A trophic model for Gulf St Vincent:

Balancing exploitation of three fisheries in an EBFM framework

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Abbreviations

BRD	Bycatch Reduction Device
CPUE	Catch per unit effort
DDF	Deposit detritivore feeder
DEPM	Daily Egg Production Method
DEWNR	Department of Water, Environment and Natural Resources
DN	Dab net
DOM	Dissolved organic matter
EBFM	Ecosystem based fishery management
EE	Ecotrophic efficiency
F	Fishing mortality
FF	Forcing functions
FIB	Fishing in balance
GN	Gill net
GSV	Gulf St Vincent
HL	Hand line
HN	Haul net
LL	Long line
mTLC	Mean trophic level of the catch
MSF	Marine Scalefish Fishery
NGSV	Northern Gulf St Vincent
P	Phytoplankton
P/B	Production/biomass
PIRSA	Primary Industries and Regions South Australia
POM	Particulate organic matter
PS	Purse seine
SAO	Sand-associated ominvore
SBT	Southern bluefin tuna
SGSV	Southern Gulf St Vincent
SS	Sums of Squares
SST	Sea surface temperature
TL	Trophic level
Q/B	Consumption/biomass
YTK	Yellowtail kingfish
Z	Zooplankton
ZF	Zooplankton feeding

Executive Summary

What the report is about
This project has developed the first whole of ecosystem model for Gulf St Vincent (GSV), South Australia. It describes the key components of the ecosystem, and provides the first integrated assessment of the stock status, performance and impact of its key fisheries over a 20-year period (1994 to 2014). The model was principally developed to assess if increases in Snapper abundance in the late 2000s could have contributed to reduced production in crustacean fisheries (Blue Crab and Prawn) in GSV, and if so, whether a directed Snapper fishery could provide ecological and economic benefits. It also examined the ecological and production implications of greater selectivity of fishing methods in the GSV Prawn fishery. The GSV ecosystem model provides a basis for an ecosystem based management decision support tool for fisheries and other key activities in GSV.

Background
A dramatic switch was observed in the spatial structure of South Australia's Snapper fishery between 2008 to 2013, leading to a marked increase in Snapper biomass in GSV that coincided with a considerable downturn in the commercial harvests in the GSV Blue Crab and Prawn fisheries. As Snapper are known to predate upon Blue Crabs and Prawns, there was concern and uncertainty among fishers and managers that the downturn in these crustacean fisheries had occurred in part, due to increases in the biomass and predation pressure of Snapper. The potential interactions between these three high-value fisheries and their respective management has consequently generated a need to: a) better understand the trophic interactions that underpin their production; b) assess the potential impacts of increased Snapper biomass on Blue Crabs, Prawns and other key species, and c) assess if such impacts could be ameliorated by directed fishing of Snapper to a level that optimised production across the three fisheries while minimising ecosystem impacts.

An additional change in the GSV ecosystem, which has uncertain ecosystem implications, is the introduction of more selective fishing methods in the GSV Prawn fishery. Rigid-grid bycatch reduction devices (BRDs) and T90-mesh cod-ends were introduced in the GSV Prawn fishery in 2012. Use of BRDs has substantially reduced sponge, elasmobranch and fish bycatch, not only improving fishing efficiency, but enhancing the ability of the T90 net to exclude juvenile prawns, small fish and crustaceans from the catch. The expectation among fishers and managers has been that reduced bycatch will bring positive ecological benefits, but the potential ecosystem and fishery production consequences of greater gear selectivity are uncertain.

Aims/objectives
The objectives of the project were to develop a GSV ecosystem model to: 1) understand the impact of changes in the abundance of Snapper on the GSV ecosystem, with particular emphasis on other high value commercial fisheries, i.e. Prawns and Blue Crabs; 2) assess temporal change, the effects of fishing and improved fishing selectivity on the GSV ecosystem over the last 20 years; and 3) assess and optimise future ecological and economic performance of multi-species fisheries in an ecosystem based fishery management (EBFM) framework.

Methodology
A GSV ecosystem, trophic mass-balance model was developed using the *Ecopath with Ecosim* software. The *Ecopath* model was constructed for 1994 and the *Ecosim* model developed for a 20-year time period (1994-2013). The model area, which included all of GSV and part of the Investigator Strait, was calculated to be 10,500 km^2. The model incorporated 75 functional or trophic groups based on similarities in diet, habitat, foraging behaviour, size, consumption and rates of production, as well as 31 fishing fleets for which landings and effort data were available for the 20-year period. Key changes in the marine ecosystem assessed were total catch; the mean trophic level of the catch (mTLC); and the Fishing in Balance Index (FIB index). Two broad scenarios were examined using the model. The first examined the potential ecosystem implications of greater gear selectivity in the GSV Prawn fishery brought about by the introduction of BRDs and T90 nets in 2012; the second examined the potential impacts on the ecosystem and high value fisheries from different Snapper biomass scenarios.

Results/key findings

This project developed the first whole of ecosystem model for GSV that describes key components of the ecosystem, and an integrated assessment of the stock status, performance and impact of its key fisheries between 1994 and 2013. In terms of fisheries impacts, model results suggest there has been a general increase in the biomass of most trophic groups as a consequence of marked declines in fisheries catch and effort, over the 20-year period.

Analyses of key ecosystem indicators identified a relatively stable mean trophic level of the catch and FIB index, against a backdrop of declining catches of most key fisheries by the end of the 20-year period. Results suggest there may have been a net loss in production or productivity to the GSV ecosystem over this period that could be attributed to underestimated catch in the model (e.g. from commercial discards and/or recreational landings and discards), and/or a real loss of productivity as a consequence of environmental change and/or environmental degradation (e.g. pollution, habitat loss).

Scenarios comparing the ecosystem response to greater gear selectively in the GSV prawn fishery, supported the expectation that the introduction of BRDs/T90 cod-end mesh has resulted in positive ecological benefits (an increase in the biomass of demersal sharks and rays). Importantly, no negative impacts on other commercially targeted species were identified.

The ecological role of Snapper and its impact on crustacean and other fisheries was assessed. Snapper were found to be important predators in the GSV ecosystem, consuming a diverse range of prey across four trophic levels, with much of their predation directed to crustaceans and molluscs. Blue Crab was important in the diet of Snapper (~24%), especially in northern GSV, but other commercially targeted species were consumed at relatively lower levels (King George Whiting 2.9%, Western King Prawns 1.1%, Calarmari 0.4%, Garfish 0.04%). Results from the GSV ecosystem model estimated that Snapper accounted for ~11% of the total consumption of Blue Crabs, and ~1% of Western King Prawns, with most of the consumption of Blue Crabs and Prawns being accounted for by omnivorous crustaceans and a range of crustacean and piscivorous feeding fish, respectively. Sensitivity and scenario analysis of ecosystem responses to changes in Snapper biomass, suggests that most taxa groups are relatively insensitive to changes in Snapper biomass. Increasing Snapper biomass had a negative effect on Blue Crab biomass (<2% decline for each 10% increase in Snapper biomass), and a slight positive effect on Western King Prawn biomass (~1% increase for each 10% increase in Snapper biomass). Snapper may form an important mesopredator in the GSV ecosystem, providing a positive benefit to Prawns and Garfish through predation and competitive interactions with their key predators (Australian Salmon, Calarmari and Cuttlefish). Results from the ecosystem model suggest that the increase in Snapper biomass in GSV was unlikely to have contributed significantly to the observed reduction in biomass of Blue Crab and Prawn.

Scenarios examining the potential ecological and economic benefits (increased biomasses of commercially targeted taxa) that may result from a reduction in Snapper biomass achieved through a directed Snapper fishery, identified that such an approach was unlikely to deliver significant benefits across fisheries. In particular, the benefits to Blue Crab (slight positive effect) and Prawns (slight negative effect) was unlikely to deliver cost-effective benefits to their respective fisheries.

Implications for relevant stakeholders

The key implication of the development of the GSV ecosystem model, is that it provides a preliminary tool to optimise future economic and ecological performance in seafood production within an EBFM framework. This has not been possible until now. The GSV ecosystem model provides a framework for managers, industry and other stakeholders to investigate the potential ecological and economic implications of a range of fishery management scenarios. In particular, it provides a means to examine the potential impacts of management measures directed at one fishery on other fisheries and on the broader GSV ecosystem. The current model was developed explicitly to address the aims of the study, but with further development could be used to investigate a range of different scenarios.

Importantly, the model results were able to determine that the marked increase in Snapper biomass in the late 2000s in GSV was unlikely to have contributed significantly to the observed biomass reductions in Blue Crab and Western King Prawn.

An important outcome of the study was an ecosystem based assessment of the performance of the key GSV fisheries over a 20-year period. Results suggest that the declining performance of many of the key fisheries in GSV in recent decades, could be due in part to a decline and/or loss of production to the ecosystem over time. The important implications of this finding are the additional questions it raises about the source(s) and cause(s) for declining productivity, and further model developments could explicitly examine these questions.

Recommendations
To improve application, utility and confidence in the GSV ecosystem model as a decision support tool for ecosystem management, we recommend: a) further research to improve the provenance and quality of core data for key species and trophic interactions (diet and biomass); b) the development of a spatially explicit trophodynamic model (*Ecospace*) incorporating habitat data, the spatial distribution of fishing catch and effort, and spatial layers that capture the full range of activities in GSV; and c) inclusion of environmental time series to assess the potential impacts of environmental change (including climate change) on the GSV ecosystem, and the industries and activities it supports.

Keywords

Gulf St Vincent (GSV), Snapper, Western King Prawn, Blue Crab, trophic interactions, *Ecopath with Ecosim*.

1 Introduction

1.1 Background

A dramatic switch in the spatial structure of South Australia's Snapper fishery was observed between 2008 and 2013. Spencer Gulf (SG) has traditionally yielded the State's highest Snapper catches, however, in recent years it has been superseded by Gulf St. Vincent (GSV) and the South East (SE), two regions that had previously attracted little attention from fishers (Fowler *et al.* 2013). This shift has been a consequence of a downturn in the commercial harvest in northern SG coupled with increases in catches and catch rates in northern GSV. The perceived increase in Snapper biomass concerned commercial Blue Crab and GSV Prawn fishers as their catches had concurrently declined. As Snapper are known to heavily predate upon Blue Crabs and to a lesser extent prawns (Lloyd 2010), the apparent inverse trend in catches with the crustacean fisheries has generated suggestions that there may be predatory 'top-down' regulation operating within GSV on Blue Crabs and Prawns. The potential interaction between these three high-value fisheries and their respective management concerns has consequently generated two broad questions: 1) what are the flow-on ecosystem effects of changes in Snapper biomass on Western King Prawns, Blue Crabs and other species; and 2) what balanced exploitation scenarios would optimise production and value of Snapper, Western King Prawns and Blue Crabs and minimise ecosystem impacts?

Ecosystem based fishery management (EBFM) seeks to sustain healthy marine ecosystems and the fisheries they support, by reversing the order of management priorities from the target species to the ecosystem. The overall objective of EBFM is to sustain healthy marine ecosystems and the fisheries or ecosystem services they support. Specifically, it seeks to address some of the unintended consequences of fishing, including habitat destruction, incidental mortality of non-target species and changes in the structure and function of ecosystems. Such considerations underpin more precautionary fishery management measures that favour the ecosystem where knowledge to aid management is insufficient (Pikitch *et al.* 2004), and more selective fishing practices in one or more of the '6-S' selection strategies, namely: species, stock, size, sex, season and space (Zhou *et al.* 2010). EBFM has been suggested to provide a more effective and holistic approach to managing fisheries than traditional approaches (Gislason *et al.* 2000, Pikitch *et al.* 2004). Despite the global recognition of the need to adopt the EBFM approach, there are very few examples where it has been utilised effectively to complement the existing single species/stock management paradigm. It has been suggested this is largely due to the science of its implementation often being considered overly complex and difficult because ecosystem approaches are so broadly inclusive (Cowan Jr *et al.* 2012).

