011). More recently, a third species of bottlenose dolphin (the Southern Australian bottlenose dolphin/Burrunan dolphin) has been described from coastal regions of Victoria, South Australia and Tasmania (Moller et al. 2008, Charlton-Robb et al. 2011). There may also be hybridisation between species (Kemper 2004). Marked population structure has been found between coastal *Tursiops* sp. in Spencer Gulf and those in the Great Australian Bight that may be associated with the oceanography of the Gulf region (Bilgmann et al. 2007).

Common dolphin

Biomass and consumption: A recent aerial survey of common dolphin across both Spencer and Gulf St Vincent and the Investigator Straight in summer and winter 2011 by L. Moller, G. Parra and K. Bilgmann (Flinders University, unpublished data) formed the basis of estimates for common dolphin biomass and densities within the SGE model area. Using their strata 1 (south of Eyre Peninsula and eastern Backstairs Passage out to 100m contour) and strata 2 (Spencer Gulf) surveys areas, the mean summer and winter estimates were 14,183 dolphins (8,362 – 24,494 ±95% confidence limits, uncorrected for availability bias) giving a mean density of 0.4712 dolphins km^{-2}. The mean mass of *Delphinus* was estimated to be 79 kg (C. Kemper pers. comm.) giving overall estimates of biomass of 1,119 t or B = 0.03721 t km^{-2}.

Prey consumption was estimated using the methods presented by Barlow et al. (2008). 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:

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

where:

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

and 3900 and 5450 are the energy densities of crustaceans and fish, respectively (kJ kg^{-1} wet weight), Z is the fraction of crustaceans in the diet, 0.8 is the assimilation efficiency (Leaper and Lavigne 2007) and $\beta = 2.5$ (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 consumption 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 abundance (Barlow et al. 2008). Following this, we estimated the annual prey consumption of common dolphins to be 23,046 t yr^{-1}. This provides a *Q/B* estimate of 20.578. *P/B* was estimated at 0.09 for *Tursiops* based on Barlow and Boveng (1991).

Diet: The diets of common dolphins were assessed from stomach contents and stable isotope analyses of individuals that were found dead in southern Australia (Gibbs et al. 2011) and summaries compiled by Page et al. (2011). Common dolphins principally forage on pelagic fish such as sardines, anchovy and jack mackerel, and they also feed on bycatch and discards from the Spencer Gulf prawn fishery (Svane 2005). Svane (2005) estimated that dolphins consumed between 18 and 183 t of discards from the fishery per year.

Bottlenose dolphin

Biomass and consumption: Bottlenose dolphin abundance was based on the mean density of dolphins (0.0325 km^{-2}) based on aerial surveys conducted by Kemper et al. (2006), assuming that bottlenose dolphins made up ~40% of those dolphins surveyed (C. Kemper pers. comm.). The mean mass of *Tursiops* was estimated to be 109 kg (Barlow et al. 2008), giving an overall estimate of biomass of 102 t or B = 0.00354 t km^{-2}. Estimates of *Q/B* followed the same approach for common dolphins, providing an estimate of *Q/B* = 18.985. *P/B* was estimated at 0.08 based on Barlow and Boveng (1991).

Diet: The diets of bottlenose dolphins were assessed from stomach contents and stable isotope analyses of individuals that were found dead in southern Australia (Kemper and Gibbs 2001, Gibbs et al. 2011) and summaries compiled by Page et al. (2011).

Seabirds

Little penguin

Biomass and consumption: There has not been a systematic survey of little penguins (*Eudyptula minor*) in South Australia or within the SGE model area. Estimates used here were based on available summaries and estimates (Copley 1996, Wiebkin 2011b). 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 and 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 estimate of breeding pairs (5,300), the total population of little penguins in the SGE model area is estimated to be 15,443. Assuming a mean mass of 1.2 kg per bird, the total biomass of the population is estimated to be 18.5 t (B = 0.00128 t km^{-2}). Non-breeding (juvenile) little penguins were estimated to consume 73.1 kg per year, based on prey consumption of 167 g kg^{-1} D^{-1} (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 per year, 1.7 per pair) (Bethge et al. 1997). This provides an estimate of total annual prey consumption (Q) in the SGE model area of 4,587 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 and Antezana 2008).

Diet: Information on diet was based on that detailed for the Reevesby Island population in the Sir Joseph's Banks Group by Weibkin (2011). These included 156 stomach contents collected over six occasions in all seasons between 2003 and 2005 (Wiebkin 2011a).

Petrels

Biomass and consumption: The dominant petrel species in the SGE region are the abundant short-tailed shearwater (*Puffinus tenuirostris*) which breed across many of the islands in lower Spencer Gulf (Copley 1996); the far less abundant flesh-footed shearwater (*Puffinus carneipes*) which is only known to breed on two islands (Lewis and Smith Islands in lower Spencer Gulf (Copley 1996, Goldsworthy et al. 2013); and the widespread white-faced storm petrel (*Pelagodroma marina*) (Copley 1996).

Of these species, data on the breeding ecology, diet and at-sea distributions within the SGE 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 and 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 an estimate of the number of breeding pairs in the SGE region for short-tailed (136,318) and flesh-foot (3,300) shearwaters (Copley 1996, Goldsworthy et al. 2013), the total number of shearwater within the SGE region is estimated to be 520,872. Assuming a mean mass of 0.7 and 0.6 kg per bird, for short-tailed and flesh-foot shearwaters, respectively, the total biomass of shearwaters is estimated to be about 372.2 t.

The active (965.9 kJ d^{-1}) and resting (296.9 kJ d^{-1}) 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 SGE and adjacent waters, 55 days incubating the egg (incubation shared equally between the sexes) and 90 days rearing chicks (Weimerskirch and Cherel 1998, Einoder and 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 80,679 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 SGE). *Q/B* is estimated to be 147.3. A *P/B* estimate of 1.0 was derived from an estimate for Antarctic seabirds (Cornejo-Donoso and Antezana 2008).

The estimate of breeding pairs of white-faced storm petrels within the SGE region is 218,125 (Copley 1996). Assuming breeding pairs make up 2/3 of the population, the total estimate of the population is 660,985. White-faced storm petrels are estimated to be present in southern Australia between October and March which includes a 45 day incubation and 51 day chick rearing period (Marchant and Higgins 1990). Assuming a mean mass of 55 g (Marchant and 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^{-1}, and a mean meal mass fed to chicks of 6.4 g (0.5 meals per night) (Marchant and Higgins 1990); prey consumption per annum is estimated to be 23,628 t (using equations in Daunt et al. 2008). Import of prey consumption from outside the SGE was estimated to be 41.9%. Total population biomass is estimated to be 36.4 t, giving a *Q/B* estimated of 640.0. A *P/B* estimate of 1.0 was derived from an estimate for Antarctic seabirds (Cornejo-Donoso and Antezana 2008).

Parameters for the Ecopath model for the petrel group were combined to provide an overall estimate of biomass (409 t), biomass in the habitat area (0.00293 t km^{-2}) and consumption (78,105 t). Based on these values, *Q/B* was estimated to be 191.2. A *P/B* estimate of 1.0 was used based on Sakshaug (1997).

Diet: Diet data for short-tailed shearwaters was based on extensive studies undertaken in South Australia by Einoder (2010) and summarised by Page et al. (2011). The diet of white-faced storm petrels was based on that detailed for the species by Imber (1981). Dietary data for the petrel functional group was weighted for each species group based on their proportion to prey biomass consumed in the habitat area. Import of prey consumption from outside the SGE for all petrels combined was estimated to be 81.6%.

Australasian gannet

Biomass and consumption: The only breeding colony of Australasian gannets (*Morus serrator*) in South Australia is at Margaret Brock Reef off Cape Jaffa where approximately 300 breeding pairs nest (Lighthouses of Australia Inc 2004). Gannets are common in Spencer Gulf year-round, where they plunge-dive on small pelagic fish. They likely originate from the Margaret Brock Reef colony, as well as other breeding colonies in Victoria and Tasmania, which number approximately 6,660 pairs (Marchant and Higgins 1990). We estimated that 30% of the Margaret Brock Reef population and about 5% of the Victoria and Tasmania populations may be foraging within the SGE region at any time. With individual gannets weighing approximately 2.5 kg (Daunt et al. 2008), SGE gannet biomass is estimated to be 2.3 t (0.0000795 t km^{-2}). Estimates of the energy needs of breeding and non-breeding birds (4,561 KJ d^{-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 kJ g^{-1} (Bunce 2001), prey consumption was estimated using the formula of Daunt et al. (2008) to be 286 t. Based on these estimates, *Q/B* is 124.0, and a *P/B* estimate of 1.0 was used based on Sakshaug (1997).

Diet: Dietary data were based on (Bunce 2001) and summarised by Page et al. (2011).

Terns

Biomass and consumption: There are three resident (breeding) tern species that occur in the SGE area, they include the Caspian tern (*Hydroprogne caspia*), crested tern (*Sterna bergii*) and fairy tern (*Sternula nereis nereis*). As the crested tern breeds in large colonies, its biomass overwhelms the other species, for which there is limited information on their population size and ecology. The total population of crested terns (*Sterna bergii*) in the SGE (19,834 t) was estimated using data on the median number of breeding pairs (8,026, Copley 1996, Page et al. 2011) and assuming that adults make up 2/3 of the total population. Total biomass is estimated to be 6.7 t (0.0000233 t km^{-2} in the 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 kJ d^{-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 kJ g^{-1} (Chiaradia et al. 2002, Daunt et al. 2008), total prey consumption was estimated at 611 t yr^{-1}. Based on these estimates, *Q/B* is 90.7. A *P/B* estimate of 1.0 was used based on Sakshaug (1997).

Diet: Dietary data were based on studies undertaken in South Australia by McLeay et al. (2009) and summarised in Page et al. (2011).

Shags and cormorants

Biomass and consumption: There are four species of cormorant that occur in the SGE (little pied, little black, pied and black-faced), but only two of these forage away from the coastal fringe, the pied cormorant (*Phalacrocorax varius*) and the marine black-faced shag (*Phalacrocorax fuscescens*). Both species are winter breeding and nest in colonies. Abundance of each species in the SGE region was based on data presented in Copley (1996), (31,250 black-faced shags; 5,000 pied cormorants). Assuming a mean mass of 1.6 kg (Riordan and Johnston 2013), the estimated biomass is 58.0 t (*B* = 0.00020 t km^{-2}). Estimates of daily food consumption of 0.65 kg d^{-1} (outside chick-rearing period) and 0.836 kg d^{-1} (chick rearing x 90 days), assuming a prey calorific value of 5.03 kJ g^{-1} and an assimilation efficiency of 0.8 (Gomez-Laich et al. 2013), provides annual prey consumption estimates of 4,489.5 t, and a *Q/B* estimate of 77.4. A *P/B* estimate of 1.0 was used based on Sakshaug (1997).

Diet: there are no published data on the diets of black-faced shags or pied cormorants from South Australia or the SGE region. Information was taken instead from limited data available for black-faced shags in (Marchant and Higgins 1990), and from pied cormorants from studies in Western Australia and Queensland (Blaber and Wassenberg 1989, Humphries et al. 1992). The proportions of prey taxa were weighted for each species by estimated biomass, 0.86 for black-faced shag and 0.14 for pied cormorant.

Gulls

Biomass and consumption: There are two species of gull that occur in the SGE region, the silver gull (*Chroicocephalus novaehollandiae*) and the Pacific gull (*Larus pacificus*). In many parts of Australia, silver gull numbers have increased substantially with increases in human populations. Estimates of the size of gull populations were based on estimates provided within Copley (1996) and an estimate of ~3,300 pairs in 1999 in the Port Lincoln region (Farlam, unpublished data in Harrison 2009). The Port Lincoln population increased substantially to 10,300 pairs in 2003 (Harrison 2009). Earlier estimates were used to derive estimates for 1993 of 6,760 pairs of silver gulls, and 163 pairs of Pacific gulls in the SGE region. Based on an assumption that adults make up 40% of the population (Coulson et al. 1982), and with a mean estimated mass of 0.3 kg for silver gulls and 1.04 kg for Pacific gulls (Lindsay and Meathrel 2008) gives a combined biomass estimate of 4.4 t (or 0.0001517 t km^{-2}). An estimated daily energy requirement of 400 kJ d^{-1}, was used for silver gulls whereas those summarised by Lindsay and Meathrel (2008) were used for Pacific gulls. Based on these values, assimilation efficiency of 0.75, a mean prey density of 4.985 kJ g^{-1} (Goldsworthy et al. 2003), and the seabird consumptions models of Daunt et al. (2008), total prey consumption was estimated at 554.7 t yr^{-1}. Based on these estimates, *Q/B* is 126.2. A *P/B* estimate of 1.0 was used based on Sakshaug (1997).

Diet: The diet of silver gulls was based on data obtained from 108 samples for southern Spencer Gulf detailed in Harrison (2009), interpreted by (Page et al. 2011). No dietary information is available for Pacific gulls in South Australia. Data from Lindsay and Meathrel (2008) were used to infer the diet in the SGE region. Proportion of prey taxa was weighted for each species by estimated biomass, 0.923 for silver gull and 0.077 for Pacific gull.

Pelagic sharks

The pelagic shark community of the SGE region is made up of five main species, the white shark (*Carcharodon carcharias*), smooth hammer head shark (*Sphyrna zygaena*), common thresher (*Alopias vulpinus*) and bronze (*Carcharhinus brachyurus*) and dusky whaler (*C. obscurus*). There was scant incidental catch data for pelagic sharks in the SGE region, 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, were 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 were available for any component of this model group and this parameter was estimated by the model. The methods for estimating P/B, Q/B and EE for pelagic sharks are detailed below.

Production per biomass (P/B) can be approximated by the instantaneous total mortality rate Z (Allen 1971). The P/B values were therefore set equal to the total mortality rates $Z = F + M$, where F is the mean fishing mortality and M is the rate of natural mortality. Non-commercial species were considered to have an $F = 0$.

The instantaneous natural mortality rate (M) was preferably taken from direct estimation. However, only a few direct estimates of instantaneous natural mortality rate have been calculated for chondrichthyans (e.g. Gruber et al. 2001, Heupel and Simpfendorfer 2002). Instead, indirect estimates of mortality were obtained through methods based on predictive equations of life history traits. Natural mortality was derived from the empirical model of Pauly (1980):

$$M = K^{0.65} \text{ x } L_{\infty}^{-0279} \text{ x } T^{0.463}$$

where K and L_{∞} (cm) refer to the curvature and asymptotic length parameters of the von Bertalanffy growth function, and T is the mean annual water temperature in Celsius.

Q/B was calculated according to the empirical regression of Christensen and Pauly (1992):

$$Q/B = 10^{6.37} \text{ x } 0.0313^{Tk} \text{ x } W_{\infty}^{-0.168} \text{ x } 1.38^{Pf} \text{ x } 189Hd$$

where W_{∞} is the asymptotic body weight in grams, calculated from L_{∞} using published length-weight regressions; Tk is the mean annual temperature expressed as 1000/(T˚C + 2.731); Pf equals one for predators and zooplankton feeders and zero for others; and Hd equals one for herbivores and zero for carnivores. W_{∞} was calculated according to the equation $W_{\infty} = a \text{ x } L_{\infty}^{b}$. Length-weight data were usually available from the area from which L_{∞} was estimated.

The von Bertalanffy growth parameters were taken from the most recent studies in Australia or New Zealand. When no studies from these areas were available, the arithmetic mean of the most recent studies from other locations was used. When available, growth parameters for combined sexes were used. Otherwise, the arithmetic mean between male and female growth parameters was used. An ecotrophic efficiency estimate of 0.95 was used.

White shark

Production per biomass and consumption: The white shark (*Carcharodon carcharias*) is a wide ranging but mostly temperate and coastal species; it has a global distribution and at times occurs in oceanic environments, in the tropics, and down to depths of at least 1280 m. It is most common over the continental shelf (often close inshore) of southern Australia. Within the SGE, white sharks are often concentrated around seal colonies including Dangerous Reef and the Neptune Islands, where they can be observed year round (Strong et al. 1996; Bruce et al. 2005). White sharks of all sizes, from less than 2 m to over 5 m, occur in areas where snapper are abundant. Small specimens (< 2 m) are commonly encountered between Streaky Bay and the Head of the Bight. Fur seal and sea lion colonies are important locations for sub-adult and adult sharks in South Australia, Victoria and Tasmania.

P/B was assumed equivalent to *M* and calculated using Pauly's (1980) empirical equation. The von Bertalanffy growth parameters for species combined from three studies were used to estimate *M* (Cailliet et al. 1985, Wintner and Cliff 1999). A temperature of 17.5˚C was assumed for the mean annual water temperature as white sharks are most often found in temperatures between 15 and 20˚C (Carey et al. 1982, Casey and Kohler 1992, Klimley et al. 2002, Dewar et al. 2004, Weng et al. 2005). *Q/B* was estimated using Christensen and Pauly's (1992) equation with W_∞ estimated by the length-weight regressions for combined sexes from Australia (Malcolm et al. 2001). *P/B* and *Q/B* estimates were then averaged across sexes and studies as 0.10 and 1.73, respectively.

Diet: The diet of white shark was based primary on the study from South Africa by Hussey et al. (2012), using their largest size class (> 2.85m) as most sharks in the SGE are 3m+ (P. Rogers pers. comm.), and adjusting for local equivalent prey species (P. Rogers and C. Huveneers unpublished data). The undifferentiated elasmobranch diet component was spread proportionally across other elasmobranchs.

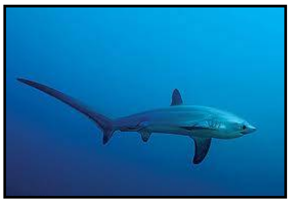
Smooth hammerhead

The smooth hammerhead (*Sphyrna zygaena*) is a wide-ranging shark with an amphitemperate distribution in, or close to, the continental shelf waters of all oceans (Compagno 1984). They are common in the SGE region.

P/B was assumed equivalent to *M* and calculated using Pauly's (1980) empirical equation. Von Bertalanffy growth parameters for each sex were taken from a study in Mexico by Garza (unpublished data). *Q/B* was estimated using Christensen and Pauly's (1992) equation with W_∞ estimated by the length-weight regression from Western Australia (McAuley and Simpfendorfer 2003). *P/B* = 0.21 and *Q/B* = 3.15 estimates were then averaged across sexes.

Diet: Dietary information was taken from the study of Rogers et al. (2012). A total of 39 stomachs (37, 95% stomachs contained prey items) were examined from samples collected from commercial catches in the Great Australian Bight (GAB) and Spencer Gulf between 2007 and 2010.

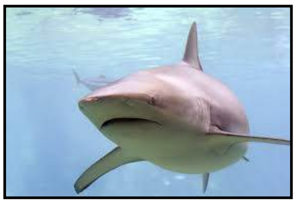
Thresher shark



The common thresher shark (*Alopias vulpinus*) is found both in coastal and oceanic waters. Von Bertalanffy growth parameters for each sex were taken from the most recent age and growth study of *A. vulpinus* (Smith et al. in press). Q/B was estimated using Christensen and Pauly's (1992) equation with W_∞ estimated by the length-weight regression from the Northwest Atlantic (Kohler et al. 1996). $P/B = 0.2$ and $Q/B = 2.78$ estimates were averaged between sexes.

Diet: Dietary information was taken from the study of Rogers et al. (2012). A total of 27 stomachs (17, 63% stomachs contained prey items) were examined from samples in the GAB between 2007 and 2009.

Whaler sharks



Bronze (*Carcharhinus brachyurus*) and dusky whaler (*C. obscurus*) sharks are the most abundant pelagic sharks in the SGE region. P/B was assumed equivalent to M and calculated using Pauly's (1980) empirical equation. Von Bertalanffy growth parameters for combined sexes were taken from a study in South Africa by Walter and Ebert (1991) and a study in Western Australia by Simpfendorfer et al. (2002). The mean summer water temperature (19.2°C) was used to account for the higher summer abundance and temperature-related migrations of *C. brachyurus* in South Australian waters. Q/B was estimated using Christensen and Pauly's (1992) equation with W_∞ estimated by the length-weight regression of Cliff and Dudley (1992) and from Western Australia (J. Chidlow, pers. comm.). $P/B = 0.095$ and $Q/B = 2.61$ estimates were averaged across species and sexes.

Diet: Dietary information was taken from the study of Rogers et al. (2012). A total of 250 bronze whaler stomachs (162, 65% contained prey items) and 49 dusky shark stomachs (32, 65% contained prey items) were examined from samples collected in the SGE, Gulf St Vincent and GAB, between 2007 and 2010.

Demersal sharks

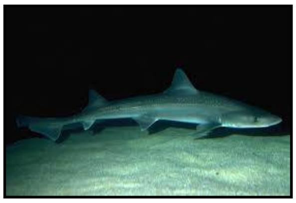
Demersal sharks were represented by three model groups within the SGE model: the Port Jackson shark (*Heterodontus portusjacksoni*); gummy shark (*Mustelus antarcticus*) and school shark (*Galeorhinus galeus*); and a third group composed of other demersal sharks that includes species such as wobbegongs or carpet sharks (Orecttolobidae), catsharks (Scyliorhinidae and Parascylliidae), angel sharks (Squatinidae), spurdogs and dogfish (Squalidae), sawsharks (Pristiophoridae) and elephant fish (Callorhinchidae).

Port Jackson shark

Port Jackson sharks (*Heterodontus portusjacksoni*) are abundant in the SGE, especially in the northern region of the Gulf (Currie and Sorokin 2010). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, using the upper SE estimates (0.0943966 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). Estimates of *P/B* = 0.25 were based on estimates of *M* (Froese and Pauly 2009) and *F* (0.15), and *Q/B* = 1.52 (Currie and Sorokin 2010).

Diet: Dietary data for Port Jackson shark were sourced from C. Beckmann (n=22, unpublished data) and Currie et al. (2010) (n=14), all samples were from Spencer Gulf.

Gummy & school shark



Gummy shark (*Mustelus antarcticus*) and school shark (*Galeorhinus galeus*) biomass estimates were based on mean annual catch of both species in Spencer Gulf between 1988 and 1999 and an estimated fishing mortality (*F*) during this period of 0.3 (*B* = 0.032388 t km^{-2}). Estimates of *P/B* = 0.515 were based on estimates of and *F* (0.3) and *Q/B* = 2.5 M (Froese and Pauly 2009).

Diet: There is limited diet information of these species. Dietary makeup was based on Currie et al. (2010) (n=1) and P. Rogers (pers. comm.).

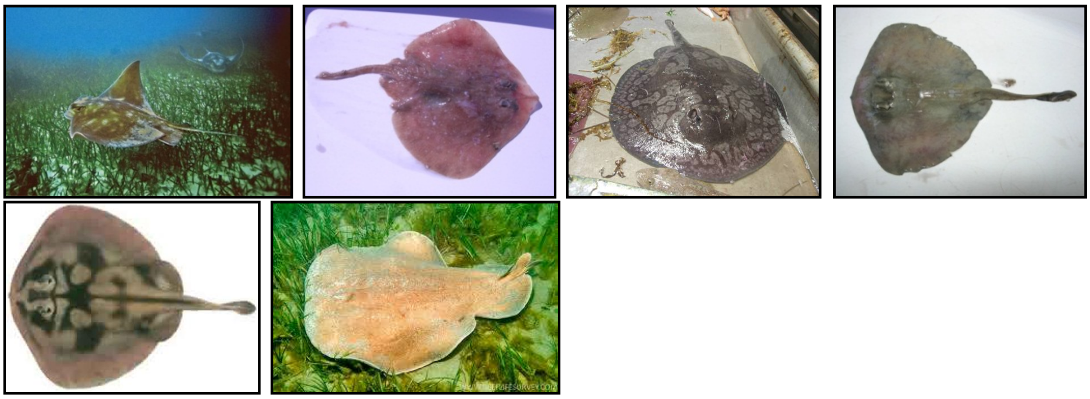
Other demersal sharks

The 'Other demersal sharks' group consisted of a range of species including the cobbler carpet shark (*Sutorectus tentaculatus*), ornate wobbegong (*Orectolobus maculatus*), saw shark (*Pristiophorus nudipinnis*), common sawshark (*Pristiophorus cirratus*), saddled catshark (*Asymbolus* sp.), Gulf catshark (*Asymbolus vincenti*), rusty catshark (*Parascyllium ferrugineum*) and angel shark (*Squatina australis*).

Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, based on the upper SE estimate for the 12 species (0.1086762 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates of 0.234 and 1.584 were based on Froese and Pauly (2009) and Currie (2009), respectively.

Diet: dietary data were based on the analyses of 17 shark stomachs (cobbler carpet shark = 1, rusty catshark = 3, angel shark =2, piked dogfish =1, saw shark = 5, elephant fish = 5), collected in Spencer Gulf, weighted for each species in proportion to their estimated biomass.

Skates & Rays



Most of the information on this group is based on 11 species that occur as bycatch in the western king prawn fishery of Spencer Gulf, for which some biological and abundance data are available (Currie et al. 2009, Currie and Sorokin 2010). These include the Melbourne skate (*Dipturus whitleyi*), southern fiddler ray (*Trygonorrhina fasciata*), southern shovelnose ray (*Aptychotrema vincentiana*), black stingray (*Dasyatis thetidis*), smooth stingray (*Dasyatis brevicaudata*), eagle ray (*Myliobatis australis*), coastal stingaree (*Urolophus orarius*), sparsely-spotted stingaree (*Urolophus paucimaculatus*), spotted stingaree (*Urolophus gigas*), banded stingaree (*Urolophus cruciatus*) and the Australian numbfish (Coffin Ray, *Hypnos monopterygium*).

Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, using upper SE estimate for the 11 species (0.3585778 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates of 0.234 and 1.757 were based on Froese and Pauly (2009) and Currie (2009), respectively.

Diet: dietary data were based on the analyses of 19 stomachs (eagle ray =1, southern fiddler ray = 1, coastal stingaree = 3, Melbourne skate = 1, sparsely-spotted stingaree = 11, southern shovelnose ray = 1, Australian numbfish =1) collected in Spencer Gulf, weighted for each species in proportion to their estimated biomass (Currie and Sorokin 2010).