In GSV, the Western King Prawn fishery has made major advances in increasing the selectivity of their fishing methods, with the introduction of rigid-grid bycatch reduction devices (BRDs) and T90-mesh cod-ends in 2012. Use of BRDs has substantially reduced sponge, elasmobranch and fish bycatch, not only improving fishing efficiency, but enhancing the ability of the T90 net to exclude juvenile prawns, small fish and crustaceans from the catch (Dixon *et al.* 2013a). The outcomes for industry are improved catch quality (fewer damaged prawns) and trawling efficiency due to longer trawl shot duration and increased fuel efficiency (total fuel consumption per hour of trawling). The expectation is that reduced bycatch will also lead to good environmental outcomes; however, the ecological changes to GSV and the flow-on effects of reduced bycatch and discards to benthic communities and to Snapper, Prawn and Blue Crab production are uncertain. Recently, the concept of 'balanced exploitation' for EBFM is gaining acceptance (Zhou *et al.* 2010). Where multiple fisheries operate in the same ecosystems, this approach favours exploitation balance among species, stocks, sexes and sizes, as a means to enhance maintenance of sustainable ecosystems and biodiversity (Zhou *et al.* 2010). A key concept to balanced exploitation is maintenance of the abundance of species, stocks, sexes and sizes above certain thresholds, in conjunction with productivity dependent exploitation, and minimising ecological impacts.

This project aims to develop a GSV ecosystem model to assess the flow-on ecosystem effects of increased Snapper abundance on other commercial fisheries within the gulf. Exploring shifts in biomass between key, high value species across a range of trophic levels, while assessing the ecological and stock production implications of greater selectivity in fishing methods is the principle focus of this model. It will also provide a potential decision support tool for multi-species fisheries management in GSV, as a step toward fully integrated (EBFM) whole-of-system management.

1.2 Need

The apparent biomass of Snapper in northern GSV increased considerably above historic levels, between 2008 and 2013. This observation coincided with a marked decline in catches by GSV Prawn and Blue Crab fisheries to the extent that it required a considerable reduction in fishing effort to promote stock recovery. As Snapper are known to predate upon Blue Crabs and Prawns, there has been concern and uncertainty among fishers and managers that the downturn in these crustacean fisheries has occurred in part, due to predatory 'top-down' regulation by Snapper within GSV. The potential interaction between these three high value fisheries and their respective management has consequently generated a need to better understand the trophic interactions that underpin production of these fisheries in GSV, how exploitation rates within each fishery may impact on production in the others, and what balanced exploitation scenarios would optimise production and value across fisheries, while minimising ecosystem impacts.

A GSV ecosystem model is needed to assess the linkages between increased Snapper abundance and reduced production in crustacean fisheries. Sensitivity analyses and scenario testing will identify the ecological factors important to production across the three fisheries, assessing the ecological and production implications of greater selectivity of fishing methods in the GSV Prawn fishery and multi-species optimisation scenarios. It will provide the basis for a potential decision support tool for multi-species fisheries management in GSV.

1.3 Objectives

The objectives of the project were to develop a GSV ecosystem model to:

(1) Assess temporal change, the effects of fishing and improved fishing selectivity on the GSV ecosystem over the last 20 years,

(2) Understand the impact of changes in the abundance of Snapper on the GSV ecosystem, with particular emphasis on other high value commercial fisheries, i.e. Prawns and Blue Crabs, and

(3) Assess and optimise future ecological and economic performance of multi-species fisheries in an EBFM framework.

2 Methodology

2.1 Ecopath and mass balance approach

The *Ecopath with Ecosim* (EwE) software (www.Ecopath.org, Version 6.5) was used to develop a trophic mass-balance model of the Gulf St Vincent (GSV) ecosystem. *Ecopath* was developed by Polovina (1984), based on a simple steady-state trophic box model, and further developed by Christensen and Pauly (1992a) 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 EwE 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, Griffiths *et al.* 2010, Piroddi *et al.* 2010, Shannon *et al.* 2008). For the GSV ecosystem, an *Ecopath* model was constructed for 1994. Time series data over a 20-year period (1994-2013) were used to develop the *Ecosim* model. A 20-year time period was chosen as it provided a suitable period to tune the model to fisheries time-series data (see Section 2.3 below) and assess ecosystem change.

2.2 Model area and structure

The model domain extended from Cape Davenport (lower Yorke Peninsula) to Cape Cassini (north-coast of Kangaroo Island) across Investigator Strait, to a line between the southernmost point of the Fleurieu Peninsula and Cape St Albans on Kangaroo Island (Figure 2.1). The model area was calculated to be 10,500 km^2.

A number of functional or trophic groups were used in the GSV ecosystem model, based on species similarity in terms of diet, habitat, foraging behaviour, size, consumption and rates of production (Table 2.1). Many commercial species were modelled as separate groups to aid scenario testing/modelling, and facilitate the assessment of impacts and drivers. The GSV ecosystem model structure is built around 75 functional groups including: mammals (4), birds (6), chondrichthyans (9), teleosts (26), cephalopods (4), other invertebrates (18), microbes (2), autotrophs (3), detritus (2) and discards (1) (Table 2.1, Appendix 2). A large dietary matrix was developed that included 328 prey taxa categories. Dietary information was based largely on those data utilised in the construction of the Spencer Gulf Ecosystem (SGE) Model (FRDC Project No. 2011/205; Gillanders *et al*. 2015), but where regional data with provenance to GSV were available, they were used in preference. Key sources of dietary data were Page et al. (2011) and Currie and Sorokin (2010). For Snapper, diet data were sourced from a large dataset of stomach contents analysis obtained across multiple seasons within GSV (Lloyd 2010, and SARDI unpublished data). 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 multiple species. The aggregation of species into trophic groups will therefore effect model dynamics in some instances; however, by matching species for diet, consumption, and production rates we attempted to constrain the errors and uncertainty of aggregating (see Appendix 3).

In addition to diet information, there are four key parameters that are required by *Ecopath* for each group to balance a model. These include biomass (B), production per unit of biomass (P/B, equivalent to the instantaneous rate of total mortality (Z) used by fisheries biologists, under the steady-state assumption of the model), 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 are provided in Appendix 2.

Fishery data on landings, discards and effort were obtained for the GSV ecosystem region and broken down into 31 fishing fleets (Table 2.2). With the exception of the Cockle, Charter boat, Recreational and 'Other' fishing fleets, the remaining fisheries all fall within six main fishery management units: South Australian (SA) Sardine (1 fleet), SA Marine Scalefish (18), GSV Prawn (1), Blue Crab (4), Abalone (2) and Southern Rock Lobster (1). Annual fishery landings and effort data were obtained for all fleets between 1994 and 2013 (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 1994 based on their proportion to landed catch or effort (Currie *et al.* 2009b, 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}).

For the GSV Prawn fishery, the discarded catch data were obtained from a project that trialled T90 cod-end nets and bycatch reduction devices (BRDs) (Dixon *et al.* 2013a). Although the raw bycatch data were not published, they were available and analysed as part of this project so they could be incorporated into the GSV ecosystem model. The standardised surveys of catch and discards in the GSV Prawn fishery followed the methods described by Dixon *et al.* (2013a), and biomass estimation of taxonomic groups followed methods described by Currie *et al.* (2009a) and Burnell *et al.* (2015). The upper standard error estimates were used to provide a starting biomass (prior to model balancing) for many taxonomic groups in the GSV model, because trawl surveys generally underestimate biomass, depending the size, mobility and net-avoidance capacity of taxa (Poiner *et al.* 1998). The biomass estimates derived for each bycatch species are presented in Appendix 4.

For the GSV Blue Crab fishery, the discarded catch data were obtained from fishery-independent surveys conducted annually since 2002. Bycatch data from 2002 to 2006 were published in the stock assessment report (Currie *et al.* 2007), but for the subsequent years is unpublished. However, the unpublished data were available and biomass estimates for these were calculated for 2002 to 2014.

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.

2.3 Model fitting

Dynamic simulations were run in *Ecosim* using 94 time-series (1994-2013) of estimates of fishing effort, biomass or relative biomass (CPUE) and fishing mortality (F) for functional groups with available data (for list of time series see Appendix 5). Several *Ecosim* scenarios were explored through adjustment of predator vulnerability using the 'fit to time series' procedure. Different numbers of predator interactions within the dietary matrix were selected (10-50) within this procedure to identify the most sensitive and optimal number of predator interactions, and their vulnerability values that would minimise the model sum of squares (SS) and Akaike Information Criteria (AIC), 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.

2.4 Ecosystem indicators

After the model fitting procedure in *Ecosim*, we examined three variables to evaluate changes in the marine ecosystem: 1) total catch; 2) 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 3) 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).

2.5 Scenario testing

Scenario testing was undertaken to explore different 'what-if' questions that can be informed by trophodynamic modelling of the GSV ecosystem. Two key scenarios were examined: the first to assess the potential ecosystem implications of greater gear selectivity in the GSV Prawn fishery brought about by the introduction of BRDs and T90 nets in 2012; the second to assess the potential impacts on the ecosystem and high value fisheries from different Snapper biomass scenarios.

For the fishing gear selectivity scenario, results from the study by Dixon *et al.* (2013a) were used as the basis to estimate the reduced extent of bycatch and discards associated with the introduction of BRDs and T90 nets into the fishery. There was no significant difference in mean catch rate between the T90 cod-end net with or without the BRD, therefore the data for landings remained unchanged. However, there were reductions in the bycatch when the T90 cod-end nets with BRDs was used (relative to no-BRD/T90). The bycatch was broadly separated into three groups: sponges, elasmobranchs (sharks and rays) and fish/crabs. The reduction in bycatch rate indicated in Figure 12 of Dixon *et al.* (2013a) was used to estimate the proportional decrease in biomass, i.e. 0.125 for sponges, 0.0227 for elasmobranchs and 0.8182 for fish/crabs. These proportions were then used to adjust the values across taxa groups in the pre-BRD/T90 discard catch data. A new 'GSV Prawn BRD/T90' fleet was added with these modified (reduced) discards. In the base (pre-BRD/T90) model scenario, fishing only occurred in the regular GSV Prawn fleet, effort in the new (BRD/T90) fleet was set to zero, and a constant level of fishing effort was maintained over a 50-year period (2014 to 2063), based on the average 2002 to 2012 effort levels (0.85 of 1994 levels). In the post-BRD/T90 scenario, fishing effort was increased in the new fleet to match that used for the old (pre-BRD/T90) fleet in the base model and run for a 50-year period (2014 to 2063) with the old fleet effort set to zero. In both scenarios, fishing effort in all other fleets was held constant at 2013 levels. The relative change in estimated biomass of groups in the post-BRD/T90 scenarios was then compared to that of the base model.

For scenarios assessing the potential impacts of increased Snapper biomass, Snapper biomass was increased by 10%, 20% and 30% of 2013 levels of the base GSV ecosystem model and then held constant for a 50-year period (2014 to 2063), while maintaining all fishing fleet efforts constant at 2013 levels (except for the GSV Prawn fleet, which was maintained at 0.85 of 1994 levels, as there was zero effort in 2013). The relative change in estimated biomass of groups under the different Snapper biomass scenarios was then compared to that of the base model. Additional scenarios were run where Snapper biomass was fixed at varying levels ranging from 0.1 and 0.7 t km^{-2}, and held constant for a 50-year period to examine the potential impacts of a range of difference Snapper biomasses on the GSV ecosystem from a base level of 1994 to an equivalent of a 7-fold increase in biomass. The group biomasses under different Snapper biomass scenarios were then compared.

Figure 2.1 Area of Gulf St. Vincent (GSV) (shaded blue) used to define the model domain in the GSV Ecosystem Model. The region of the Spencer Gulf Ecosystem Model (FRDC 2011/205) is also indicated, bounded by the black line.

Table 2.1 Functional or trophic groups used in the GSV ecosystem model as defined in Appendix 2. 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 detritivore feeding; DOM = dissolved organic matter; POM = particulate organic matter; ZF = zooplankton feeding.