The State managed (< 3nm from shore) component was mostly taken in the Marine Scalefish 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 are patchy and undoubtedly biased by the fact that most large Dasyatidae are released. There was limited discard information available for State fisheries that take this model group as bycatch, with the exception of 2007 when a dedicated bycatch program was implemented in State waters (Fowler et al. 2009).

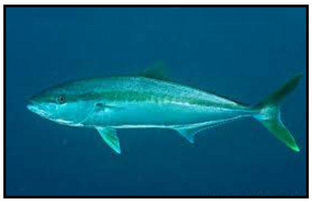
Teleosts

Southern bluefin tuna

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 and Young 1999). Most (99.6%) of the Australian SBT catch is taken in the eastern GAB 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).

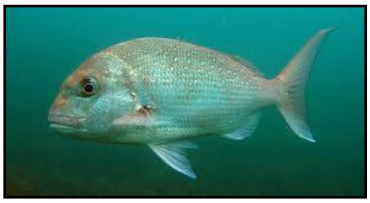
No biomass data were available, and this parameter was estimated by the model. *P/B* and *Q/B* estimates were 0.2 and 1.6 respectively (Bulman et al. 2006, Froese and Pauly 2009). Diet data were sourced from Caines (2005), Ward et al.(2006) and Page et al. (2011).

Yellowtail kingfish



Yellowtail kingfish (*Seriola lalandi*). No biomass data were available, and this parameter was estimated by the model. *P/B* and *Q/B* estimates were 0.2 and 3.2 respectively (Bulman et al. 2006, Froese and Pauly 2009). Diet data were sourced from Caines (2005) and Page et al. (2011).

Snapper



Snapper (*Chrysophrys auratus*) are an abundant, inshore, demersal fish species that occurs throughout temperate and sub-tropical waters of the Indo-Pacific region (Paulin 1990, Kailola et al. 1993). Snapper is the most valuable species of fish in the Marine Scalefish fishery in South Australia (Knight and Tsolos 2009). This fishery is a multi-gear, multi-species fishery that operates throughout all coastal waters of the State. Snapper are targeted with hand lines and long lines in this fishery (McGlennon et al. 2000). South Australia now has the highest State-based commercial catch in Australia (Fowler et al. 2010).

Key estimates of biomass, exploitation rate and recruitment are available for this species as part of dynamic, spatial age-length structure developed to facilitate management of this fishery. Biomass was estimated to be 0.2083 t km^{-2}. *P/B* and *Q/B* estimates were 0.493 and 3.8, respectively (Fulton and Smith 2004, Froese and Pauly 2009). Diet data were sourced from 2,056 stomach samples obtained across multiple seasons within the SGE region (M. Lloyd, unpublished data).

Snook

Snook (*Sphyraena novaehollandiae*) biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, using upper SE estimate for the 11 species (0.0035659 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 0.411 and 3.51, respectively (Fulton and Smith 2004, Currie and Sorokin 2010). Diet data were sourced from Caines (2005) and Page et al. (2011) based on 181 stomach samples.

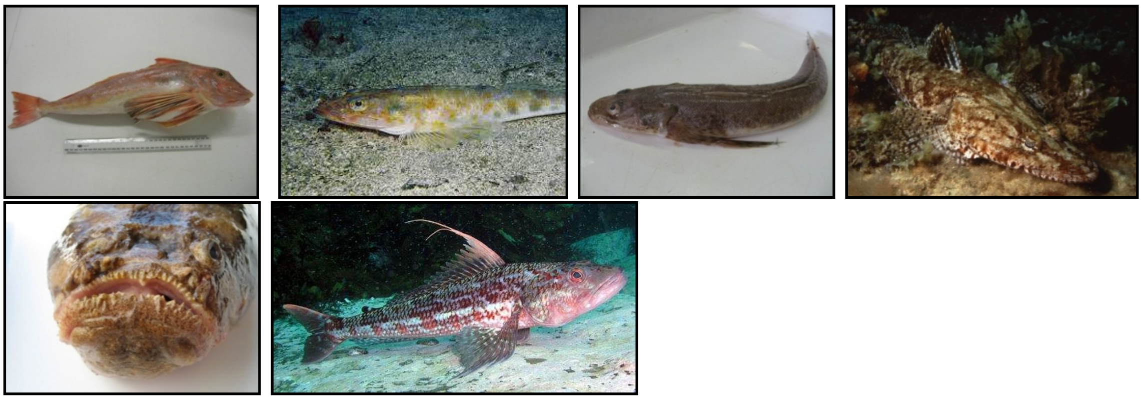
Barracouta

Biomass estimates of barracouta (*Thyrsites atun*) were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, using upper SE estimate (0.0029386 t.km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 0.411 and 3.64, respectively (Fulton and Smith 2004, Currie and Sorokin 2010). Diet data were sourced from Caines (2005) and Page et al. (2011) based on 71 stomach samples.

Skipjack trevally

Skipjack trevally (*Pseudocaranx wright*) is one the dominant consumers in the northern parts of Spencer Gulf, where it was estimated to eat approximately 21% of the total prey volume (Currie and Sorokin 2010). Skipjack trevally biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, using upper SE estimate (0.16815970 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 0.411 and 4.17, respectively (Fulton and Smith 2004, Currie and Sorokin 2010). Diet data were sourced from Currie and Sorokin (2010) based on 42 stomach samples collected from Spencer Gulf.

Medium demersal piscivores



The medium demersal piscivore group consists of eight main species that primarily consumed fish. This piscivorous fish group included red gurnard (*Chelidonichthys kumu*), a range of flathead species including tiger flathead (*Neoplatycephalus richardsoni*), toothy flathead (*Neoplatycephalus aurimaculatus*), yank flathead (*Platycephalus speculator*), and longhead flathead (*Leviprora inops*), common stargazer (*Kathetostoma leave*), red cod (*Pseudophycis bachus*), and sergeant baker (*Aulopus cf purpurissatus*). These species were grouped due to dietary similarities identified by Currie and Sorokin (2010). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, summing the upper SE estimate for all species (0.1078318 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 0.636 and 1.58 (mean of eight species), respectively (Fulton and Smith 2004, Currie and Sorokin 2010).

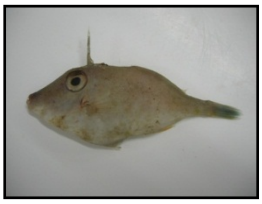
Diet: data were sourced from Currie and Sorokin (2010) based on 54 stomach samples collected from Spencer Gulf, weighted for each species in proportion to their estimated biomass (red gurnard =2, tiger flathead = 31, toothy flathead = 6, yank flathead = 6, longhead flathead = 1, common stargazer = 6, red cod = 1, sergeant baker = 1).

Large echinoderm feeding teleosts

This specialist group of fish was identified as one of the smallest fish guilds in Spencer Gulf (Currie and Sorokin 2010), being represented by two fish species, blue morwong (*Nemadactylus douglasii*) and short boarfish (*Parazanclistius hutchinsi*). These fish feed almost exclusively on ophiuroids and echinoids, and were therefore recognised as a discrete guild of echinoderm specialists (Currie and Sorokin 2010). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, summing the upper SE estimate for each species (0.0070316 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 0.625 and 2.34 (mean of eight species), respectively (Fulton and Smith 2004, Currie and Sorokin 2010).

Diet: data were sourced from Currie and Sorokin (2010) based on only 3 stomach samples (blue morwong =1, short boarfish = 2) collected from Spencer Gulf, weighted for each species in proportion to their estimated biomass.

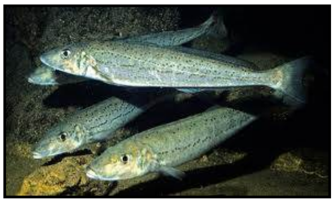
Degens/Rough leatherjackets




Degens leatherjacket (*Thamnaconus degeni*) represent the most abundant species collected during the trawl surveys in the Spencer Gulf Prawn fishery (Currie et al. 2009, Currie and Sorokin 2010). This small (< 20 cm) monacanthid was found at an average density of 627 per hectare, accounting for almost 70% of the total catch (Currie and Sorokin 2010). Whereas degens leatherjacket are abundant in the southern parts of Spencer Gulf, the rough leatherjacket (*Scobinichthys granulatus*), is the predominant monocanthid species in the northern parts of Spencer Gulf (Currie and Sorokin 2010). Both species have a wide dietary range, feeding on crustaceans, algae, molluscs, echinoderms, bryozoans, hydroids, ascidians and annelids (Currie and Sorokin 2010). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, summing the upper SE estimate for each species (1.2139581 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 0.900 and 2.26 (mean of two species), respectively (Fulton and Smith 2004, Currie and Sorokin 2010).

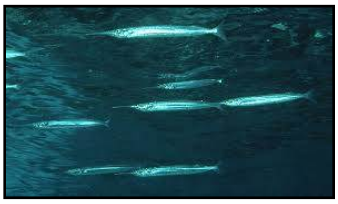
Diet: data were sourced from Currie and Sorokin (2010) based on the analyses of 62 stomach samples (degens leatherjackets =27, rough leatherjacket = 35) collected from Spencer Gulf, weighted for each species in proportion to their estimated biomass.

King George whiting



King George whiting (*Sillaginodes punctata*) are an important commercially and recreationally caught species in Spencer Gulf. Key estimates of biomass, exploitation rate and recruitment are available for this species as part of dynamic, spatial age-length structure model (WhiteEst), developed to facilitate management of this fishery (Fowler and McGarvey 2000, Fowler et al. 2011). Legal-size population biomass was used to estimate overall biomass for the SGE region as 0.06269 t km^{-2}. *P/B* and *Q/B* estimates were 0.548 and 2.29, respectively (Fulton and Smith 2004, Currie and Sorokin 2010). Diet data are limited for this species, and were sourced from Currie and Sorokin (2010) based on the analyses of 19 stomach samples from Spencer Gulf.

Southern sea garfish



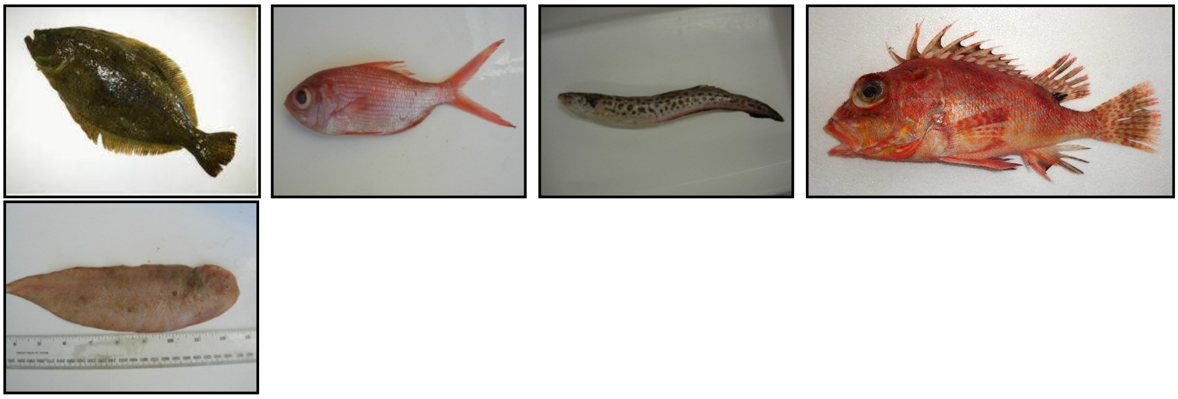
Southern sea garfish (*Hyporhamphus melanochir*) are an important commercially and recreationally caught species in Spencer Gulf. Key estimates of biomass, exploitation rate and recruitment are available for this species as part of dynamic, spatial age-length structure model (GarEst), developed to facilitate management of this species (McGarvey and Feenstra 2004, McGarvey et al. 2007). The GarEst model assesses garfish catch and effort data broken down into the four gear types (haul net targeting garfish, haul net non-targeting, dab net plus all other gears and recreations). Legal-size (fishable) population biomass was used to estimate overall biomass for the SGE region as 0.1150 t km^{-2}. *P/B* and *Q/B* estimates were 0.329 and 4.73, respectively (Fulton and Smith 2004, Currie and Sorokin 2010).

Diet data were based on the study by Earl et al. (2011), who examined 300 garfish from Gulf St Vincent.

Red mullet

Red mullet (*Upeneichthys vlamingii*) was the most widespread fish encountered in the Spencer Gulf, during standardised trawl surveys in the Spencer Gulf Prawn fishery, found at 93% (of 120) trawl stations (Currie et al. 2009, Currie and Sorokin 2010). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, using the upper SE estimate (0.0473865 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 0.790 and 4.4, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet data were sourced from Currie and Sorokin (2010) based on the analyses of 55 stomach samples collected from Spencer Gulf.

Medium crustacean teleosts



Medium crustacean teleosts were composed of a group of medium sized fishes whose diet was mostly composed of crustaceans, and included small tooth flounder (*Pseudorhombus jenynsii*), nannygai (*Centroberyx affinis*), gurnard perch (*Neosebastes pandus*), rock ling (*Genypterus tigerinus*) and southern tongue sole (*Cynoglossus br*oadhursti) (Currie and Sorokin 2010). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, summing the upper SE estimate for each species (0.0520051 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 0.546 and 2.97, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet data were sourced from Currie and Sorokin (2010) based on the analyses of 44 stomach samples (small tooth flounder =17, nannygai = 13, gurnard perch =4, rock ling =3, southern tongue sole =7), collected from Spencer Gulf, weighted for each species in proportion their estimated biomass.

Medium molluscan teleosts

Medium molluscan teleosts were composed of a group of medium sized fishes whose diet was mostly composed of molluscs, and included spikey globefish (*Diodon nicthemerus*), common stink fish (*Foetorepus calauropomus*), spotted stinkfish (*Repomucenus calcaratus*), beaked salmon (*Gonorynchus greyi*), senator wrasse (*Pictilabrus laticlavius*), fringed stargazer (*Ichthyscopus barbatus*), southern gobbleguts (*Vincentia conspersa*), and Chinaman leather jacket (*Nelusetta ayraudi*) (Currie and Sorokin 2010). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, summing the upper SE estimate for each species (0.05467 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 0.869 and 2.26, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet data were sourced from Currie and Sorokin (2010) based on the analyses of 54 stomach samples (spikey globefish = 17, common stink fish = 10, spotted stinkfish = 16, beaked salmon = 2, senator wrasse = 1, fringed stargazer = 2, southern gobbleguts = 3, and Chinaman leather jacket = 3), collected from Spencer Gulf, weighted for each species in proportion to their estimated biomass.

Small crustacean teleosts

Small crustacean teleosts were composed of a large group of at least 29 species of small sized fish (< 30 cm) whose diet was mostly composed of crustaceans (see Table 12.1), based on the study of (Currie and Sorokin 2010). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, summing the upper SE estimate for each species (0.09583 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 1.315 and 3.32, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet data were sourced from Currie and Sorokin (2010) based on the analyses of 230 stomach samples from 29 species (see table below). All were collected in Spencer Gulf; with prey biomass weighted for each species in proportion to their estimated biomass.

Small annelid teleosts

Small annelid teleosts comprised a large group of 13 species of small sized fish (<30 cm) whose diet was mostly composed of polychaetes (see Table 12.2, Currie and Sorokin 2010). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, summing the upper SE estimate for each species (0.13269 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 0.992 and 2.82, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet data were sourced from Currie and Sorokin (2010) based on the analyses of 133 stomach samples from 23 species (see Table 12.2). All were collected in Spencer Gulf; with prey biomass weighted for each species in proportion to their estimated biomass.

Table 12.1 Fish species and the number of stomach contents analysed within the small crustacean teleost group. Data from Currie and Sorokin (2010).

Family	Common name	Species	No. stomachs analysed
Aploactinidae	Deep Velvet fish	*Kanekonia queenslandica*	1
Apogonidae	Scarlet Cardinal fish	*Vincentia badia*	13
Apogonidae	Smooth Cardinal fish	*Vincentia macrocauda*	2
Clinidae	Southern Crested Weed Fish	*Cristiceps australis*	2
Enoplosidae	Old Wife	*Enoplosus armatus*	11
Monacanthidae	Four-spine Leather Jacket	*Eubalichthys quadrispinis*	1
Monacanthidae	Smoothspine Leather Jacket	*Cantheschenia longipinnis*	4
Monacanthidae	Sthn. Pygmy Leatherjacket	*Brachaluteres jacksonianus*	3
Neosebastidae	Gulf Gurnard Perch	*Neosebastes bougainvillii*	20
Neosebastidae	Little Gurnard Perch (little scorpion fish)	*Maxillicosta scabriceps*	27
Odacidae	Rainbow Cale	*Odax acroptilus*	5
Pegasidae	Sculptured Seamoth	*Pegasus lancifer*	7
Pempherididae	Common Bullseye	*Pempheris multiradiata*	5
Pempherididae	Slender Bullseye	*Parapriacanthus elongatus*	37
Pinguipedidae	Spotted Grubfish	*Parapercis ramsayi*	5
Pinguipedidae	Wavy Grubfish	*Parapercis haackei*	4
Pleuronectidae	Derwent Flounder	*Taratretis derwentensis*	2
Pleuronectidae	Spotted Flounder	*Ammotretis lituratus*	2
Serranidae	Barber Perch	*Caesioperca rasor*	5
Soleidae	Many Banded Sole	*Zebrias scalaris*	1
Tetraodontidae	Orange barred Puffer fish	*Polyspina piosae*	5
Tetraodontidae	Prickly Toadfish	*Contusus brevicaudus*	7
Tetraodontidae	Smooth Toadfish	*Tetractenos glaber*	1
Tetrarogidae	Soldier Fish	*Gymnapistes marmoratus*	5
Trachichthyidae	Roughy (Southern Roughy)	*Trachichthys australis*	3
Trachichthyidae	Sandpaper Fish	*Paratrachichthys macleayi*	7
Triglidae	Latchet	*Pterygotrigla polyommata*	5
Triglidae	Southern Shortfin Gurnard	*Lepidotrigla spinosa*	8
Triglidae	Spiny Gurnard	*Lepidotrigla papilio*	32

Table 12.2 Fish species and the number of stomach contents analysed within the small annelid teleost group. Data from Currie and Sorokin (2010).

Family	Common name	Species	No. stomachs analysed
Sillaginidae	Sthn. School Whiting (silver whiting)	*Sillago bassensis*	20
Monacanthidae	Toothbrush Leatherjacket	*Acanthaluteres vittiger*	36
Terapontidae	Striped Perch	*Pelates octolineatus*	6
Ostraciidae	Ornate Cowfish	*Aracana ornata*	6
Ostraciidae	Shaws Cowfish	*Aracana aurita*	7
Pempherididae	Rough bullseye	*Pempheris klunzingeri*	20
Monacanthidae	Bridled Leatherjacket	*Acanthaluteres spilomelanurus*	17
Monacanthidae	Mosaic Leatherjacket	*Eubalichthys mosaicus*	7
Bothidae	Crested Flounder	*Lophonectes gallus*	4
Monacanthidae	Velvet Leatherjacket	*Meuschenia scaber*	4
Chaetodontidae	Squareback Butterflyfish	*Chelmonops curiosus*	4
Monacanthidae	Gunn's Leatherjacket	*Eubalichthys gunnii*	1
Tetrarogidae	Goblin Fish	*Glyptauchen panduratus*	1

Syngnathids

The syngnathid group was composed of seven pipefish, seahorse and seadragon: tiger pipefish (*Filicampus tigris*), brushtail pipefish (*Leptoichthys fistularius*), leafy seadragon (*Phycodurus eques*), common seadragon (*Phyllopteryx taeniolatus*), spotted pipefish (*Stigmatopora argus*), bigbelly seahorse (*Hippocampus abdominalis*), Macleays crested pipefish (*Histiogamphelus cri*status). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, summing the upper SE estimate for each species (0.00085 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 1.000 and 4.70, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet data were sourced from Currie and Sorokin (2010), based on the analyses of 10 stomach samples from 4 species (common seadragon = 4, bigbelly seahorse = 3, leafy seadragon = 2, brushtail pipefish = 1). Most species consumed small crustaceans. All were collected in Spencer Gulf; with prey biomass weighted for each species in proportion to their estimated biomass.

Australian salmon

Australian salmon (*Arripis truttaceus*) are predominantly found in the Gulfs, inshore areas in shelf waters and around offshore islands. *P/B* and *Q/B* estimates were 0.45 and 4.7, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet data for *A. truttaceus* were sourced from (Caines 2005) and Page et al. (2011).

Historically, the commercial catch has mostly been extracted from 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.

Australian herring

Australian herring or tommy rough (*Arripis georgianus*) are predominantly found in the Gulfs, inshore areas in shelf waters and around offshore islands. *P/B* and *Q/B* estimates were 1.64 and 6.32, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Little is known about the diet of the species, although they are considered to predate mainly on small crustaceans/zooplankters (isopods and mysids), and juvenile fish (e.g. sprats, juvenile sardine and anchovy) (P. Rogers pers. comm.).

Historically, the commercial catch has mostly been extracted from State waters using purse seine nets, gill-nets, haul-nets and hand-lines, and 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.

Small pelagic fish

Jack/yellowtail mackerel

Jack (*Trachurus declivis*) and yellowtail (*Trachurus novaezelandiae*) mackerel are common small pelagic fish in the SGE regions. Biomass of these two species was estimated by the model. *P/B* and *Q/B* estimates were 0.52 and 5.37, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet information was only available for jack mackerel, based on the analyses of 40 stomach samples collected in South Australia (Daly 2007, Page et al. 2011).

Blue mackerel

Blue mackerel (*Scomber australasicus*) are a common small pelagic fish in the SGE region. Biomass of the species within the SGE region was estimated by the model. *P/B* and *Q/B* estimates were 0.52 and 5.37, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet information was only available for jack mackerel, based on the analyses of 40 stomach samples collected in South Australia (Daly 2007, Page et al. 2011)

Sardine

Sardine (*Sardinops sagax*), is an abundant small pelagic fish in the SGE region. 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 from 146,000 – 264,000 t (Ward et al. 2009b). Sardine biomass datasets collected by the Small Pelagic Fishes Subprogram of SARDI were available between 1995 and 2007. A biomass estimated for 1998 within Spencer Gulf was used as an estimate for 1991. *P/B* and *Q/B* estimates were 1.6 and 5.04, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet of sardines was based on the analyses of 218 stomach samples collected in South Australia (Daly 2007, Page et al. 2011).

Anchovy

Anchovy (*Engraulis australis*), are an abundant small pelagic fish in the SGE region. Recent DEPM based estimates of anchovy spawning biomass in SA Gulfs during the 2000 season was 25,374 t (9561 km^{-2}, survey area), providing a biomass estimate of 0.26535 t km^{-2} (upper 95%CL) (Dimmlich et al. 2009). *P/B* and *Q/B* estimates were 0.98 and 5.76, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet information was based on the analyses of 15 stomach samples collected in South Australia (Daly 2007, Page et al. 2011).

Sprats

The sprat group includes two clupeids, the sandy sprat (*Hyperlophus vittatus*) and the blue sprat (*Spratelloides robustus*). Biomass within the SGE region was estimated by the model. *P/B* and *Q/B* estimates were 1.80 and 5.76, respectively (Froese and Pauly 2009, Currie and Sorokin 2010). Diet information was based on the blue sprat, based on the analyses of 17 stomach samples from South Australia (Daly 2007, Page et al. 2011).

Finfish aquaculture

Farmed southern bluefin tuna

Southern bluefin tuna (SBT) farming began in 1991 as a result of a declining wild fishery, and a reduction in tuna supply prompted a move away from canning the product to a strategy of value-adding through farming (Harrison 2009). The SBT are mainly fed local or imported baitfish, with 60,000 tonnes distributed per annum (Harrison 2009). Habitat area was estimated as the total area of the SBT licences used for SBT production (4,238 km^2). Biomass estimate for 1991 (1,042 t) and time series was based on SARDI and industry data (B. Jeffreiss in litt); biomass in the habitat area was calculated as fished quotax(4/12)/area of production, assuming the fish spend on average 4 out of 12 months within the modelled area before being harvested (following methods detailed in Forrestal et al. 2012); *P/B* was estimated as fished quota/harvest; *Q/B* was estimated as feed/fished quota less 1.3% feed lost to birds (Harrison 2009), and 1% lost to water column (Aguado et al. 2004, in Forrestal et al. 2012). The feed/fished quota ratio of 11.6 t of feed per tonne of SBT was based on mean catch between 2000 – 2005 of 5,174 t and mean tonnage of feed of 60,000 t (Harrison 2009).

Farmed yellowtail kingfish

Juvenile yellowtail kingfish (YTK) fingerlings weighing ~5 grams are dispatched into sea cages on farms to grow out to a product of between 3 and 5 kg. YTK are hand fed on pellets which are specifically designed for production of the species. Habitat area was estimated as the area used in YTK aquaculture (3,208 km^2); *B*, marginal estimates for biomass were used for 1991, as the industry developed in the mid 2000s and production figures were only available between 2005 and 2011 (source PIRSA). *P/B* was estimated as harvest/stock; *Q/B* as feed/catch.

Cephalopods

The cephalopod group consists of southern calamary, giant Australian cuttlefish, ‘other squids’, and ‘octopuses’. All of these groups are commercially harvested. Calamary (*Sepioteuthis australis*) and giant Australian cuttlefish (*Sepia apama*) are targeted in the MSF fishery in Spencer Gulf. *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 and Pauly 2009). Diet data were sourced from Braley et al. (2010), Bulman et al. (2006), Grubert et al. (1999) and Page et al. (2011).

Southern calamary

Calamary (*Sepioteuthis australis*) are a common and commercially harvested cephalopod in the SGE. Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery (0.04129 t.km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 1.83 and 18.25, respectively (Officer & Parry 1996 in Fulton and Smith 2004). Diet of calamary was based on the analyses of 85 stomachs examined by both macro and molecular analyses (Roberts 2005, in Page et al. 2011).

Giant Australian cuttlefish

Biomass estimates for giant Australian cuttlefish (*Sepia apama*) were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery (0.02053 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 2.37 and 5.80, respectively (Loneragan et al. 2010). Little is known about the diet of giant Australian cuttlefish. The literature consistently suggests crustaceans > fish > molluscs and we used a ratio of 7:2:1 (M. Steer pers. comm.).