	Group name	Trophic level	Habitat area (fraction)	Biomass (t/km²)	P/B (/year)	Q/B (/year)	EE
1	Australian Sea Lion	4.93	1	0.00627	0.79200	29.44000	**0.00006**
2	Long Nosed Fur Seal	5.00	1	0.00213	1.18400	49.86000	**0.04310**
3	Bottlenose Dolphin	4.85	1	0.00354	0.08000	18.99000	**0.04543**
4	Common Dolphin	5.09	1	0.02210	0.09000	20.58000	**0.00525**
5	Petrel	4.62	1	0.00514	1.00000	191.18000	**0.57278**
6	Australian Gannet	5.40	1	0.00003	1.00000	125.33000	**0.00000**
7	Little Penguin	4.87	1	0.00148	1.29000	85.64000	**0.47190**
8	Shags & Cormorants	4.46	1	0.00051	1.00000	77.41000	**0.00000**
9	Terns	4.84	1	0.00003	1.00000	90.23000	**0.87527**
10	Gulls	3.96	1	0.00203	1.00000	129.35000	**0.00000**
11	White Shark	5.87	1	0.00001	0.10000	1.73000	**0.00000**
12	Whaler Shark	5.15	1	**0.03000**	**0.09500**	2.61000	0.95000
13	Smooth Hammerhead	5.63	1	**0.00003**	**0.10000**	3.15000	0.95000
14	Common Thresher Shark	5.00	1	**0.00001**	**0.15000**	2.78000	0.95000
15	Gummy shark	3.69	1	0.07472	0.55000	2.60000	**0.28899**
16	School shark	5.15	1	**0.00205**	**0.88000**	2.50000	0.90000
17	Port Jackson shark	4.22	1	0.02529	0.25000	1.52000	0.71081
18	Other demersal shark	3.66	1	0.03622	0.35100	2.60000	0.53671
19	Rays & skates	3.65	1	0.16777	0.41800	1.76000	0.32163
20	Southern Bluefin Tuna	5.16	1	**0.00258**	**0.20000**	1.60000	0.90000
21	Yellowtail Kingfish	5.22	1	**0.00287**	**0.20000**	2.50000	0.90000
22	Snapper	3.85	1	**0.13065**	**0.49300**	3.80000	0.90000
23	Snook	4.80	1	**0.04247**	**0.41100**	3.51000	0.90000
24	Barracouta	5.22	1	**0.22935**	**0.41100**	3.64000	0.90000
25	Skipjack Trevally	3.70	1	0.21971	0.48000	4.17000	**0.79584**
26	Medium piscivore fish	4.48	1	**0.37753**	**0.63600**	1.58000	0.90000
27	Medium echinoderm fish	3.33	1	**0.01362**	**0.62500**	2.34000	0.90000
28	Australian Salmon	4.95	1	**0.59737**	**0.45000**	4.70000	0.90000
29	Australian Herring	3.83	1	**1.04027**	**0.45000**	4.70000	0.90000
30	King George Whiting	3.58	1	**0.07879**	**0.54800**	2.29000	0.90000
31	Garfish	2.94	1	**0.14984**	**0.32900**	4.73000	0.90000
32	Red Mullet	3.66	1	**0.07984**	**0.79000**	2.36000	0.90000
33	Silverbelly	3.62	1	**0.22685**	**1.10000**	4.40000	0.90000
34	Medium crustacean fish	3.72	1	**0.09585**	**0.54600**	2.97000	0.90000
35	Medium molluscan fish	3.38	1	**0.35628**	**0.86900**	2.26000	0.90000
36	Small crustacean fish	3.46	1	**0.48935**	**1.31500**	3.32000	0.90000
37	Degens/Rough Leatherjacket	3.10	1	**0.85509**	**0.90000**	2.26000	0.90000
38	Small polychaete fish	3.22	1	**0.32935**	**0.99200**	2.82000	0.90000
39	Syngnathids	3.63	1	**0.00393**	**1.00000**	4.70000	0.90000

40	Blue Mackerel	4.14	1	**1.23074**	**0.49000**	6.40000	0.90000
41	Jack/Yellowtail Mackerel	4.24	1	**3.30417**	**0.52000**	5.37000	0.90000
42	Sardine	4.18	1	1.79589	1.00000	5.04000	**0.45379**
43	Anchovy	3.97	1	**2.19924**	**0.98000**	5.76000	0.90000
44	Sprats	3.30	1	**0.58648**	**1.00000**	5.76000	0.90000
45	Fish larvae	2.82	1	**1.77291**	**4.00000**	20.00000	0.99000
46	Southern Calarmari	5.05	1	**0.08870**	**1.83000**	18.25000	0.90000
47	Giant Australian cuttlefish	3.72	1	**0.06592**	**2.37000**	5.80000	0.90000
48	Other squids	4.51	1	**0.17060**	**1.80000**	17.50000	0.90000
49	Octopus	3.74	1	**0.07866**	**2.37000**	7.90000	0.90000
50	Rock Lobster	2.87	0.5	0.02941	0.73000	12.41000	**0.47857**
51	Western King Prawn	2.38	1	0.08225	7.57000	37.90000	**0.61467**
52	Blue Swimmer Crab	2.99	1	**0.43756**	**2.80000**	8.50000	0.90000
53	Sand Crab	3.06	1	**1.01104**	**2.80000**	8.50000	0.90000
54	Other large crabs/bugs	2.01	1	**23.99724**	**2.80000**	8.50000	0.90000
55	Sand associated omnivore crustacean	2.50	1	**28.98130**	**0.79000**	11.30000	0.90000
56	Herbivorous macrobenthos	2.32	1	**35.80787**	**2.80000**	14.00000	0.90000
57	Sand zoobenthos feeder	2.27	1	**118.67940**	**0.65000**	7.50000	0.90000
58	Greenlip Abalone	2.00	0.2	0.12000	1.50000	15.00000	**0.00152**
59	Blacklip Abalone	2.00	0.2	**0.00339**	**1.50000**	15.00000	0.90000
60	Small mobile DDF crustacean	2.51	1	**1.15765**	**7.01000**	27.14000	0.90000
61	Small mobile ZF crustacean	3.48	1	**39.18450**	**1.12000**	9.50000	0.90000
62	Polychaetes DDF	2.62	1	**6.56530**	**1.60000**	6.00000	0.90000
63	Sessile epifauna	2.47	1	0.95748	2.80000	11.80000	0.28940
64	Gelatinous zooplankton	3.38	1	0.20000	16.50000	80.00000	0.16542
65	Large carnivorous zooplankton	2.95	1	**45.87358**	**5.00000**	32.00000	0.99000
66	Small herbivorous zooplankton	2.03	1	**42.59775**	**29.50000**	55.00000	0.99000
67	Meiofauna	2.56	1	**0.88714**	**35.00000**	125.00000	0.99000
68	Microphytobenthos	1.65	1	0.50000	9500.0000	12000.000	**0.26935**
69	Planktonic microflora	1.62	1	**2.20880**	**571.00000**	1028.0000	0.99000
70	Macroalgae	1.00	0.012	46.80000	10.00000	0.00000	**0.82785**
71	Seagrass	1.00	0.202	667.87260	0.93800	0.00000	**0.04255**
72	Phytoplankton	1.00	1	22.00000	190.00000	0.00000	**0.50691**
73	Detritus DOM water column	1.00	1	20.40000			**0.95179**
74	Detritus POM sediment	1.00	1	18.50000			**0.76073**
75	Discards	1.00	1	0.10490			

Table 2.2. Details of the 31 different fishing fleets examined in the GSV Ecosystem Model. PS = purse seine; HN = haul net; DN = dab net; LL = long line; HL = hand line; CP = crab pot; CN = crab net; LP = lobster pot; CR = cockle rake. Other refers to minor fisheries (ocean jackets, octopus, poles/rods, troll line, and trot line).

No.	Fleet name	Fishery Management Unit
1	Snapper-HL	Marine Scalefish Fishery
2	Snapper-LL	Marine Scalefish Fishery
3	King George Whiting-HN	Marine Scalefish Fishery
4	King George Whiting-HL	Marine Scalefish Fishery
5	Australian herring-HN	Marine Scalefish Fishery
6	Australian salmon-HN	Marine Scalefish Fishery
7	Australian salmon-PS	Marine Scalefish Fishery
8	Garfish-HN	Marine Scalefish Fishery
9	Garfish-DN	Marine Scalefish Fishery
10	Sardine-PS	SA Sardine Fishery
11	Shark-HL	Marine Scalefish Fishery
12	Small mesh net	Marine Scalefish Fishery
13	Large mesh set net	Marine Scalefish Fishery
14	Troll line	Marine Scalefish Fishery
15	Drop line	Marine Scalefish Fishery
16	Fish trap	Marine Scalefish Fishery
17	Other-HN	Marine Scalefish Fishery
18	Other-HL	Marine Scalefish Fishery
19	Squid jig	Marine Scalefish Fishery
20	GSV Prawn	GSV Prawn Fishery
21	Blue crab-CP	Blue Crab Fishery
22	Blue crab-HN	Blue Crab Fishery
23	Blue crab-CN	Blue Crab Fishery
24	Sand crab-CN	Blue Crab Fishery
25	Rock lobster-LP	Southern Rock Lobster Fishery
26	Blacklip abalone	Abalone Fishery
27	Greenlip abalone	Abalone Fishery
28	Cockle-CR	
29	Charter Boat	
30	Recreational Fishing	
31	Other	

3 Results

3.1 Trophic structure and flow (*Ecopath*)

The basic parameters used to inform the 75 functional groups within the *Ecopath* model are presented in Table 2.1, 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.

The trophic flows between the functional groups in the GSV ecosystem model estimated by *Ecopath* are summarised in Figure 3.1. The trophic level of the functional groups ranged from 1 to 5.87, with the highest values for White Shark, Smooth Hammerhead, Australian Gannet, Yellowtail Kingfish, Barracouta, Southern Bluefin Tuna, Whaler Sharks, School Shark, Common Dolphin, Southern Calarmari, Long-Nosed Fur Seals and Common Thresher Shark (TL >5). Australian Salmon, Australian Sea Lion, Little Penguin, Bottlenose Dolphin, Terns, Snook, Petrels and other squids had TL ≥ 4.5. (Table 2.1, Fig. 3.1). Medium piscivorous fish, Shags and Cormorants, Jack/Yellowtail Mackerel, Port Jackson Shark, Gulls, Sardine, Blue Mackerel, Anchovy, Gulls, Snapper, Australian Herring, Octopus, medium crustacean eating fish, Giant Australian Cuttlefish, Skipjack Trevally, Gummy Shark, other demersal sharks, Red Mullet, rays and skates, syngnatids, silverbelly, and King George Whiting had trophic levels ranging between 3.5 and 4.5. In terms of biomass, the lower trophic levels of the GSV ecosystem are dominated by crustacean groups, seagrass and macroalgae (Fig. 3.1).

3.2 Model fitting (*Ecosim*)

A total of 94 time-series were loaded into the *Ecosim* model (Appendix 5). The modelled time-series of biomass and estimated catch and observed trends, for the six key GSV fished species are presented in Figure 3.2. Modelled estimates of biomass tracked those indicated by CPUE data reasonably well for most groups (Fig. 3.2). However, for Snapper the model struggled to replicate the full extent of the increase after 2008. For Western King Prawn, the model does not capture the peaks in prawn catches, but does track the general decline. For Garfish, the model failed to track the ongoing decline in CPUE, instead predicting a slight increase in biomass over the modelled period (Fig. 3.2). Estimated modelled catches of the six key fished taxa tracked actual catches reasonably well (Fig. 3.2). However, for Snapper the model anticipated a gradual increase in catch throughout the late 1990s and early mid-2000s, when in fact catch was fairly flat until the sudden increase in the late 2000s (Figure 3.2).