Other squids

The other squids group consists broadly of all other remaining squid taxa found in the SGE region. These included the Gould's (red arrow) squid (*Nototodarus gouldi*), Nova cuttlefish (*Sepia novaehollandae*), the southern bottletail squid (*Sepiadarium austrinum*), and striped pyjama squid (*Sepioloidea lineolata*). Biomass estimates for these species were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, with biomass data for all species combined (0.01190 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 1.80 and 17.50, respectively (Loneragan et al. 2010). Diet of this group was based around a study of Gould's squid diet which analysed the contents of 215 stomachs using both macro and molecular methods (Braley et al. 2010, Page et al. 2011).

Octopus

The octopus group consists of a range of octopus species (Octopodidae) found in the SGE region. Key taxa include: Maori octopus (*Octopus maorum*), southern keeled octopus (*Octopus berrima*), southern sand octopus (*Octopus kaurna*), and the southern hammer octopus (*Octopus australis*). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, for *O. australis* and *O. berrima* (0.00320 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 2.37 and 7.90, respectively (Loneragan et al. 2010). Diet was based on Grubert et al. (1999), and studies therein.

Crustaceans

Rock lobster

Southern rock lobster (*Jasus edwardsii*) is a major fishery for South Australia and is fished in southern Spencer Gulf as part of the North Zone management region for the fishery. Biomass for 1991 was estimated from the catch within the SGE (378.9 t) and assuming fishing mortality of 0.3 (0.087 t km^{-2}) with the SGE habitat fraction of 0.5. *P/B* and *Q/B* estimates were 0.73 and 12.41, respectively (Fulton and Smith 2004). Diet was based on the study of Hoare (2008).

Western king prawn

The western king prawn (*Melicertus latisulcatus*) has a wide distribution over the Indo-Pacific region with the world's largest known population occurring in Spencer Gulf. Biomass estimates were based on standardised trawl surveys undertaken in the Spencer Gulf Prawn fishery (0.57055 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 7.57 and 37.90, respectively (Ayers et al. 2013). There is little known about the diet of king prawns. King (1977) observed prawns feeding on algae and possibly bacteria films on the surfaces of seagrass and shells; prawns may also scavenge on small dead animals, and take live annelids. They are considered to be opportunistic scavengers.

Blue swimmer crabs

Blue swimmer crabs are one of the dominant crab species in Spencer Gulf. They have a predatory/scavenging lifestyle, feeding mainly on molluscs, crustaceans and polychaetes. Diet has not been studied in this species in South Australia; here we based diet on the study of Edgar (1990) and descriptions in Bryars and Svane (2008). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery (0.68529 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 2.80 and 8.50, respectively (Loneragan et al. 2010).

Sand crabs

Diet: little is known of the diet of sand crabs in Spencer Gulf. It is considered to have similar feeding behaviour and diet to the blue swimmer crab, with bivalves being a main prey item (Bryars and Svane 2008). Biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery (0.00057 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 2.80 and 8.50, respectively (Loneragan et al. 2010).

Large crabs

The large crab group consisted of at least 11 species from 6 families (Portunidae, Pilumnidae, Diogenidae, Dromiidae, Majidae and Xanthidae) and include (from most to least biomass contribution) the balmain bug (*Ibacus peronei*), rock crab (*Nectocarcinus integrifrons*), great spider crab (*Leptomithrax gaimardii*), bristle sponge crab (*Austrodromidia octodentata*), hairy shore crab (*Pilumnus* sp.), smooth seaweed crab (*Naxia aurita*), shaggy sponge crab (*Lamarckdromia globosa*), spider crab (*Naxia aries*), common hermit crab (*Paguristes frontalis*), facetted crab (*Actaea calculosa*), and the southern sponge crab (*Austrodromidia australis*) (Currie et al. 2009, Currie and Sorokin 2010). Biomass estimates for these species were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, with biomass data for all species combined (0.02408 t km^{-2}) (Currie et al. 2009, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 2.80 and 8.5, respectively (Loneragan et al. 2010). Diet was based on that detailed for large crabs in an ecopath model developed for Jurien Bay, WA (Loneragan et al. 2010).

Sand associated omnivorous crustaceans

Sand associated omnivorous (SAO) crustaceans were loosely based on the Jurien Bay Ecopath Model (Loneragan et al. (2010). This group consisted of smaller crab species, prawns (strawberry, Pandalid), shrimps and mysids. No biomass estimates were available for this group. *P/B*, *Q/B* and *EE* estimates were 0.79, 11.30 and 0.87, respectively (Loneragan et al. 2010). Diet was also based on Loneragan et al. (2010).

Herbivorous macrobenthos

Herbivorous macrobenthos included a range of echinoderms (starfish, sea urchins and holothurians), molluscs (chitons, sea slugs, gastropods). Diet was based on reef associated herbivore diet detailed in Loneragan et al. (2010), from the Jurien Bay Ecopath Model. No biomass estimates were available for this group. *P/B*, *Q/B* and *EE* estimates were 2.80, 14.00 and 0.6, respectively (Loneragan et al. 2010).

Sand, zoobenthos feeders

Sand, zoobenthos feeders were composed of a broad group of molluscs, especially bivalves including cockles (*Tawera lagopus*), bivalve cockle (*Solemya australis*), *Donax* sp., doughboy scallop (*Mimachlamys asperrima*), *Nuculana crassa*, queen scallop (*Equichlamys bifrons*), commercial scallop (*Pecten fumatus*), *Katelysia* sp., *Placamen flindersi*, *Phasionella australis*, *Lima vulgaris*, southern hammer oyster (*Malleus meridianus*), mud oyster (*Ostrea angasi*), razorfish (*Pinna bicolor*), and corbula clam (*Corbula coxi*). No biomass estimates were available for this group. *P/B*, *Q/B* and *EE* estimates were 0.65, 7.50 and 0.2, respectively (Loneragan et al. 2010). Diet was based on sand associated zoobenthos feeder diet detailed in Loneragan et al. (2010) from the Jurien Bay Ecopath Model.

Abalone

Five species of abalone occur in the SGE region, the greenlip (*Haliotis laevigata*), blacklip abalone (*H. rubra*), roe's abalone (*H. roei*), *H. scalaris* and *H. cyclobates*. Greenlip and blacklip abalone are the two major commercial species taken in SA. In the SGE, most of the fishery falls into the Central Zone management region for the fishery, with a small part of the Western Zone occurring in the south-western region of Spencer Gulf.

Greenlip abalone

No biomass estimates were available for this group. *P/B*, *Q/B* and *EE* estimates were 0.73, 12.41 and 0.90, respectively (Fulton and Smith 2004). Diet: greenlip abalone are estimated to consume 70% red algae, 11% brown algae, 15% seagrass and 4% detritus and browsed organic matter, based on diet studies at Tipara Reef, Spencer Gulf (Shepherd 1972).

Blacklip abalone

No biomass estimates were available for this group. *P/B*, *Q/B* and *EE* estimates were 0.73, 12.41 and 0.90, respectively (Fulton and Smith 2004). Diet: blacklip abalone are estimated to consume 55% red algae, 7% brown algae, 34% seagrass and 5% detritus and browsed organic matter, based on diet studies at Tipara Reef, Spencer Gulf (Shepherd 1972).

Small mobile crustaceans – deposit detritovore feeders

The small mobile crustacean (deposit detritovore feeders, DDF) were composed mainly of small crustaceans including caridean shrimps (e.g. *Ogyrides delli*), ghost shrimp (e.g. *Axiopsis werribee*), mantis shrimp (e.g. *Erugosquilla graham*), snapping shrimps, comma shrimps, sea lice (*Natatolana* spp.), isopods (e.g. *Cerceis* sp., *Cymodoce* sp., *Chitonopsis* sp., *Haswellia* sp.) and Pycnogonida (e.g.

Halicarcinus rostratus). No biomass estimates were available for this group. *P/B*, *Q/B* and *EE* estimates were 7.01, 27.14, 0.90, respectively (Loneragan et al. 2010). Diet information was based on deposit feeding invertebrates detailed in Loneragan et al. (2010) from the Jurien Bay Ecopath Model.

Small mobile crustacean zooplankton feeders (ZF)

This group was based on combined inshore pelagic zooplankton feeders and reef associated zooplankton feeders in Jurien Bay, detailed in Loneragan et al. (2010). It includes taxa such as krill (*Nyctiphanes australis*), Gammaridea and Photidea amphipods, copepods and ostracods. No biomass estimates were available for this group. *P/B*, *Q/B* and *EE* estimates were 1.12, 9.50 and 0.9, respectively (Loneragan et al. 2010). Diet was based on combined inshore pelagic zooplankton feeders and reef associated zooplankton feeders diet information detailed in the Jurien Bay Ecopath model (Loneragan et al. 2010).

Polychaetes – deposit detritovore feeders

This group consists of principally of deposit detritovore feeding polychaetes including trumpet worms (Pectinariidae), bobbit worms (Eunicidae), sea mice (Aphroditidae), rag worms (Nereidae), spaghetti worms (Terebellidae), Capitellidae, Cirratilidae, peanut worms (Sipuncula), spoon worms (Echiuroidea), Oligochaeta, horse-shoe worms (Phoronid) and acorn worms (Hemichordata). *P/B*, *Q/B* and *EE* estimates were 1.60, 6.00 and 0.9, respectively (Bulman et al. 2006). Diet was based on Mackinson and Daskalov (2008).

Sessile epifauna – zooplankton feeders

This group consist of zooplankton/phytoplankton feeding sessile epifauna, including some polychaetes such as beak-thrower worms (Glyceriidae), feather duster worms (Sabellidae); molluscs such as hairy mussel (*Trichomya hirsute*), the Pacific oyster (*Crassostrea gigas*); barnacles (Cirripedia); Brachiopoda; Bryozoa; Crinoidea; Ascidians (Ascidiacea); Didemnidae; Tunicates; Cnidaria; Gorgonacea; and Porifera. *P/B, Q/B* and *EE* estimates were 2.80, 11.80 and 0.91, respectively (Fulton and Smith 2004). Diet was based on Mackinson and Daskalov (Mackinson and Daskalov 2008).

Gelatinous zooplankton

Gelatinous zooplankton consisted of all the jellies, salps and ctenophores. Very little is understood about taxa within these groups within the SGE. Estimates of *P/B, Q/B* and *EE* used were 16.50, 80.00 and 0.00, respectively, based on those used for the Jurien Bay Ecopath model (Loneragan et al. 2010). Diet was based on Mackinson and Daskalov (2008).

Large zooplankton (carnivores)

The large zooplankton group consisted of krill (Nyctiphanes), copepods and amphipods. Biomass was estimated based on in-situ measurements within Spencer Gulf (0.3711 t km^{-2}) (M. Doubell in litt). Estimates of *P/B* and *Q/B* were 5.0 and 32.0 from Bulman et al. (2006) that were originally derived from studies in the Northern Hemisphere (Guenette and Morato 2002). Diet was based on Mackinson & Daskalov (2008).

Small zooplankton (herbivores)

Small zooplankton comprised copepods, pteropods and ostracods. Biomass was estimated based on in-situ measurements within Spencer Gulf (0.0989 t km^{-2}) (M. Doubell in litt). Estimates of *P/B* and *Q/B* were 29.5 and 55.0, and were based on those used for the Jurien Bay Ecopath model (Loneragan et al. 2010). Diet was based on Mackinson and Daskalov (2008).

Meiofauna

Meiofauna include small benthic organisms that live in sediments, loosely defined as a group of organisms by their size (larger than microfauna but smaller than macrofauna) generally able to pass through a 1 mm mesh but will be retained by a 40 µm mesh (Mackinson and Daskalov 2008). They include a variety of taxa such as nematodes, harpacticoid copepods, tubellarians, polychaetes, oligochaetes, ostracods, tardigrades, isopods, gastrotrichs, and kinorhynchs. Estimates of *P/B, Q/B* and *EE* were 35.00, 125.00 and 0.99, respectively, from Mackinson & Daskalov (2008). Diet was based on Mackinson and Daskalov (2008).

Primary Producers

Phytoplankton

Estimates on phytoplankton biomass and primary productivity (*P/B*) in the SGE were estimated from C^{14} incubations at 3 sites in Spencer Gulf over 4 seasons (van Ruth and Doubell 2013). Phytoplankton biomass was estimated to be 19.9 t km^{-2}. Primary productivity (*P/B*) was estimated to be 135.73.

Macroalgae and Seagrass

Details on estimates of habitat area, biomass in habitat area and *P/B* for macroalgae and seagrass are detailed in Table 3.2 of the report.