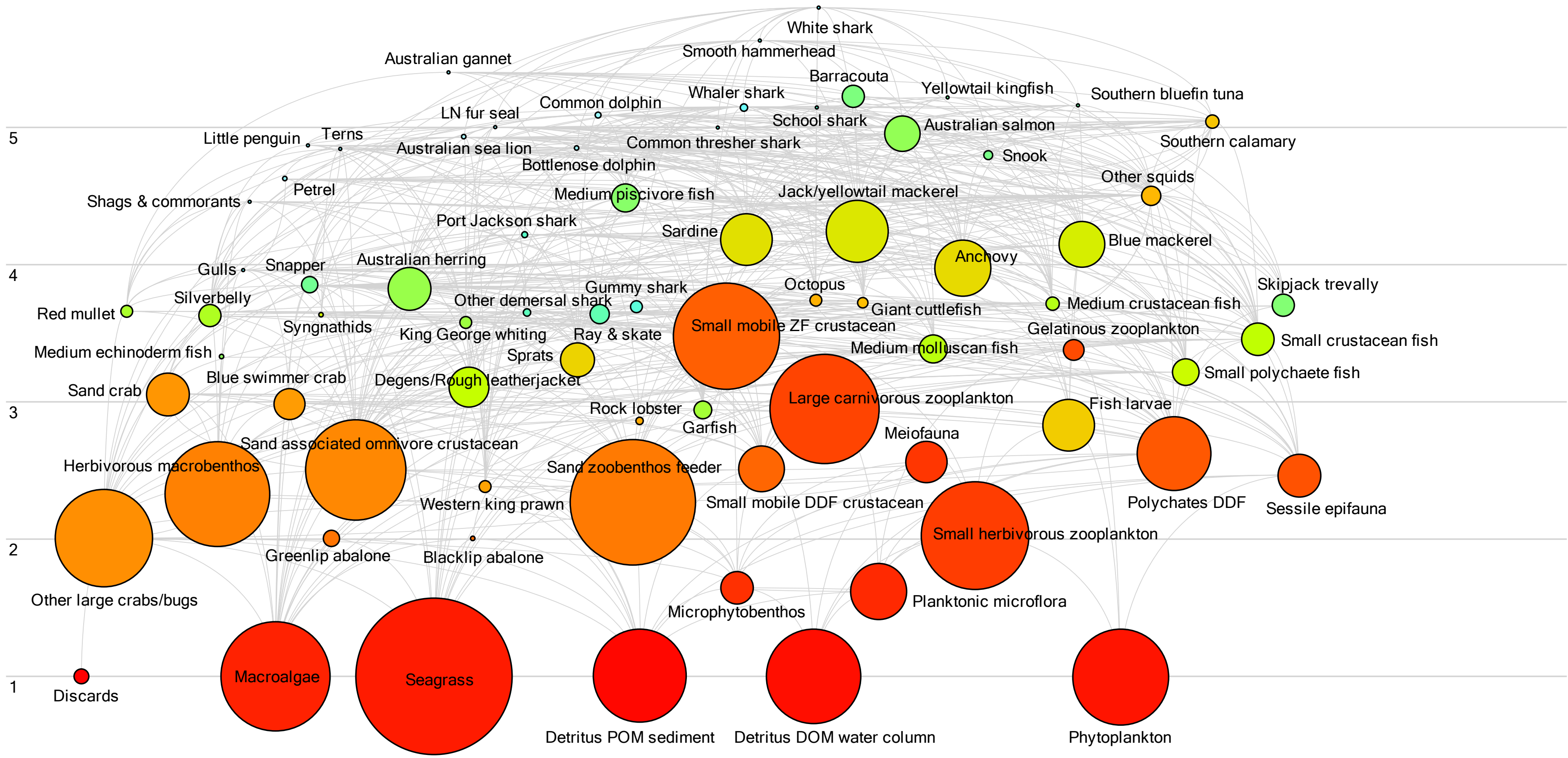


Figure 3.1 Flow diagram expression of trophic flows and trophic levels in the GSV ecosystem. Functional groups are represented by a circle; the size of the circle is proportional to its biomass (colour of circles is unrelated to any parameter).

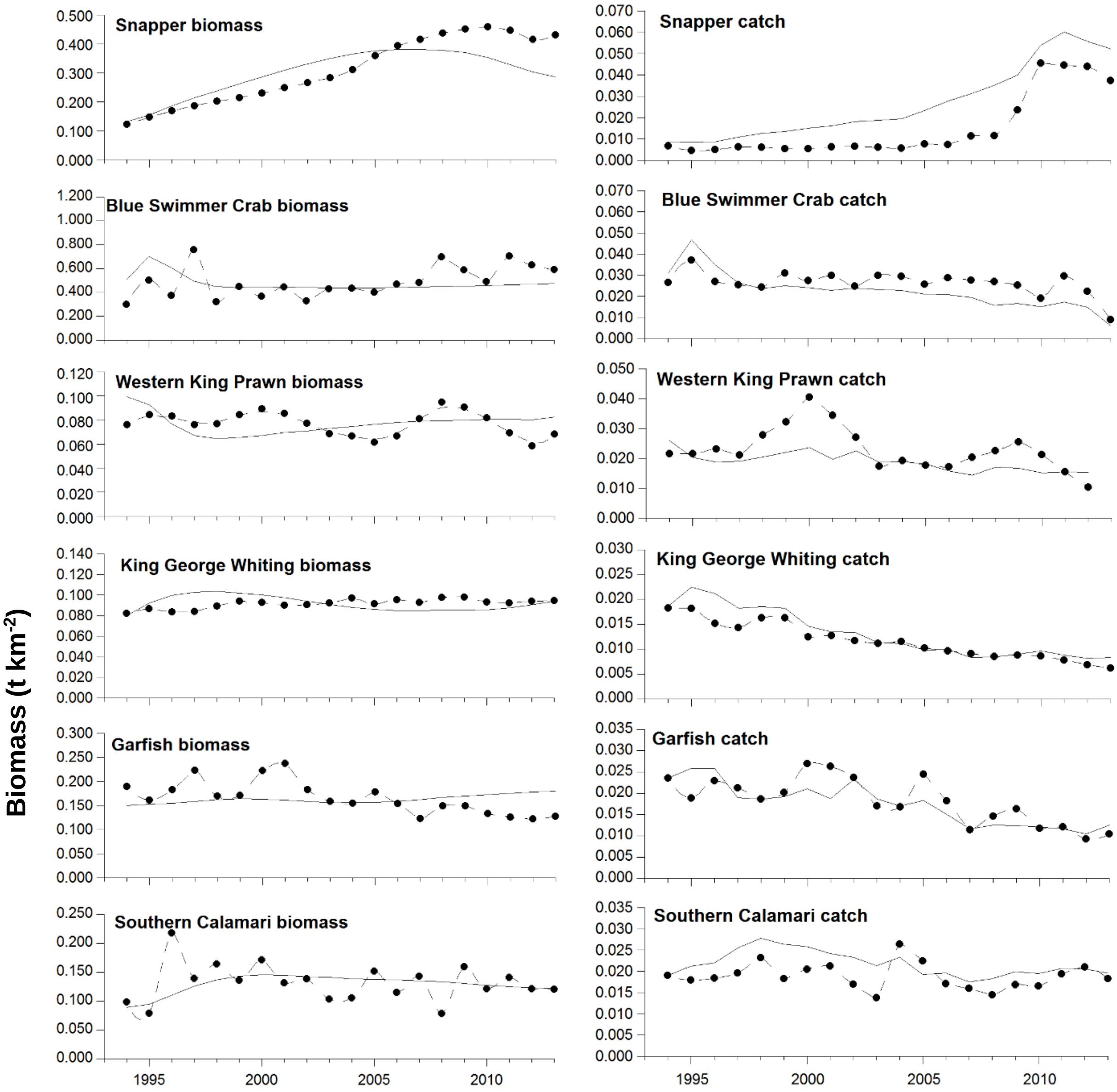


Figure 3.2 Example of time series fits of the Gulf St Vincent ecosystem model (thin line) to observed biomass (CPUE) and catch (dots and dashed trend line) data for six key commercial groups between 1994 and 2013.

3.3 Temporal changes in GSV ecosystem

Trends in catch

Trends in total catches (landings and discards), catches of the six high value commercial fisheries (prawn, Blue Crab, Snapper, Garfish, King George Whiting and Calamari), and of other groups in the GSV ecosystem between 1994 and 2013 are presented in Figure 3.3. Trends in total catch show marked peaks as a consequence of sardine catch in GSV. The sardine fishery developed in southern Spencer Gulf, and there was significant growth in the fishery between 1991 and 2010, with the major growth period occurring since 2000. Although most of the catch occurs in southern Spencer Gulf, occasionally catch is taken in the GSV region (mainly in Investigator Strait and lower GSV) (Ward *et al.* 2015). When sardine catches are excluded, the remaining aggregated GSV catch has declined steadily over the 20-year period by about one-third (Figure 3.3a), with mean catch over the last five years (2,057 t; 2009-2013) being about 68% of that caught in the first five years of the study period (3,016 t; 1994-1998). Also, the total catch in the final year (1,389 t, 2013) was less than half that in the first year of the study (3,026 t; 1994).

A major component in the decline in catch is attributable to a major reduction in effort in the Australia salmon purse seine fishery, which resulted in a substantial (~90%) reduction in the catch of this species over the study period (Figure 3.3 c). There was also a major reduction in large mesh net set (demersal gillnet) effort targeting gummy and school shark (Figure 3.3 c-d), which followed transfer of management of the fishery from the State to the Commonwealth in 2000, resulting in major closures in State waters including GSV. This resulted in a significant (~60%) decline in the combined catch of gummy, school and other demersal sharks over the 20-year period (Figures 3.3 c-d).

There were significant changes in the catches of the six main high-value commercially targeted groups between 1994 and 2013 (Figure 3.3 b). Prawn and Blue Crab catches declined by ~37% (~21% if 2013 is excluded when the fishery was closed) and ~25% between the first and last five years of the study, respectively. Significant declines in King George Whiting (~54%) and Garfish (~43%) also occurred (Figure 3.3 b). In contrast, catches of Calamari were relatively stable. Snapper catches remained relatively stable between 1994 and 2004, before increasing sharply between 2005 and 2010 (Figure 3.3b). There was a 5.75-fold increase in Snapper catch between the first five and last five years of the study period.

Group biomasses

The relative change in group biomasses between 1994 and 2013, estimated by the GSV ecosystem *Ecosim* model, are presented in Figure 3.4. Across the array of different trophic groups, most showed an increase in relative biomass over the study period (Figure 3.4). A major part of this is directly related to the marked reduction in catch (~46%) of both landed (-54%) and discarded (-33%) groups and the flow-on effects through trophic cascades to higher trophic levels (Figure 3.4). This is apparent with yearly changes in biomass over the study period for teleost fish, sharks and rays, and cephalopods (Figure 3.5). Teleost groups estimated to increase significantly over the study period included Snapper, Snook, Australian Salmon and Herring, medium molluscan feeding fish, syngnathids and anchovy (Figure 3.4). For chondrichthyans, notable recoveries included School Shark, Common Thresher, Smooth Hammerhead, and Whaler Sharks (Figure 3.4). The estimated increase in the cephalopod groups was largely driven by a notable recovery in Giant Australian Cuttlefish (Figure 3.4). For most commercially-targeted groups, the model estimated an increase in biomass over the study period, although as noted above, the projected increase in Garfish is at odds with declines in CPUE, harvest fraction and fishable biomass since 2010 (Figure 3.2, 3.3) (Steer *et al.* 2016). Declines in biomass were estimated to have occurred for Blacklip Abalone, Western King Prawn and Blue Swimmer Crab (Figure 3.4). The model estimated increases in the biomass of marine mammal and seabird groups throughout the study period (Figure 3.4, 3.5). Although the model includes fishing mortality for many of the targeted (landed) and discarded groups, it does not include any non-fishery related anthropogenic mortality. This may be significant for some of the higher trophic level groups. For example, the model projected an increase in the biomass of Australian sea lion over the 20 year study period, whereas recent surveys for the species indicate that the population has declined significantly over that time (Goldsworthy *et al.* 2015).

Effects of fishing

Ecosystem indicators

In addition to trends in catches and changes to group biomasses, two additional ecosystem indicators (the mean trophic level of the catch and FIB index, were used to evaluate ecosystem change and the effects of fishing. The mean trophic level of the catch remained relatively stable throughout the study period and ranged from 3.3 to 3.8 (Figure 3.6a). The lowest point of 3.3 occurred in 2007, and then increased steadily to 2013 to 3.8 (Figure 3.6a). Declines in the catch of Australian Salmon (TL=4.95) contributed to the slight decline in mean trophic level of the catch to 2007, whereas increases in the catch of Snapper (TL=3.85) and declines in catches of Blue Crab (TL=2.99), Prawn (TL=2.38), Garfish (TL=2.94) and steady catches of southern Calarmari (TL=5.05) contributed to the increase in mean trophic level of the catch between 2007 and 2013 (Figure 3.5a).

The FIB index was stable across the 20 years in absolute terms (Figure 3.6b). In general, the FIB index increases if the underlying fishery expands beyond its traditional fishing area or ecosystem, or if bottom-up effects result in more catch than expected; and decreases if the geographic area contracts, or if the underlying food web is collapsing (Coll *et al.* 2009, Pauly and Palomares 2005). The magnitude of change in FIB index typically reported for fisheries-impacted ecosystems are generally in the order of 0.1 to 1.0 or more over time (Large Marine Ecosystems of the world at www.seaaroundus.org, Kleisner and Pauly 2011, Tsikliras *et al.* 2015). In contrast, the magnitude of the range of FIB index values for the 20-year study period in GSV was two to three orders of magnitude lower (i.e. stable).