Microflora

A large part of primary production flows through the pool of dissolved organic matter (DOM), either after excretion by phytoplankton or through the lysis of ungrazed cells. This part of primary production is not available to herbivorous zooplankton and is mainly used by bacteria and auto/heterotrophic nanoflagellates that form a link between dissolved primary production and higher trophic levels (Mackinson and Daskalov 2008). The role of bacteria and auto/heterotrophic nanoflagellates in the SGE was based on that developed for the North Sea by Mackinson & Daskalov (2008). Essentially, organic matter produced from phytoplankton is portioned between three-concurrent pathways: 1) direct grazing by zooplankton; 2) incorporated into the microbial loop and 3) sedimentation and incorporation into benthic food chains. Characterisation of these pathways and the microbial loop in the SGE model was achieved and simplified by including heterotrophic flagellates (which prey on bacteria) and bacteria included in the same group, and enabling the group to feed on itself to represent flagellate-bacteria dynamics levels (Mackinson and Daskalov 2008). Importantly, this group and the dynamic it represents captures its role in utilising the primary production of phytoplankton from lysis and excretion that is not consumed by zooplankton (and higher trophic levels), representing the process of remineralisation where energy is fed back into the system to support production of higher trophic levels through the microbial loop (Mackinson and Daskalov 2008). In addition, by having two detritus groups (dissolved organic matter, DOM sediment; and particulate organic matter, POM) in the water column (DOM) and as sediment (POM), respectively, these two groups enable plankton and benthic bacteria to utilise their respective sources of organic matter. In the planktonic pathway, phytoplankton derived organic matter to be used by planktonic microflora are then eaten by zooplankton; whereas in the benthic pathway, dead ungrazed phytoplankton are used by benthic microflora that are then grazed by meiofauna and benthic microfauna (Mackinson and Daskalov 2008). Details on key parameters for each group are detailed below.

Benthic microflora

Biomass of benthic microflora (microphytobenthos) for the SGE was estimated to be 10 mg Chl m^{-2} and assuming a C:Chl ratio of 50:1 (Lauer et al. 2007), the biomass in units of carbon was estimated to be 0.5 t km^{-2} (Doubell in litt). Estimates of *P/B* and *Q/B* were 29,200 and 18,940, respectively, from Mackinson & Daskalov (2008). Diet was based on Mackinson & Daskalov (2008).

Planktonic microflora

Estimates of *P/B, Q/B* and *EE* were 571, 1142 and 0.99, respectively, from Mackinson & Daskalov (2008). Diet was based on Mackinson & Daskalov (2008).

Detritus

Four sources of detritus were estimated in the model, that from dissolved organic matter (DOM), particulate organic matter (POM), fish-farm feed, and fishery discard.

Detritus – DOM in water column

The biomass of DOM in the water column in the SGE was estimated from a biogeochemical model for the region to be 20.4 t km^{-2} (Doubell et al. 2013).

Detritus – POM in sediment

The biomass of POM in the sediment was based on Lauer et al. (2007), who show that ~ 2.7 % of sediment organic carbon was microphytobenthos. Assuming the remaining fraction is benthic detritus (benthic POM) then POM Biomass = 0.5*(1/0.027) = 18.5 t km^{-2}.

Fish farm feed

Habitat area was estimated as the combined total area of the SBT and YTK licences used for production (0.2178 of SGE area). Biomass of fish farm feed was estimated as the total annual SBT feed for 2001, 1.91 t km^{-2}.

Discards

Discards were calculated as the total fishery discards 0.4419 t km^{-2}.

Appendix B References

Ayers, M. J., Scharler, U. M. and Fennessy, S. T. (2013). Modelling ecosystem effects of reduced prawn recruitment on the Thukela Bank trawling grounds, South Africa, following nursery loss. Marine Ecology Progress Series, 479: 143-161.

Barlow, J. and Boveng, P. (1991). Modelling age-specific mortality for marine mammals populations. Marine Mammal Science, 7: 84-119.

Barlow, J., Kahru, M. and Mitchell, B. G. (2008). Cetacean biomass, prey consumption, and primary production requirments in te California Current ecosystem. Marine Ecology Progress Series, 371: 285-295.

Baylis, A. M. M., Page, B. and Goldsworthy, S. D. (2008). Colony-specific foraging areas of lactating New Zealand fur seals. Marine Ecology Progress Series, 361: 279-290.

Baylis, A. M. M., Page, B., McKenzie, J. and Goldsworthy, S. D. (2012). Individual foraging site fidelity in lactating New Zealand fur seals: Continental shelf vs. oceanic habitats. Marine Mammal Science, 28(2): 276-294.

Bethge, P., Nicol, S., Culik, B. M. and Wilson, R. P. (1997). Diving behaviour and energetics in breeding little penguins (*Eudyptula minor*). Journal of Zoology, London, 242: 483-502.

Bilgmann, K., Möller, L. M., Harcourt, R. G., Gibbs, S. E. and Beheregaray, L. B. (2007). Genetic differentiation in bottlenose dolphins from South Australia: association with local oceanography and coastal geography. Marine Ecology Progress Series, 341: 265-276.

Blaber, S. J. M. and Wassenberg, T. J. (1989). Feeding ecology of the piscivorous birds *Phalacrocorax varius*, *Phalacrocorax melanoleucos* and *Sterna bergii* in Moreton Bay, Australia - diets and dependence on trawler discards. Marine Biology, 101(1): 1-10.

Braley, M., Goldsworthy, S. D., Page, B., Steer, M. and Austin, J. J. (2010). Assessing morphological and DNA-based diet analysis techniques in a generalist predator, the arrow squid *Nototodarus gouldi*. Molecular Ecology Resources, 10: 466-474.

Bryars, S. and Svane, I. (2008). Chpter 35. Blue swimmer and sand crabs in Gulf St Vincent and Investigator Strait. In: Shepherd, S. A., Bryars, S., Kirkegaard, I., Harbinson, P. and Jennings, J. T. (Eds). Natural History of Gulf St Vincent. Royal Society of South Australia Inc, Adelaide, pp. 464-479.

Bulman, C., Condie, S., Furlani, D., Cahill, M., Klaer, N., Goldsworthy, S. and Knuckey, I. (2006). Trophic dynamics of the eastern shelf and slope of the South East Fishery: impacts of and on the fishery. Final Report to the Fisheries Research and Development Corporation (FRDC 2002/028). CSIRO Marine and Atmospheric Research, Hobart, Tasmania. pp 197.

Bunce, A. (2001). Prey consumption of Australiasian gannets (*Morus serrator*) breeding in Port Phillip Bay, southeast Australia, and potential overlap with commercial fisheries. ICES Journal of MArine Science, 58: 904-915.

Cailliet, G. M., Natanson, L. J., Welden, B. A. and Ebert, D. A. p. (1985). Preliminary studies on the age and growth of the white shark, *Carcharodon carcharias*, using vertebral bands. In: Sibley, G. (Ed.) Biology of the White Shark, a Symposium. Memoirs of the Southern California Academy of Sciences, Volume 9. Southern California Academy of Sciences, Los Angeles, vol. 9, pp. 49-60.

Caines, R. (2005). Trophic Guild-Structures in Pelagic Fish Communities in the Eastern Great Australian Bight Ecosystem. Honours Thesis. University of South Australia. 110 pp.

Carey, F. G., Kanwisher, J. W., Brazier, O., Gabrielson, G., Casey, J. G. and Pratt, H. L. J. (1982). Temperature and activities of white shark, *Carcharodon carcharias*. Copeia, 1982: 254-260.

Casey, J. G. and Kohler, N. E. (1992). Tagging studies on the shortfin mako (*Isurus oxyrinchus*) in the western North Atlantic. Australian Journal of Marine and Freshwater Research, 43(1): 45-60.

Charlton-Robb, K., Gershwin, L.-A., Thompson, R., Austin, J. J., Owen, K. and McKechnie, S. (2011). A new dolphin species, the Burrunan dolphin *Tursiops australis* sp. nov., endemic to southern Australian coastal waters. PLoS ONE, 6(9): e24047.

Chiaradia, A., Dann, P., Jessop, R. and Collins, P. (2002). The diet of Crested Tern (*Sterna bergii*) chicks on Phillip Island, Victoria, Australia. Emu, 102: 367-371.

Christensen, V. and Pauly, D. (1992). A guide to Ecopath II software system (version 2.1). ICLARM, Malaysia.

Cliff, G. and Dudley, S. F. J. (1992). Sharks caught in the protective gillnets off Natal, South Africa. 6. The copper shark *Carcharhinus brachyurus* (Gunther). South African Journal of Marine Science, 12: 663–674.

Compagno, L. J. V. (1984). Sharks of the world. An annotated and illustrated catalogue of shark species known to date. FAO, Rome, Italy. pp.

Copley, P. B. (1996). The status of seabirds in South Australia. The Status of Australia's Sea birds: Proceedings of the National Seabird Workshop, Canberra 1-2 November 1993.

Cornejo-Donoso, J. and Antezana, T. (2008). Preliminary trophic model of the Antarctic Peninsula Ecosystem (Sub-area CCAMLR 48.1). Ecological Modelling, 218: 1-17.

Costa, D. P., Dann, P. and Disher, W. (1986). Energy requirements of free ranging little penguin, *Eudyptula minor*. Comparative Biochemistry and Physiology, 85A: 135-138.

Costa, D. P. and Gentry, R. L. (1986). Reproductive energetics of the northern fur seal. In: Gentry, R. L. and Kooyman, G. L. (Eds). Fur Seals: Maternal Strategies on Land and at Sea. Princeton University Press, Princeton, New Jersey, pp. 79-101.

Costa, D. P. and Gales, N. J. (2000). Foraging energetics and diving behaviour of lactating New Zealand sea lions, *Phocarctos hookeri*. Journal of Experimental Biology, 203: 3655-3665.

Coulson, J. C., Duncan, N. and Thomas, C. (1982). Changes in the breeding biology of the herring gull (*Larus argentatus*) induced by reduction in the size and density of the colony. Journal of Animal Ecology, 51(3): 739-756.

Currie, D. R., Dixson, C. D., Roberts, S. D., Hooper, G. E., Sorokin, S. J. and Ward, T. M. (2009). Fishery-independent by-catch survey to inform risk assessment of the Spencer Gulf Prawn Trawl Fishery. Report to PIRSA Fisheries. South Australian Research and Development Institue (Aquatic Sciences), Adelaide. SARDI Publication No. F2009/000369-1. SARDI Research Report Series No. 390. 121 pp.

Currie, D. R. and Sorokin, S. J. (2010). A preliminary evaluation of the distribution and trophodynamics of demersal fish from Spencer Gulf. Report to the South Australian Department for the Environment and Heritage. South Australian Research and Development Institue (Aquatic Sciences), Adelaide. SARDI Publication No. F2010/000088-1. SARDI Research Report Series No. 424. 179 pp.

Daly, K. (2007). The diet and guild structure of the small pelagic fish community in the eastern Great Australian Bight, South Australia. Honours. Adelaide University. 42 pp.

Dann, P. and Cullen, J. M. (1990). Survival, Patterns of Reproduction and Lifetime Reproductive Success in Little Blue Penguins (*Eudyptula minor*) in Victoria, Australia. In: Davis, L. A. J. D. (Ed.) Penguin Biology. Academic Press, San Diego, pp. 63-84.

Dann, P., Carron, M., Chambers, B., Chambers, L., Dornom, T., McLaughlin, A., Sharp, B., Talmage, M. E., Thoday, R. and Unthank, S. (2005). Longevity in Little Penguins *Eudyptula minor*. Marine Ornithology, 33: 71-72.

Daunt, F., Wanless, S., Greenstreet, S. P. R., Jensen, H., Hamer, K. C. and Harris, M. P. (2008). The impact of the sandeel fishery closure on seabird food consumption, distribution, and productivity in the northwestern North Sea. Canadian Journal of Aquatic Sciences, 65: 362-381.

Dewar, H., Domeier, M. and Nasby-Lucas, N. (2004). Insights into young of the year white shark, *Carcharodon carcharias*, behavior in the Southern California Bight. Environmental Biology of Fishes, 70: 133-143.

Dimmlich, W. F., Ward, T. M. and Breed, W. G. (2009). Spawning dynamics and biomass estimates of an anchovy *Engraulis australis* population in contrasting gulf and shelf environments. Journal of Fish Biology, 75(7): 1560-1576.

Doubell, M., James, C., van ruth, P., Luick, J. and Middleton, J. (2013). Modelling biogeochemical cycles in spencer Gulf: development of a nitrogen based ecosystem model and implications for aquaculture. In Middleton et al. (2013). PIRSA Initiative II: Carrying Capacity of Spencer Gulf: Hydrodynamic and Biogeochemical Measurement Modelling and Performance Monitoring. Final report for the Fisheries Research and Development corporation. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2013/000311-1. SARDI Research Report Series No. 705. 96pp.

Earl, J., Fowler, A. J. and Dittmann, S. (2011). Temporal variation in feeding behaviour and trophic ecology of the temperate hemiramphid, *Hyporhamphus melanochir*. Environmental Biology of Fishes, 90: 71-83.

Edgar, G. J. (1990). Predator-prey interactions in seagrass beds. 2. Distriburtion and diet of blue mann crabs *Portunus pelagicu*s Linnaeus at Cliff Head, Western Australia. Journal of Experimental Marine Biology and Ecology, 139(1-2): 23-32.

Einoder, L. (2010). The foraging ecology of the short-tailed shearwater *Puffinus tenuirostris* PhD. University of Adelaide. pp.

Einoder, L. D. and Goldsworthy, S. D. (2005). Foraging flights of short-tailed shearwaters (*Puffinus tenuirostris*) from Althorpe Island: Assessing their use of neritic waters. Transactions of the Royal Society of South Australia, 129: 209-216.

Forrestal, F., Coll, M., Die, D. J. and Christensen, V. (2012). Ecosystem effects of bluefin tuna *Thunnus thynnus* aquaculture in the NW Mediterranean Sea. Marine Ecology Progress Series, 456: 215-231.

Fowler, A. J. and McGarvey, R. (2000). Development of an integrated fisheries managment model for King George Whiting (*Sillaginodes punctata*) in South Australia. Final report to FRDC for project 95/008. 232 pp.

Fowler, A. J., Lloyd, M. T. and Schmarr, D. (2009). A preliminary consideration of by-catch in the Marine Scalefish fishery of South Australia. SARDI Publication Number F2009/000097-1, SARDI Research Report Series No. 365. 79 pp.

Fowler, A. J., McGarvey, R., Steer, M. A. and Feenstra, J. E. (2010). The South Australian Marine Scalefish Fishery – Stock Status Report. Report to PIRSA. (PDF 664.4 KB). South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2007/000565-5. SARDI Research Report Series No. 507. 29pp.

Fowler, A. J., McGarvey, R., Burch, P. and Feenstra, J. E. (2011). King George Whiting (*Sillaginodes punctata*) Fishery. Fishery Assessment Report to PIRS Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2007/000843-3. SARDI Research Report Series No. 562. 82 pp.

Froese, R. and Pauly, D. (2009). FishBase. World Wide Web Electronic Publication, version (11/2009) www.fishbase.org. Available at www.fishbase.org.

Fulton, E. A. and Smith, D. M. (2004). Lessons learnt from a comparison of three ecosystem models for Port Phillip Bay, Australia. African Journal of Marine Science, 26: 219-243.

Gibbs, S. E., Harcourt, R. G. and Kemper, C. M. (2011). Niche differentiation of bottlenose dolphin species in South Australia revealed by stable isotopes and stomach contents. Wildlife Research, 38(4): 261-270.

Goldsworthy, S. D., Bulman, C., He, X., Larcombe, J. and Littnan, C. (2003). Trophic interactions between marine mammals and Australian fisheries: an ecosystem approach. In: Gales, N., Hindell, M. and Kirkwood, R. (Eds). Marine Mammals and Humans: Fisheries, tourism and management. CSIRO Publishing, Melbourne, pp. 69-99.

Goldsworthy, S. D. and Page, B. (2007). A risk-assessment approach to evaluating the significance of seal bycatch in two Australian fisheries. Biological Conservation, 139: 269-285.

Goldsworthy, S. D., Page, B., Shaughnessy, P. D. and Linnane, A. (2010). Mitigating seal interactions in the SRLF and the gillnet sector SESSF in South Australia. FRDC Project 2007/041 Final Report. SARDI Aquatic Sciences Publication Number F2009/000613-1, SARDI Research Report Series Number 405. 213 pp.

Goldsworthy, S. D., Page, B., Lowther, A. and Shaughnessy, P. D. (2012). Maintaining the monitoring of pup production at key Australian sea lion colonies in South Australia (2010/11). South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2010/000665-2. SARDI Research Report Series No. 601. 64pp.

Goldsworthy, S. D., Lavers, J., Carey, M. and Lowther, A. D. (2013). Assessment of the status of the Flesh- footed Shearwater in South Australia: population status, subspecies status and foraging ecology. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No.F2013/000013-1. SARDI Research Report Series No. 736. 9pp.

Gomez-Laich, A., Wilson, R. P., Shepard, E. L. C. and Quintana, F. (2013). Energy expenditure and food consumption of foraging Imperial cormorants in Patagonia, Argentina. Marine Biology, 160(7): 1697-1707.

Gruber, S. H., de Marignac, J. R. C. and Hoenig, J. M. (2001). Survival of juvenile lemon sharks at Bimini, Bahamas, estimated by mark-depletion experiments. Transactions of the American Fisheries Society, 130(3): 376-384.

Grubert, M. A., Wadley, V. A. and White, R. W. G. (1999). Diet and feeding strategy of *Octopus maorum* in southeast Tasmania. Bulletin of Marine Science, 65(2): 441-451.

Guenette, S. and Morato, T. (2002). The Azores Archipelago, 1997. In: Guenette, S., Christensen, V. and Pauly, D. (Eds). Fisheries impacts on North Atlantic Ecosystems: Modes and analysis. Fisheries Centre, University of British Columbia, Canada, pp. 241–247.

Gunn, J. and Young, J. (1999). Environmental determinants of the movement and migration of juvenile southern bluefin tuna. In: Gunn, J. and Block, B. (Eds). Fish Movement and Migration. Australian Society for Fish Biology, pp. 123-128.

Harrison, S. (2009). Interactions between Silver Gulls (*Larus novaehollandiae*) and Southern Bluefin Tuna (*Thunnus maccoyii*) aquaculture in the Port Lincoln area. PhD. Flinders University. 311 pp.

Heupel, M. R. and Simpfendorfer, C. A. (2002). Estimation of mortality of juvenile blacktip sharks, *Carcharhinus limbatus*, within a nursery area using telemetry data. Canadian Journal of Fisheries and Aquatic Sciences, 59: 624-632.

Hoare, M. J. (2008). The effects of southern rock lobster (*Jasus edwardsii*) translocation on inshore rocky reef communities in South Australia Honours. Flinders University. 92 pp.

Hooker, S. K., Whitehead, H. and Gowans, S. (2002). Ecosystem considerations in conservation planning: energy demand for foraging bottlenose whales (*Hyperoodon ampullatus*) in a marine protected area. Biological Conservation, 104: 51-58.

Humphries, P., Hyndes, G. A. and Potter, I. C. (1992). Comparisons between the diets of distant taxa (teleost and cormorant) in an Australian estuary. Estuaries, 15(3): 327-334.

Hunter, C. M., Moller, H. and Fletcher, D. (2000). Parameter uncertainty and elasticity analyses of a population model: setting research priorities for shearwaters. Ecological Modelling, 134(2-3): 299-323.

Hussey, N. E., McCann, H. M., Cliff, G., Dudley, S. F. J., Wintner, S. P. and Fisk, A. T. (2012). Size-based analysis of diet and trophic position of the white shark, *Carcharodon carcharias*, in South African waters. In: Domeier, M. L. (Ed.) Global perspectives on the biology and life history of the white shark.pp. 27-49.

Imber, M. J. (1981). Diets of storm petrels *Pelagodrama* and *Garrodia* and of prions *Pachyptila* (Procellariiformes). Proceedings of the symposium on birds of the sea and shore.

IUCN (2010). IUCN Red List of Threatened Species. Version 2010.2. Available at http://www.iucnredlist.org [accessed 17 August 2010].

Kailola, P. J., Williams, M. J., Stewart, P. C., Reichelt, R. E., McNee, A. and Grieve, C. (1993). Australian Fisheries Resources. Bureau of Resource Sciences and the Fisheries Research and Development Corporation, Canberra.

Kemper, C. and Ling, J. (1991). Whale strandings in South Australia (1881 - 1989). Transactions of the Royal Society of South Australia, 115: 37-52.

Kemper, C., Harcourt, R., Gibbs, S., Miller, C. and Wright, A. (2006). Estimating population size of 'at risk' bottlenose and common dolphins in Spencer Gulf and Gulf St Vincent, South Australia. Consultancy Report to Department of the Environment and Heritage, Canberra (May 2006). 41 pp.

Kemper, C. M. and Gibbs, S. E. (2001). Dolphin interactions with tuna feedlots at Port Lincoln, South Australia and recommendations for minimising entanglements. Journal of Cetacean Research and Management, 3(3): 283-292.

Kemper, C. M. (2004). Osteological variation and taxonomic affinities of bottlenose dolphins, *Tursiops* spp., from South Australia. Australian Journal of Zoology, 52(1): 29-48.

Kenney, R. D., Scott, G. P., Thompson, T. J. and Winn, H. E. (1997). Estimates of prey consumption and trophic impacts of cetaceans in the USA Northwest Continental Shelf Ecosystem. Journal of the Northwest Atlantic Fishery Science, 22: 155-171.

King, M. G. (1977). The biology of the of the western king prawn (*Penaeus latisulcatus*) and aspects of the fishery in South Australia. MSc Thesis, University of Adelaide.

Kirkwood, R., Lynch, M., Gales, N., Dann, P. and Sumner, M. (2006). At-sea movements and habitats of adult male Australian fur seals (*Arctocephalus pusillus doriferus*). Canadian Journal of Zoology, 84(12): 1781-1788.

Kleiber, M. (1975). The fire of life: an introduction to animal energetics. R. E. Krieger Publishing: Huntington, NY.

Klimley, A. P., Beavers, S. C., Curtis, T. H. and Jorgensen, S. J. (2002). Movements and swimming behavior of three species of sharks in La Jolla Canyon, California. Environmental Biology of Fishes, 63: 117-135.

Knight, M. and Tsolos, A. (2009). South Australian wild fisheries information and statistics report. (PDF 5.2 MB). SARDI Aquatic Sciences Publication No F2008/000804-1. SARDI Research Report Series No. 305. January 2009.

Kohler, N. E., Casey, J. G. and Turner, P. A. (1996). Length-Length and Length-Weight Relationships for 13 Shark Species from the Western North Atlantic, NOAA Technical Memorandum NMFS-NE-110.

Laidre, K. L., Heide-Jørgansen, M. P., Jorgensen, O. A. and Treble, M. A. (2004). Deep-ocean predation by a high Arctic cetacean. ICES Journal of MArine Science, 61: 430-440.

Lauer, P., Fernandez, M., Fairweather, P., Cheshire, A. C. and Tanner, J. (2007). Chapter 2: Spencer Gulf natural sedimentary setting. In: Fernandes, M., Lauer, P., Cheshire, A., Svane, I., Putro, S., Mount, G., Angove, M., Sedawie, T., Tanner, J., Fairweather, P., Barnett, J. and Doonan, A. (Eds). (2007). Aquafin CRC – Southern Bluefin Tuna Aquaculture Subprogram: Tuna Environment Subproject – Evaluation of Waste Composition and Waste Mitigation. Technical report, Aquafin CRC Project 4.3.2, FRDC Project 2001/103. (PDF 3.9 MB). SARDI Publication No. RD03/0037-9. SARDI Research Report Series No. 207 (Ed. M. Fernandes). Aquafin CRC, Fisheries Research & Development Corporation and South Australian Research & Development Institute (Aquatic Sciences), Adelaide. pp. 19-46.

Lavigne, D. M. (1996). Ecological interactions between marine mammals, commercial fisheries, and their prey: unravelling the tangled web. In: Montevecchi, W. A. (Ed.) High-lattitude seabirds 4. Trophic relationships and energetics of endotherms in cold ocean systems. Canadian Special Publication of Fisheries and Aquatic Sciences, vol. 9, pp. 59-71.

Leaper, R. and Lavigne, D. (2007). How much do large whales eat? Jornal of Cetacean Research and Managment, 9: 179-188.

Lighthouses of Australia Inc (2004). The Bulletin, No 1/2004.

Lindsay, M. C. M. and Meathrel, C. E. (2008). Where, When and How? Limitations of the Techniques Used to Examine Dietary Preference of Pacific Gulls (*Larus pacificus*) Using Non-consumed Parts of Prey and Regurgitated Pellets of Prey Remains. Waterbirds, 31(4): 611-619.

Loneragan, N. R., Babcock, R. C., Lozano-Montes, H. and Dambacher, J. M. (2010). Evaluating how food webs and the fisheries they support are affected by fishing closures in Jurien Bay, temperate Western Australia. FRDC Report, Project No. 2006/038.

Mackinson, S. and Daskalov, G. (2008). An ecosystem model of the North Sea to support an ecosystem approach to fisheries managment: description and parameterisation. Science Series Technical Reports. Cefas Lowestoft, 142: 196pp.

Malcolm, H., Bruce, B. D. and Stevens, J. D. (2001). A review of the biology and status of white sharks in Australian waters. Report to the Department of the Environment. pp 113.

Marchant, S. and Higgins, P. J. (Eds) (1990). Handbook of Australian, New Zealand and Antarctic Birds. Oxford University Press: Melbourne. pp.

McAuley, R. and Simpfendorfer, C. (2003). Catch composition of the Western Australian temperate demersal gillnet and demersal longline fisheries, 1994–1999; p. 78. Department of Fisheries; Perth, Western Australia. Fisheries Research Report, 146.

McGarvey, R. and Feenstra, J. E. (2004). Stock assessment models with graphical user interfaces for key South Australian marine finfish stocks. Australian Fisheries Research and Development Corporation Final Report No. 199/145. 176 pp.

McGarvey, R., Feenstra, J. E. and Ye, Q. (2007). Modelling fish numbers dynamically by age and length: partitioning cohorts into 'slices'. Canadian Journal of Fisheries and Aquatic Sciences, 64: 1157-1173.

McGlennon, D., Jones, G. K., Baker, J., Jackson, W. B. and Kinloch, M. A. (2000). Ageing, catch-at-age and relative year-class strength for snapper (*Pagrus auratus*) in northern Spencer Gulf, South Australia. Marine and Freshwater Research, 51: 669–677.

McIntosh, R. R., Page, B. and Goldsworthy, S. D. (2006). Dietary analysis of regurgitates and stomach samples from free-living Australian sea lions. Wildlife Research, 33: 661-669.

McIntosh, R. R. (2007). The life history and population demographics of the Australian sea lion, *Neophoca cinerea.* PhD Thesis. La Trobe University, Bundoora, Victoria. 367 pp.