Improved gear selectivity in GSV prawn fishery

The ecosystem impacts of improved gear selectivity following the introduction of devices BRD and T90 nets in the GSV Prawn fishery were examined by comparing the estimated biomasses of groups following pre- and post-BRD/T90 fishing scenarios, run over a 50-year period (2014 to 2063) (Figure 3.7). The most notable impact following post-BRD/T90 fishing scenarios was the marked reduction in discards (~43%) relative to pre- BRD/T90 fishing scenarios (Figure 3.7). There was also a marked increase in biomass of demersal chondrichthyans, particularly Port Jackson sharks (~22%), rays and skates (~10%) and other demersal sharks (~6%) following post-BRD/T90 fishing scenarios, relative to pre-BRD/T90 fishing scenarios (Figure 3.7). For all other groups, biomass changes estimated under a post-BRD/T90 fishing scenario were relatively minor (<1%), and there were no projected negative impacts on commercially-targeted species (Figure 3.7).

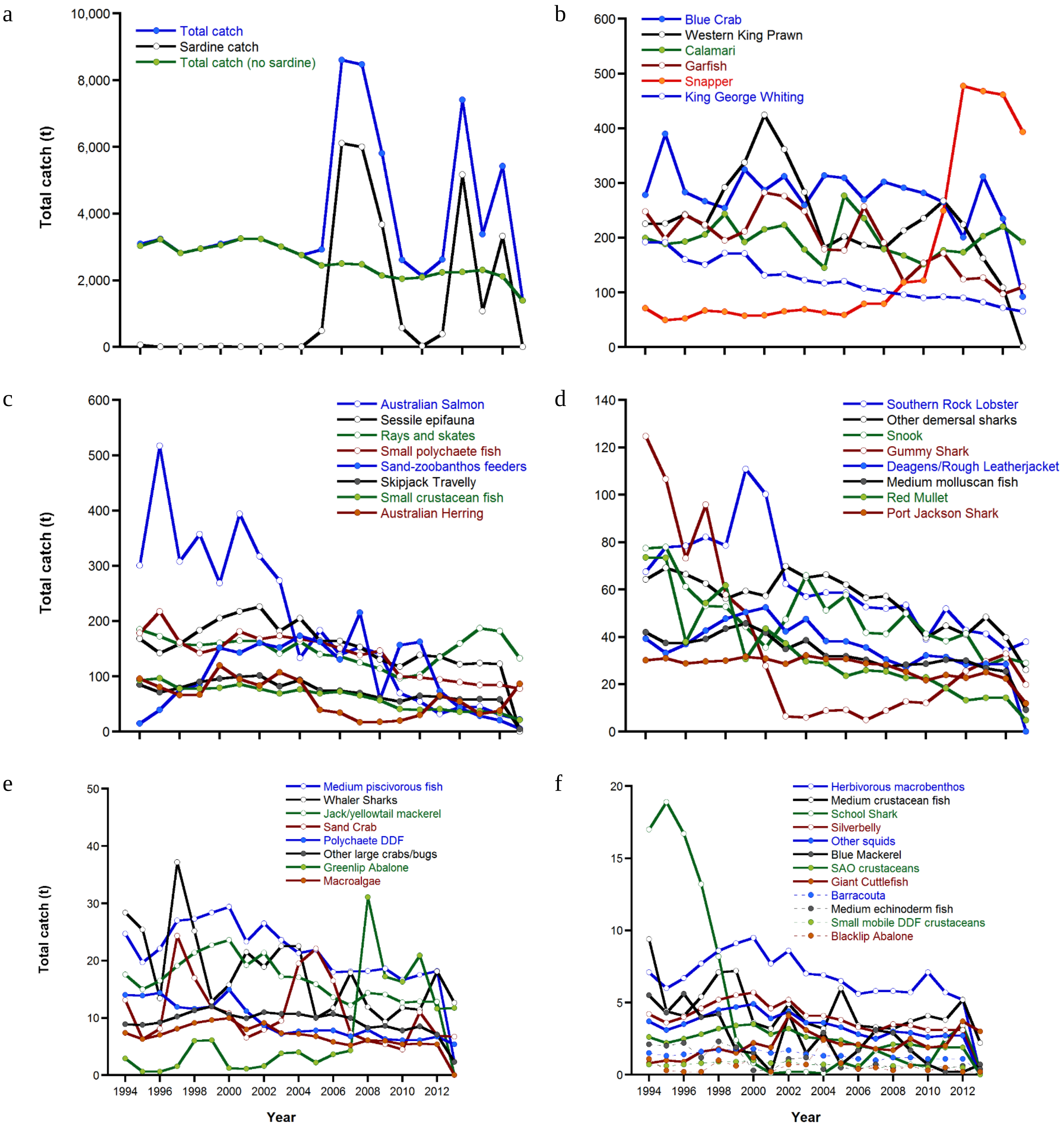


Figure 3.3 Trends in the total catch (t y^{-1}) (observed landings plus estimated discards) of groups in the GSV ecosystem between 1994 and 2013.

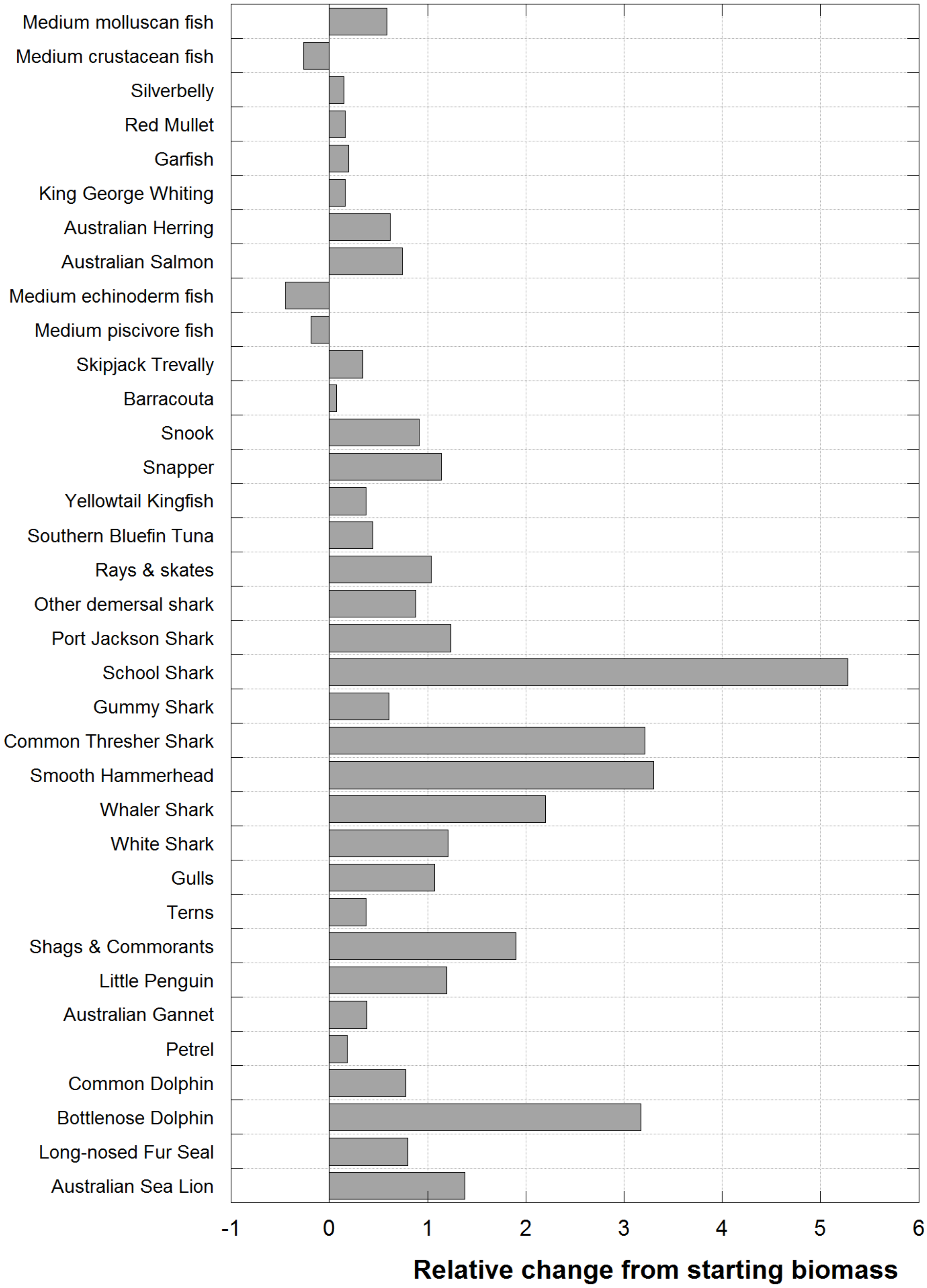


Figure 3.4. Estimated relative change in the biomass of functional groups in the GSV ecosystem model from 1994 to 2013. A value of 1 indicates a doubling of biomass, -1 a loss of that group.

Figure. 3.4 continued.

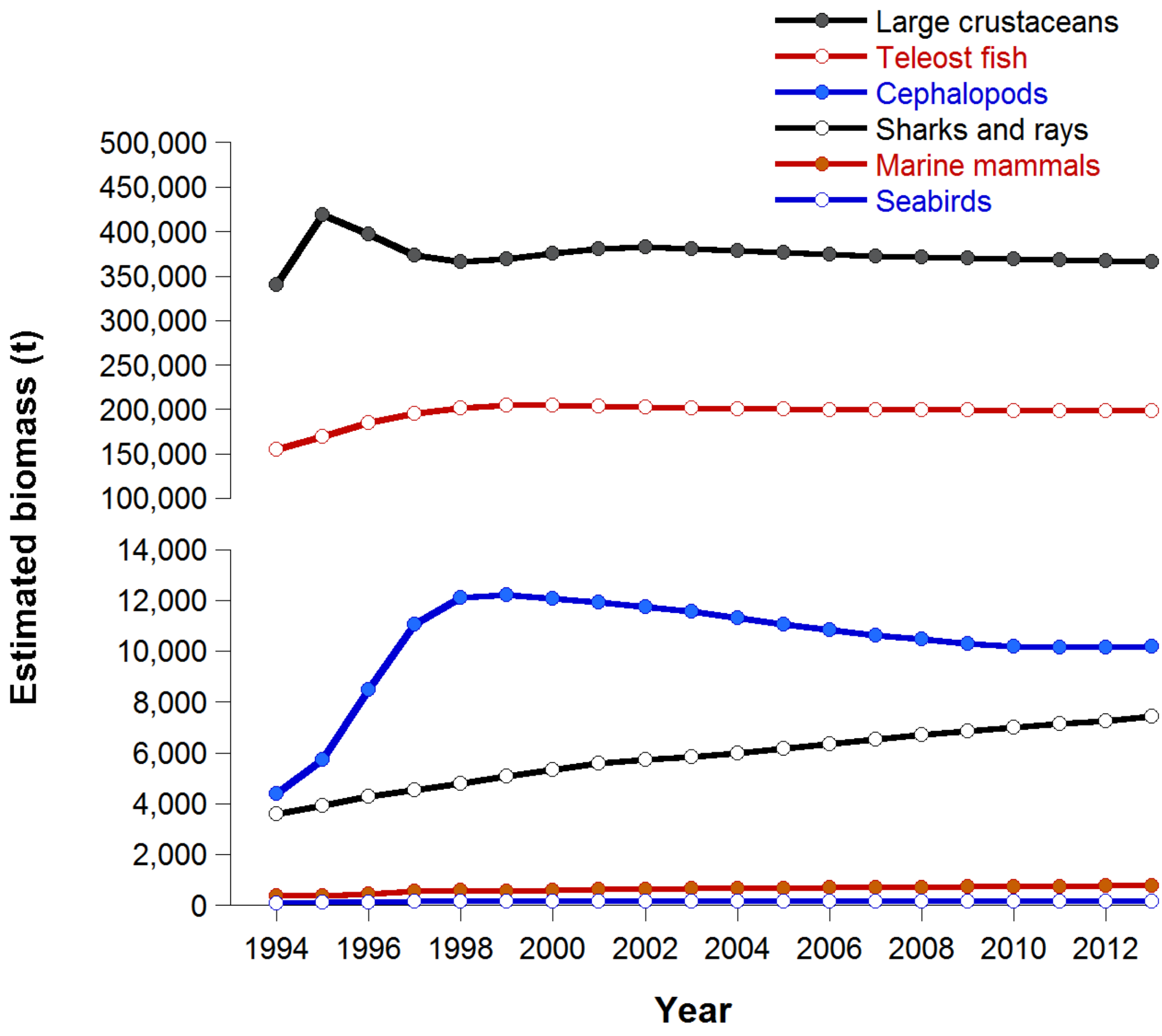


Figure 3.5 Estimated changes in the biomass of major taxonomic groups in the GSV ecosystem groups by year between 1994 and 2013.

a.

b.