McKenzie, J. (2006). Population demographics of New Zealand fur seals (*Arctocephalus forsteri*). PhD. La Trobe University. 166 pp.

McKenzie, J., Page, B., Goldsworthy, S. D. and Hindell, M. A. (2007a). Growth strategies of New Zealand fur seals in southern Australia. Journal of Zoology, 272: 377-389.

McKenzie, J., Page, B., Shaughnessy, P. D. and Hindell, M. A. (2007b). Age and reproductive maturity of New Zealand fur seals (*Arctocephalus forsteri*) in southern Australia. Journal of Mammalogy, 88(3): 639-648.

McLeay, L. J., Page, B., Goldsworthy, S. D., Ward, T. M. and Paton, D. C. (2009). Size matters: variation in the diet of chick and adult crested terns. Marine Biology, 156: 1765-1780.

Mcleay, L. J. (2010). The Life History Characters, Reproductive Constraints and Foraging Strategies of a Neritic Seabird, The Crested Tern. PhD. University of Adelaide. 182 pp.

Mecenero, S., Kirkman, S. P. and Roux, J.-P. (2006). A refined consumption model for lactating Cape fur seals (*Arctocephalus pusillus pusillus*), based on scat analyses. ICES Journal of Marine Science, 63: 1551-1566.

Moller, L. M., Bilgmann, K., Charlton-Robb, K. and Beheregaray, L. (2008). Multi-gene evidence for a new bottlenose dolphin species in southern Australia. Molecular phylogenetics and evolution, 49(2): 674-681.

Page, B., McKenzie, J. and Goldsworthy, S. D. (2005). Dietary resource partitioning among sympatric New Zealand and Australian fur seals. Marine Ecology Progress Series, 293: 283-302.

Page, B., McKenzie, J., Sumner, M. D., Coyne, M. and Goldsworthy, S. D. (2006). Spatial separation of foraging habitats among New Zealand fur seals. Marine Ecology Progress Series, 323: 263-279.

Page, B., Goldsworthy, S. D., 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. and Ward, T. (2011). The diets of marine predators in southern Australia: assessing the need for ecosystem-based management of the South Australian sardine fishery. In: Goldsworthy, S. D., Page, B., Rogers, P. and Ward, T. M. (Eds). 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, PN 2005/031.SARDI Publication No. F2010/000863-1. SARDI Research Report Series No. 529. South Australian Research and Development Institute, pp. 31-60.

Paulin, C. D. (1990). *Pagrus auratus*, a new combination for the species known as “snapper” in Australasian waters (Pisces: Sparidae). New Zealand Journal of Marine and Freshwater Research, 24: 259-265.

Pauly, D. (1980). On the interrelationship between natural mortality, growth parameters, and mean environmental temperature in 175 fish sticks. Journal du Conseil International pour l'Exploration de la Mer, 39(2): 175-192.

Riordan, J. and Johnston, G. (2013). Morphological Sex Determination in Black-faced Cormorants (*Phalacrocorax fuscescens*). Waterbirds, 36(1): 94-101.

Rogers, P., Huveneers, C., Page, B., Hamer, D., Goldsworthy, S. D., Mitchell, J. G. and Seuront, L. (2012). A quantitative comparison of the diets of sympatric pelagic sharks in gulf and shelf ecosystems off southern Australia. ICES Journal of MArine Science, 69: 1382–1393.

Sakshaug, E. (1997). Biomass and productivity distributions and their variability in the Barents Sea. ICES Journal of MArine Science, 54: 341-350.

Shaughnessy, P. D., Gales, N. J., Dennis, T. E. and Goldsworthy, S. D. (1994). Distribution and abundance of New Zealand fur seals, *Arctocephalus forsteri*, in South Australia and Western-Australia. Wildlife Research, 21(6): 667-695.

Shaughnessy, P. D., Stirling, I. and Dennis, T. E. (1996). Changes in the abundance of New Zealand fur seals, *Arctocephalus forsteri,* on the Neptune Islands, South Australia. Wildlife Research, 23(6): 697-709.

Shaughnessy, P. D. and McKeown, A. (2002). Trends in abundance of New Zealand fur seals, *Arctocephalus forsteri*, at the Neptune Islands, South Australia. Wildlife Research, 29(4): 363-370.

Shaughnessy, P. D., McIntosh, R. R., Goldsworthy, S. D., Dennis, T. E. and Berris, M. (2006). Trends in abundance of Australian sea lions, *Neophoca cinerea*, at Seal Bay, Kangaroo Island, South Australia. Sea Lions of the World, Anchorage, Alaska, USA.

Shaughnessy, P. D., Goldsworthy, S. D., Hamer, D. J., Page, B. and McIntosh, R. R. (2011). Australian sea lions *Neophoca cinerea* at colonies in South Australia: distribution and abundance, 2004 to 2008. Endangered Species Research, 13: 87-98.

Shepherd, S. A. (1972). Studies on southern Australian abalone (genus *Haliotis*) I. Ecology of five sympatric species. Australian Journal of Marine and Freshwater Research, 24: 217-257.

Simpfendorfer, C. A., McAuley, R., Chidlow, J. and Unsworth, P. (2002). Validated age and growth of the dusky shark, *Carcharhinus obscurus*, from Western Australian waters. Marine and Freshwater Research, 53: 567-573.

Svane, I. (2005). Occurrence of dolphins and seabirds and their consumption of by-catch during prawn trawling in Spencer Gulf, South Australia. Fisheries Research, 76(3): 317-327.

van Ruth, P. and Doubell, M. (2013). Spatial and temporal variation in primary and secondary productivity and lower trophic ecosystem function in Spencer Gulf. In Middleton et al. (2013). PIRSA Initiative II: Carrying Capacity of Spencer Gulf: Hydrodynamic and Biogeochemical Measurement Modelling and Performance Monitoring. Final report for the Fisheries Research and Development corporation. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2013/000311-1. SARDI Research Report Series No. 705. 96pp.

Walter, J. P. and Ebert, D. A. (1991). Preliminary estimates of age of the bronze whaler *Carcharhinus brachyurus* (Chondrichthyes: Carcharhinidae) from southern Africa, with a review of some life history parameters. South African Journal of Marine Science, 10: 37-44.

Ward, T. M., Mcleay, L. J., Dimmlich, W. F., Rogers, P. J., McClatchie, S., Matthews, R., Kampf, J. and van Ruth, P. D. (2006). Pelagic Ecology of a Northern Boundary Current System: Effects of Upwelling on the Production and Distribution of Sardine (*Sardinops sagax*), Anchovy (*Engraulis australis*) and Southern Bluefin Tuna (*Thunnus maccoyii*) in the Great Australian Bight. Fisheries Oceanography, 15: 191-207.

Ward, T. M., Burch, P., Mcleay, L. J. and Ivey, A. R. (2009a). Guidelines for using the daily egg production method for stock assessment of sardine, *Sardinop sagax*, off South Australia.SARDI Publication No. F2009/000249-1. SARDI Research Report Series No. 355. pp.

Ward, T. M., Ivey, A. R., Mcleay, L. J. and Burch, P. (2009b). Spawning biomass of sardine, *Sardinops sagax*, in waters off South Australia in 2009. Final Report to PIRSA Fisheries. South Australian Research and Development Institue (Aquatic Sciences), Adelaide. SARDI Publication No. F2007/000566-3.

Warham, J. (1996). The behaviour, population biology and physiology of the petrels. Acedemic Press: London, UK.

Weimerskirch, H. and Cherel, Y. (1998). Feeding ecology of short-tailed shearwaters: breeding in Tasmania and foraging in the Antarctic? Marine Ecology-Progress Series, 167: 261-274.

Weng, K. C., Castilho, P. C., Morrissette, J. M., Landeira-Fernandez, A. M., Holts, D. B., Schallert, R. J., Goldman, K. J. and Block, B. A. (2005). Satellite Tagging and Cardiac Physiology Reveal Nich Expansion in Salmon Sharks. Science, 310: 104-106.

Wiebkin, A. (2011a). Feeding and Breeding Ecology of Little Penguins (*Eudyptula minor*) in the Eastern Great Australian Bight. The University of Adelaide, South Australia. 191 pp.

Wiebkin, A. (2011b). Conservation managment priorities for little penguin populations in Gulf St Vincent. Report to the Adelaide and Mount Lofty Ranges Natural Resources Managment Board. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2011/000188-1. SARDI Research Report Series No. 588. 97 pp. .

Wilson, D., Curtotti, R., Begg, G. and Phillips, K. (2009). Fishery status reports 2008: status of fish stocks and fisheries managmed by the Australian Government. Bureau of Rural Sciences and Australian Bureau of Agriculture and Resource Economics: Canberra.

Winship, A. J., Trites, A. W. and Rosen, D. A. (2002). A bioenergetic model for estimating the food requirements of Steller sea lions *Eumatopias jubatus* in Alaska, USA. Marine Ecology Progress Series, 229: 291-312.

Wintner, S. P. and Cliff, G. (1999). Age and growth determinations of the white shark *Carcharodon carcharias*, from the east coast of South Africa. Fisheries Bulletin, 153: 153-169.

Wooller, R. D., Bradley, J. S., Skira, I. J. and Serventy, D. L. (1990). Reproductive success of short-tailed shearwaters *Puffinus tenuirostris* in relation to their age and breeding experience. Journal of Animal Ecology, 59(1): 161-170.

Young, J. W., Bradford, R. W., Lamb, T. D. and Lyne, V. D. (1996). Biomass of zooplankton and micronekton in the southern bluefin tuna fishing grounds off eastern Tasmania, Australia. Marine Ecology-Progress Series, 138(1-3): 1-14.

13 Appendix C

Table showing time series of taxa used for the Ecosim model. For some commercial taxa data for multiple gear types were used. Abbreviations: B, biomass; CPUE, catch per unit effort; DDF, deposit detritovore feeder; DN, dab net; F, fishing mortality; GN, gill net; HN, haul net; HL, hand line; LL, long line; PS, purse seine; SAO, sand-associated omnivore.

Group	Data type
Sardine	Effort
Prawn	Effort
Salmon PS	Effort
Salmon HN	Effort
Herring HN	Effort
Garfish HN	Effort
KGW HN	Effort
Snook HN	Effort
Other HN	Effort
Garfish DN	Effort
Snapper LL	Effort
Snapper HL	Effort
KGW HL	Effort
Other HL	Effort
Shark GN	Effort
Shark LL	Effort
Dropline	Effort
Calamary jig	Effort
Cuttlefish jig	Effort
Rock lobster	Effort
Blue crab trap	Effort
Sand crab net	Effort
Ocean jacket trap	Effort
Small mesh GN	Effort
Greenlip abalone	Effort
Blacklip abalone	Effort
Other	Effort
Sardine	Catch
Sardine	Biomass
Prawn	Catch
Salmon	Catch
Herring	Catch
Garfish	Catch
Garfish Fishing mortality	Fishing mortality
Garfish Biomass	Biomass
King George whiting	Catch
King George whiting F	Fishing mortality
King George whiting B	Biomass
Snook	Catch
Snapper	Catch
Snapper Fishing mortality	Fishing mortality
Snapper Biomass	Biomass
Blue crab	Catch

Calamary	Catch
Cuttlefish	Catch
Whaler sharks	Catch
Smooth hammerhead	Catch
Thresher shark	Catch
Gummy shark	Catch
School shark	Catch
Port Jackson shark	Catch
Other demersal sharks	Catch
Rays & skates	Catch
Yellowtail kingfish	Catch
Barracouta	Catch
Skipjack trevally	Catch
Medium piscivorous fish	Catch
Medium echinoderm fish	Catch
Red mullet	Catch
Silverbelly	Catch
Medium crustacean fish	Catch
Medium mollusc fish	Catch
Small crustacean fish	Catch
Degens/Rough leatherjacket	Catch
Small polychaete fish	Catch
Syngnathids	Catch
Blue mackerel	Catch
Jack/yellowtail mackerel	Catch
Anchovy	Catch
Sprats	Catch
Other squids	Catch
Octopus	Catch
Rock lobster	Catch
Western king prawn	Catch
Blue swimmer crab	Catch
Sand crab	Catch
Other large crabs/bugs	Catch
SAO crustaceans	Catch
Hebivorous macrobenthos	Catch
Sand-zoobenthos feeders	Catch
Greenlip abalone	Catch
Black abalone	Catch
Small mobile DDF crustaceans	Catch
Sessile epifauna	Catch
Macroalgae	Catch
Prawn	CPUE
Salmon	CPUE
Herring	CPUE
Garfish	CPUE
King George whiting	CPUE
Snook	CPUE
Snapper	CPUE
Blue crab	CPUE
Calamary	CPUE
Cuttlefish	CPUE

Snook	CPUE
Bronze whaler	CPUE
Gummy shark	CPUE
School shark	CPUE
Rays/skates	CPUE
Blacklip abalone	CPUE
Greenlip abalone	CPUE
Rock lobster	CPUE
New Zealand fur seal	Biomass
Australian sea lion	Biomass
Common dolphin bycatch	Fishing mortality
Australian sea lion bycatch	Fishing mortality
Farmed southern bluefin tuna	Biomass
Farmed Kingfish	Biomass
Fishfarm feed	Biomass