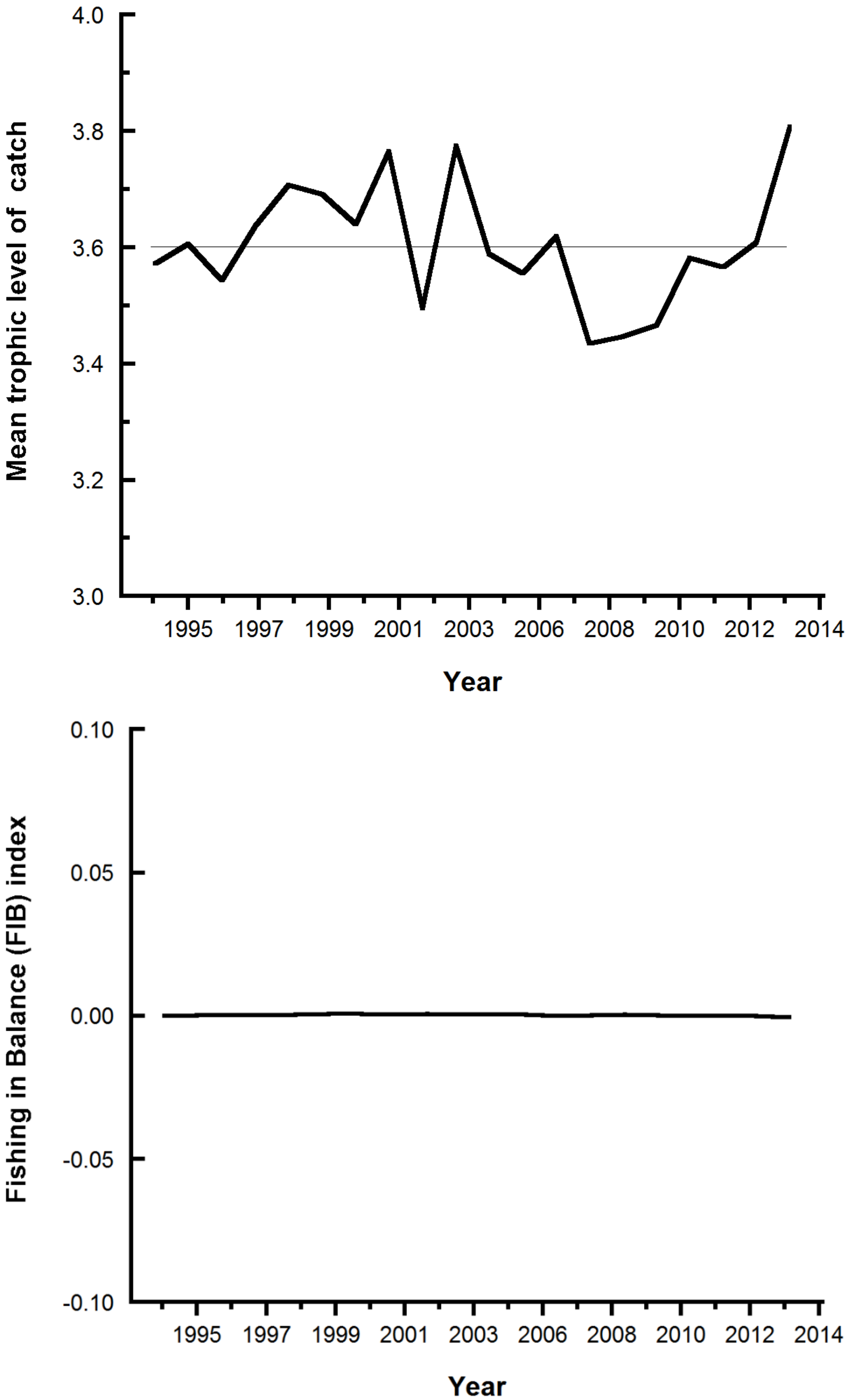


Figure 3.6 Ecosystem indicators calculated from the GSV ecosystem (*Ecopath with Ecosim*) model for the period 1994 to 2013; a) mean trophic level of the catch (line indicating mean for time-series), and b) Fishing in Balance (FIB) index.

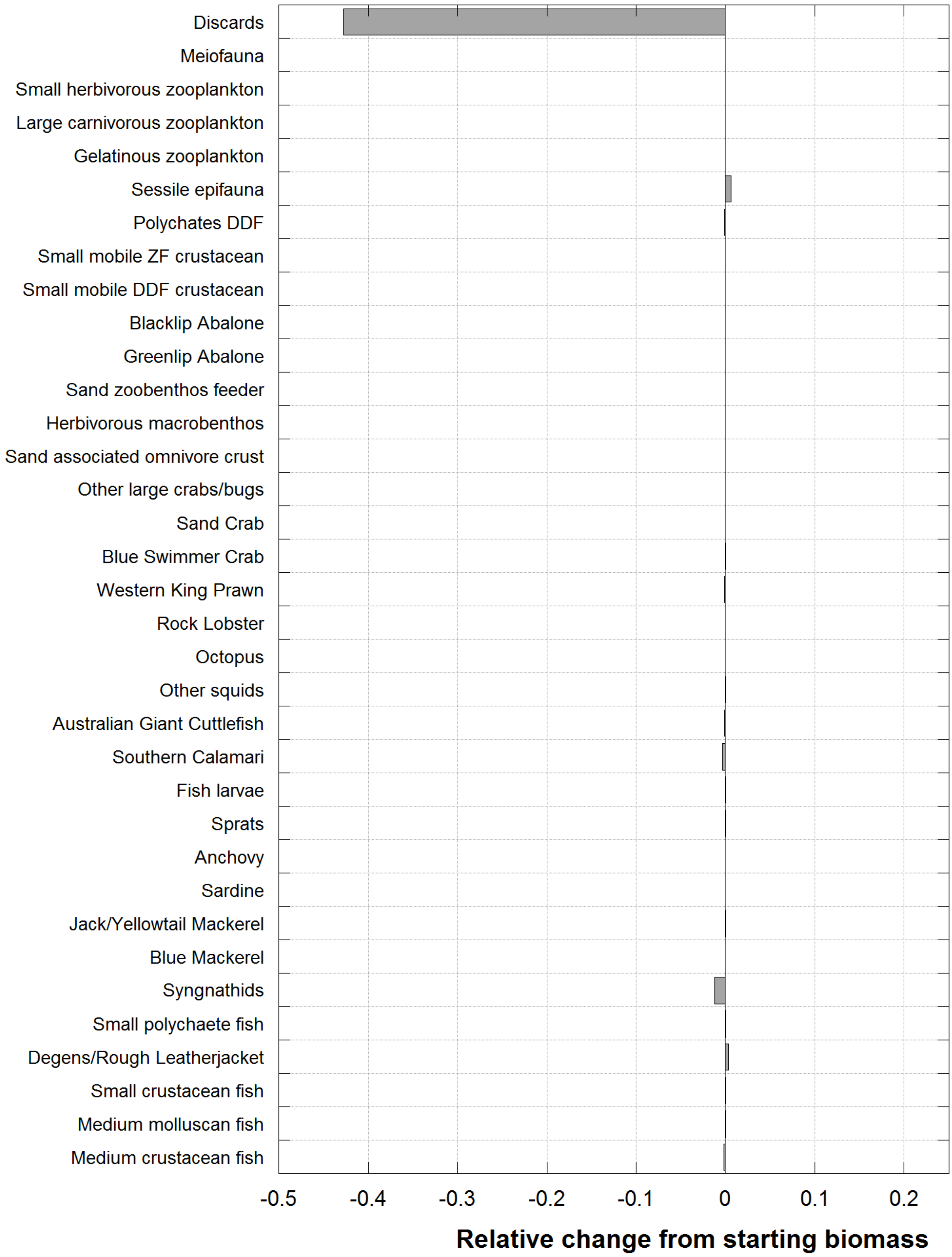


Figure 3.7 Estimated relative change in the biomass of functional groups following introduction of BRD/T90 net in the GSV prawn fishery in the GSV Ecosystem. Biomass change with BRD/T90 (reduced discards) is plotted relative to base model scenario (pre-BRD/T90 net) output run over a 50-year period.

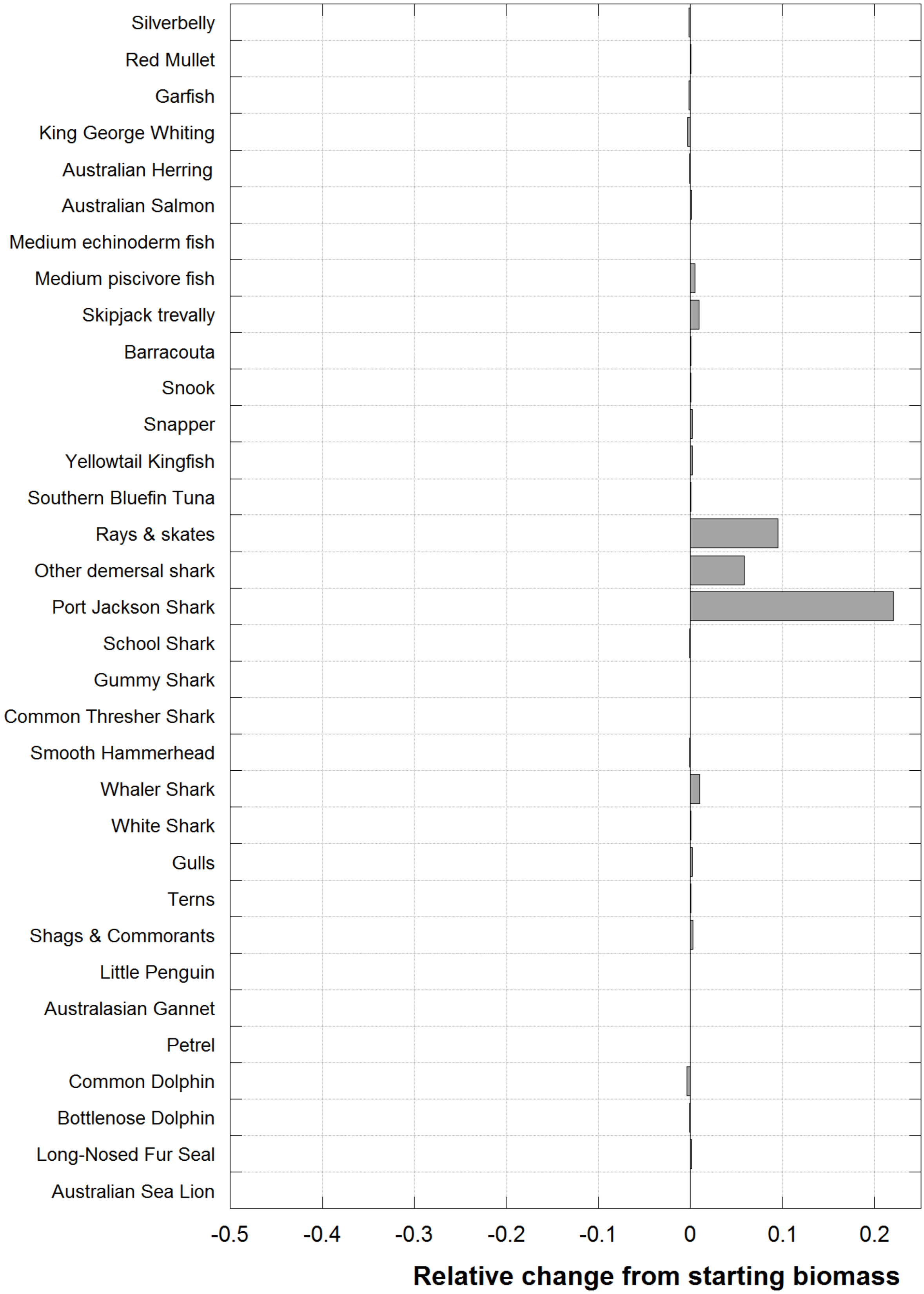


Figure 3.7 continued.

3.2 Trophic role of three key species and impact of Snapper

Diet of three key species

The estimated trophic relationships of the three key species within the GSV ecosystem are presented in Figure 3.8. These demonstrate a complex suite of trophic interactions that connect the species to all trophic levels within the GSV ecosystem. These relationships are dependent on the dietary information available for the key taxa and other groups. The quality of these data and their provenance to GSV also vary markedly among trophic groups.

For Snapper, information on diet was obtained from 735 stomachs sampled across multiple seasons within the GSV region (Lloyd 2010) (see Appendix 2 and 3). Snapper diet included 33 taxonomic groups, but only five of these accounted for >5% of the diet by mass: Blue Crab (23.5%), other large crabs/bugs (18.3%), Australian Herring (10.6%), sessile epifauna (9.6%) and herbivorous macrobenthos (6.8%) (Appendix 2). Figure 3.8a depicts the trophic position of Snapper prey taxa throughout the GSV food web. Snapper predate on a number of other commercially targeted taxa in GSV. With the exception of Blue Crab, all other commercially targeted taxa are consumed at relatively low levels: King George Whiting (2.9%), Australian Salmon (1.2%), Western King Prawns (1.1%), Sardine (0.8%), Southern Calarmari (0.4%), Garfish (0.04%), Southern Rock Lobster (0.03%), and Snapper (<0.01%) (Appendix 3).

The diet of the Blue Crab has not been studied in GSV or other parts of South Australia. The limited information on the diet of the species comes from SW Western Australia where they are reported to have a predatory/scavenging lifestyle, predating mainly on molluscs, crustaceans and polychaetes (Edgar 1990). In Spencer Gulf, Blue Crabs have been reported as being one of the main scavengers on discarded bycatch from prawn trawling (Svane 2003). Given the paucity of information, the diet of the Blue Crab used in the GSV ecosystem model was based on the study by Edgar (1990), descriptions in Bryars and Svane (2008) and that used in the Spencer Gulf ecosystem model (Gillanders *et al.* 2015). Diet was estimated as follows: sand associated omnivorous crustaceans (10%), herbivorous macrobenthos (5%), sand zoobenthos feeders (17%), small mobile deposit-detritovore feeding (DDF) crustaceans (10%), DDF polychaetes (25%), macroalgae (8%), detritus (POM) (10%) and discards (15%) (Appendix 3). However, during the *Ecopath* model balancing procedures, ecotrophic efficiencies (EE) for discards were excessive in early model runs (EE=48.6). To balance the model, all of the detritus fates (100%) for discards were returned to detritus (POM) and the 15% discard diet was added to detritus (POM, total 25%) (see Appendix 3). The predicted trophic position of Blue Crabs and their prey taxa throughout the GSV food web is presented in Figure 3.8b.

As with Blue Crabs, there is little known about the diet of Western King Prawns in GSV. King (1977) considered them to be opportunistic scavengers and observed them feeding on algae and possibly bacterial films on the surfaces of seagrass and shells. They are thought to also scavenge on small dead animals, and take live annelids (Appendix 2). Diet was estimated as follows: DDF polychaetes (10%), meiofauna (10%), benthic microflora (10%), macroalgae (50%), detritus (POM) (20%) (Appendix 2). The predicted trophic position of Western King Prawns and their prey taxa throughout the GSV food web is presented in Figure 3.8c.

Predation on three key species

Ecopath outputs for the GSV Ecosystem model estimated the consumers of Snapper to be ‘other squids’ (87%), Whaler Sharks (11%) and Common Dolphins (2%) (Figure 3.9a). Predation by White and Smooth Hammerhead Shark, and from cannibalism, accounted for <1% of Snapper consumed (Figure 3.9a). The major consumers of Blue Crabs were sand associated omnivore crustaceans (76%) (presumably of juveniles), followed by Snapper (11%), Giant Australian Cuttlefish (5%) and Gummy Shark (5%) (Figure 3.9b). Medium crustacean feeding fish (3%) and gulls (<1%) were relatively minor consumers of Blue Crab (Figure 3.9b). For Western King Prawns, the principal consumers were medium (31%) and small (16%) crustacean feeding fish taxa, medium piscivorous fish taxa (14%), Gummy Shark (13%), Giant Australian Cuttlefish (11%), and other squids (8%) (Figure 3.9c). Other demersal sharks (3%), Snapper (1%) and Port Jackson shark (1%) consumed a relative small proportion of the total biomass consumed (Figure 3.9c). King George Whiting, Australian sea lion, Little Penguin, Whaler Shark, Terns and Smooth Hammerhead all accounted for <1% of the estimated total consumption of Western King Prawns in GSV (Figure 3.9c).

Sensitivity analysis

The mixed trophic impacts routine in the network analysis tools within *Ecopath* was used to identify important interactions that influence the available biomass of the six high value commercial taxa in the GSV ecosystem (Figure 3.10). The Leontif matrix produced through this routine visually represents the effects of increasing biomass of one functional group or fishing fleet on other groups or fisheries, and provides a form of sensitivity analysis (Figure 3.10). The biomass of the six commercial taxa examined influenced their respective fishing fleets positively, and most fishing fleets impacted their fleets and target species negatively (Figure 3.10). In general, groups containing commercially targeted species influenced their respective fishing fleets positively, and most groups affected themselves negatively (Figure 3.10). Snapper negatively impacted King George Whiting and its main fisheries (e.g. hand-line and haul-net) and to a lesser extent Blue Crab and its main fisheries (Figure 3.10). Blue Crab positively impacted Snapper biomass and its main fisheries and negatively impacted King George Whiting and its main fisheries, and to a lesser extent Western King Prawn and the GSV Prawn fishery (Figure 3.10). Western King Prawn had a positive impact on hand-line, long-line and large-mesh fisheries, and a negative impact on Blue Crab and its fisheries (Figure 3.10). King George Whiting had a minor positive impact on Snapper biomass and its fisheries, as well as other hand-line, and small-mesh net fisheries (Figure 3.10). Garfish had a minor negative impact on King George Whiting, its hand-line fishery and the Snapper long-line fishery, but positive impacts on whiting and ‘other’ haul-net fisheries (Figure 3.10). Southern Calarmari had a positive impact on Western King Prawns and its fishery, the Australian herring, King George Whiting and ‘other’ haul-net fisheries for which it also forms part of the landed catch (Figure 3.10). It had a negative impact on King George Whiting and Garfish, on Australian salmon fisheries and handline fisheries for King George Whiting and Snapper, long-line fisheries for Snapper and the Garfish dab-net fishery (Figure 3.10). All these sensitivity analyses assume a steady-state system, and do not take into account the changing abundances or diets of groups.

Impact of increasing biomass of Snapper on GSV ecosystem

A better understanding of the sensitivity of taxonomic groups in the GSV ecosystem to increasing Snapper biomass, can be gained from *Ecosim* scenarios that take into account changing time series and group abundances. Results from scenarios that examined the potential impacts of increasing Snapper biomass by 10%, 20% and 30% of 2013 levels (base model) are presented in Figure 3.11. The majority of groups were relatively insensitive to changes in Snapper biomass. For example, under a scenario with a 10% increase in Snapper biomass, the greatest decrease in biomass was -4.8% (King George Whiting), and increase in biomass was +1.4% (small polychaete feeding fish). With a 10% increase in Snapper biomass, the mean and median biomass declines were just -0.6% and -0.3%, respectively for negatively impacted groups, and just +0.3% and +0.1% respectively, for positively impacted groups (Figure 3.11). Under a scenario with a 30% increase in Snapper biomass, the greatest decrease in biomass was -12.9% (King George Whiting), and increase in biomass was +3.9% (small polychaete feeding fish) and the mean and median biomass declines were just -1.6% and -0.8%, respectively for negatively impacted groups, and just +0.8% and +0.4% respectively, for positively impacted groups (Figure 3.11). Other than King George Whiting, the main groups negatively impacted by increasing Snapper biomass were giant cuttlefish, Australian herring, shags and cormorants, bottlenose dolphin, Australian Sea Lions, Blue Swimmer Crab, Smooth Hammerhead, octopus, and Rock Lobster, but all by <2% under a 10% increase in Snapper biomass scenario (Figure 3.12). Other than small polychaete feeding fish, groups that responded positively to increases in Snapper biomass included medium crustacean eating fish, Sand Crabs, Sprats, Terns, Western King Prawns and Little Penguins, but all by <1% under a 10% increase in Snapper biomass scenario (Figure 3.11).

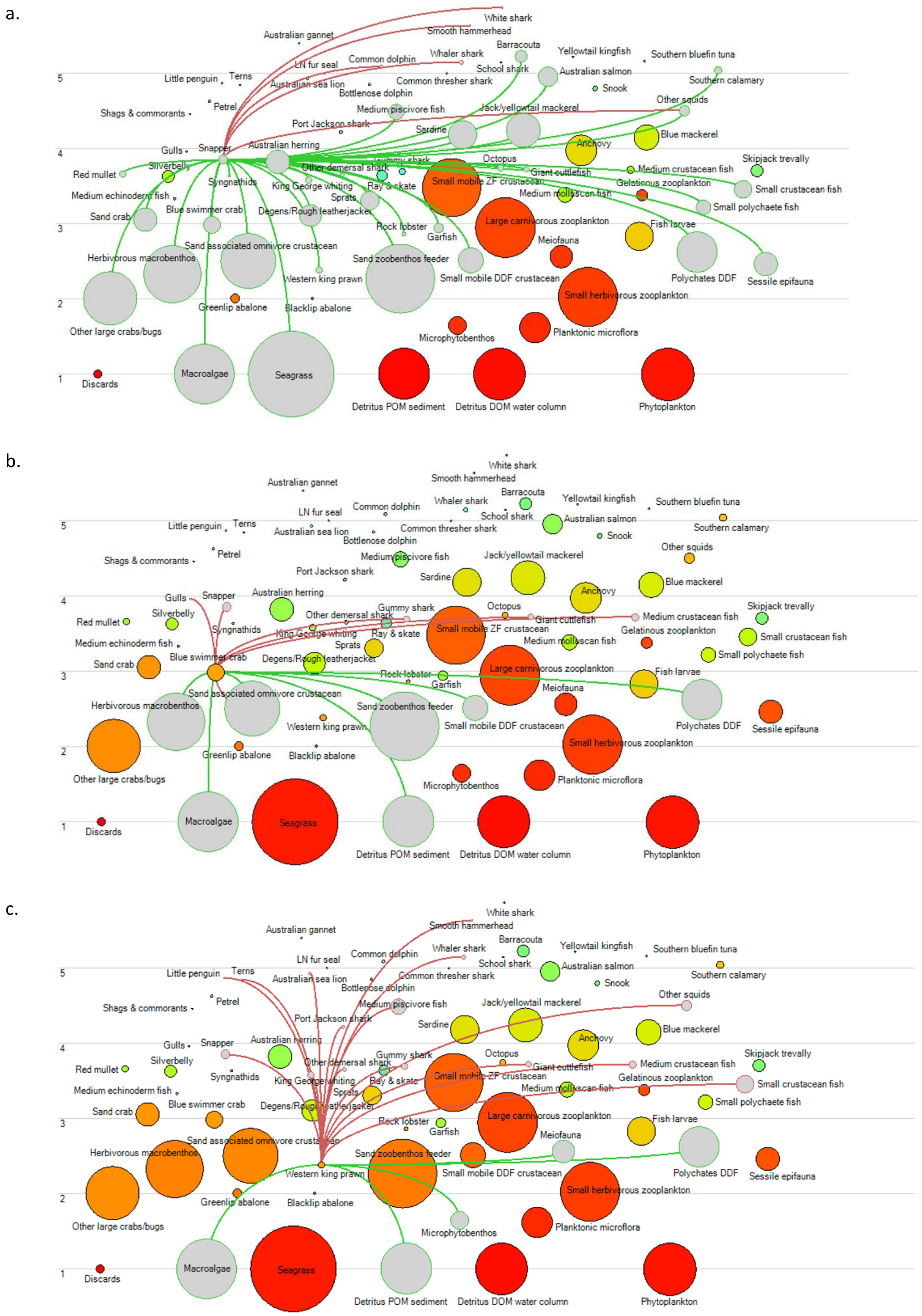


Figure 3.8 Trophic flows in the GSV ecosystem relating to Snapper (a), Blue Crabs (b) and Western King Prawns (c). Green and red lines connect prey and predator groups, respectively.

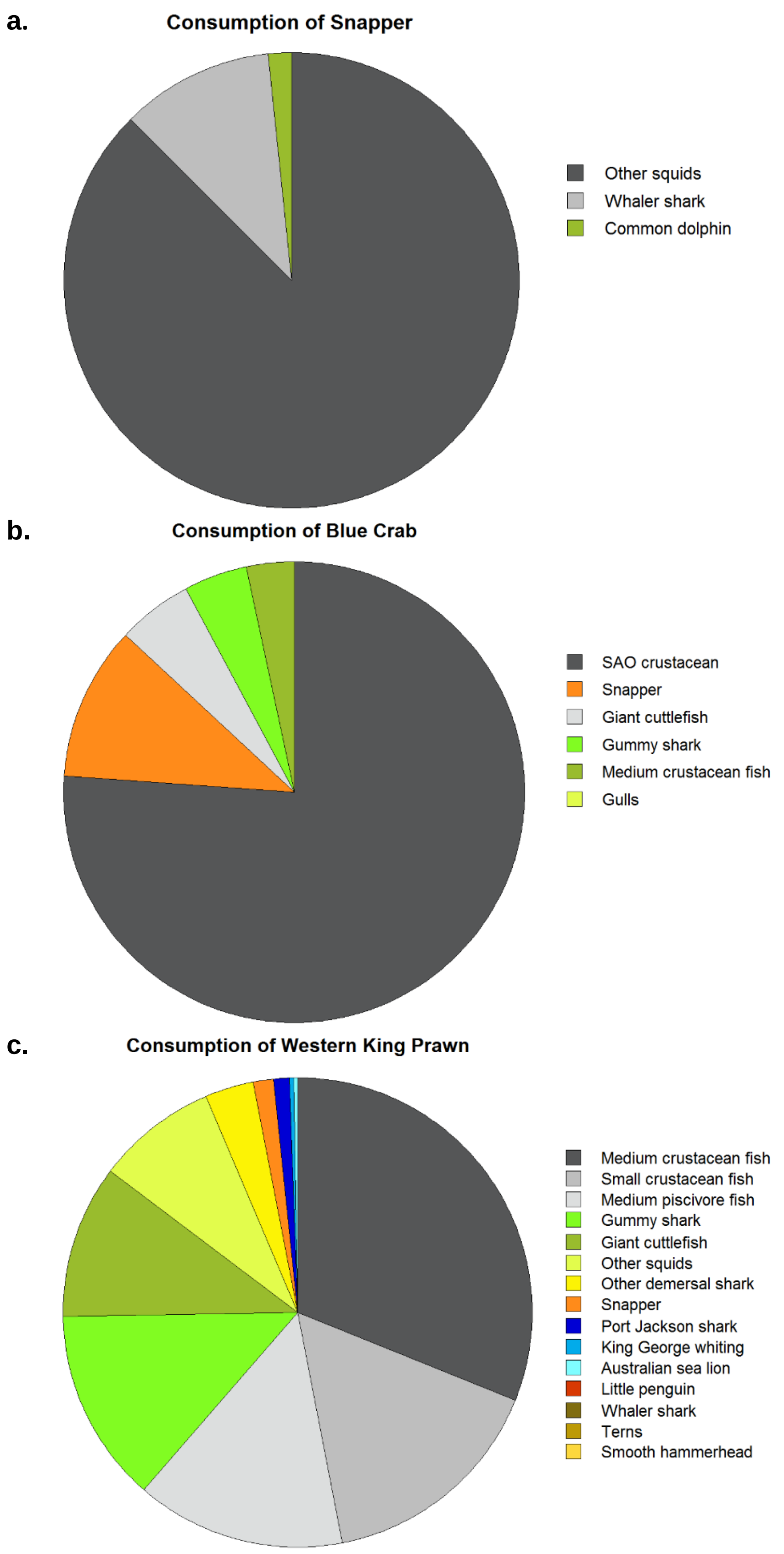


Figure 3.9 Estimated breakdown of relative consumption (key predators) of Snapper (A), Blue Crabs (B) and Western King Prawns (C) in the GSV ecosystem.

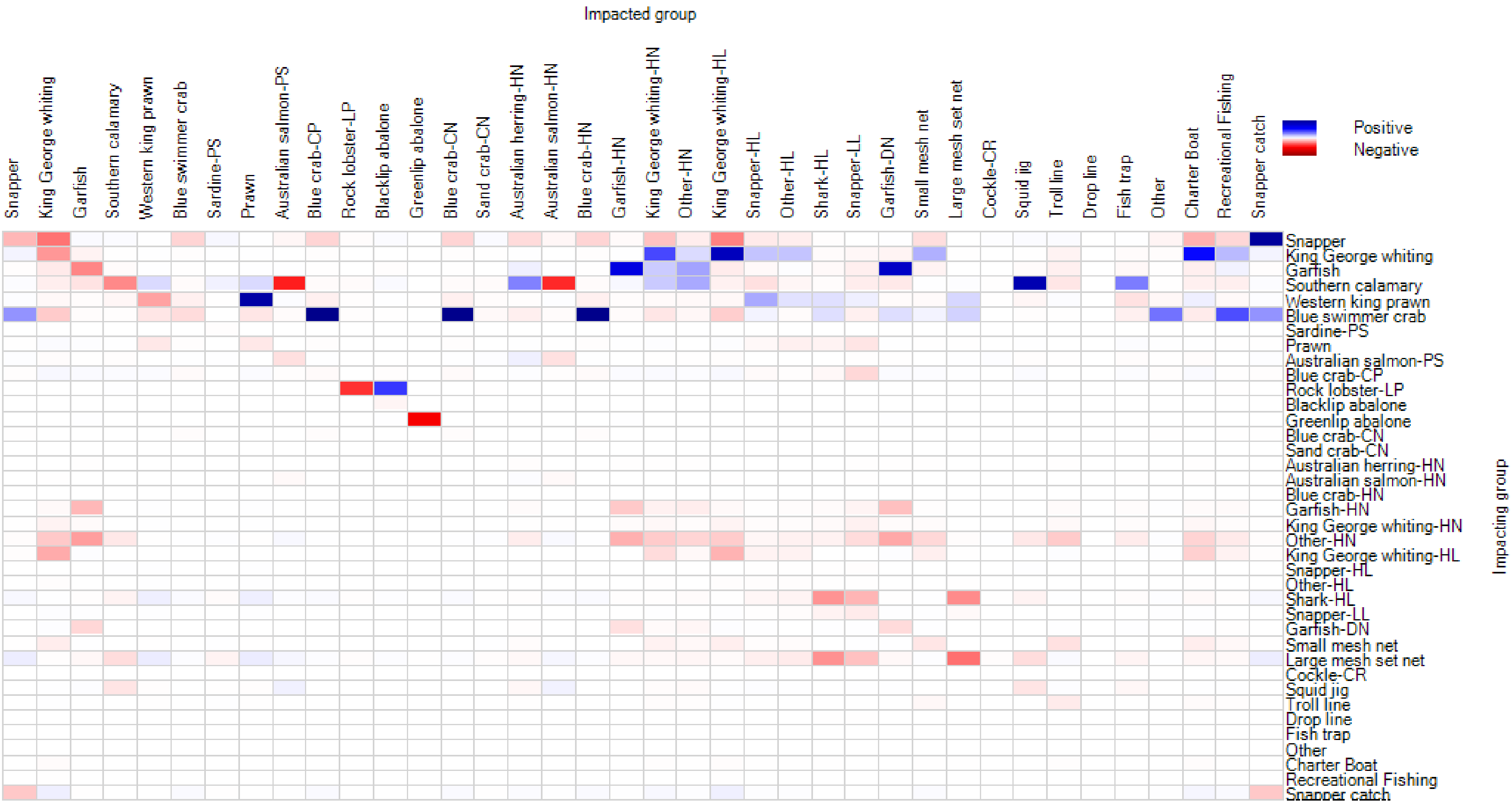


Figure 3.10 Leontif sensitivity matrix showing impacts of increasing abundance of groups on the y-axis on groups on the x-axis. Impacts are expressed as relative % changes, not all impacts are discernible on this figure.

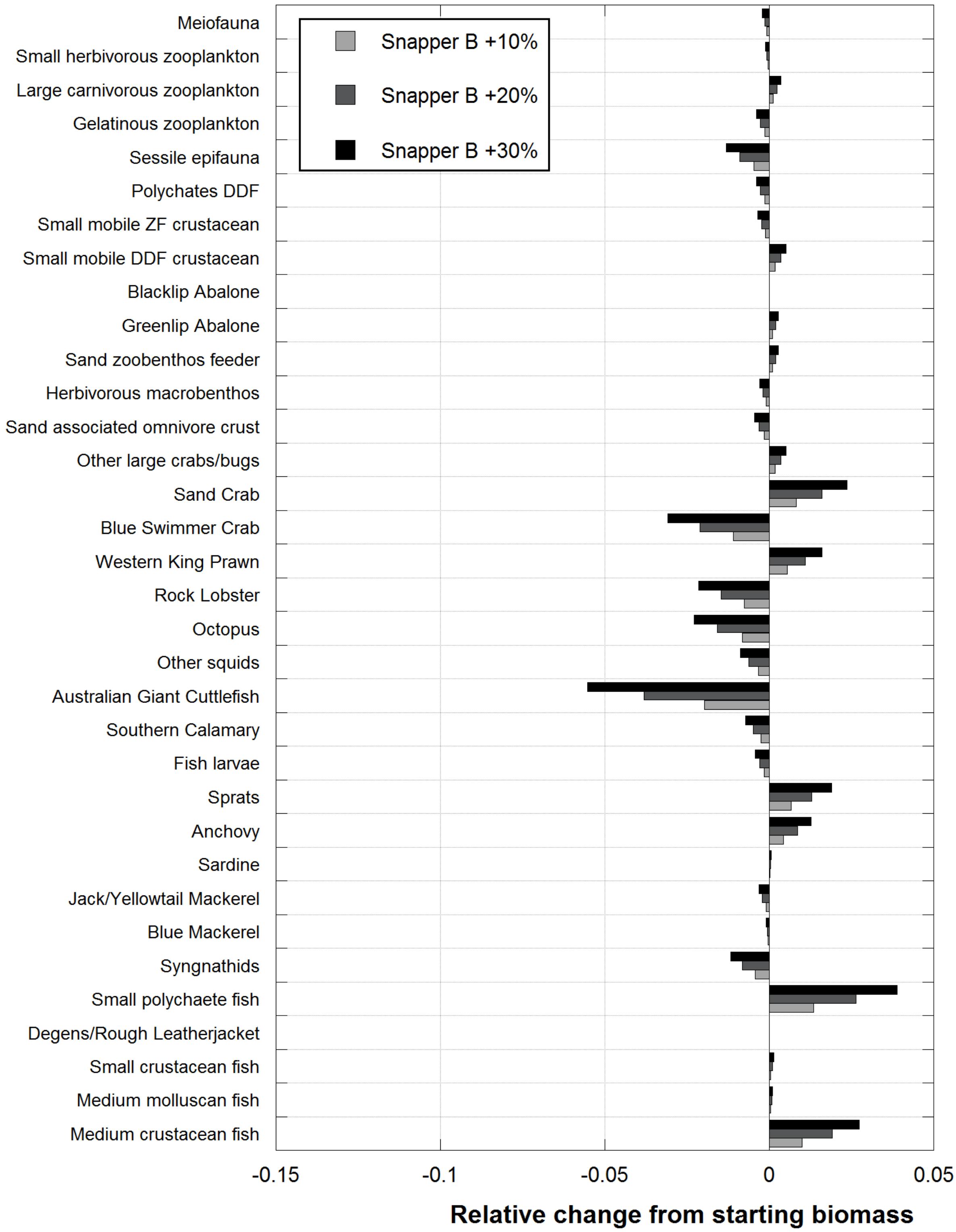


Figure 3.11 Estimated relative change in the biomass of functional groups following scenarios that examined the potential impacts of increasing Snapper biomass by 10%, 20% and 30% of 2013 levels. Biomass change under each scenario is plotted relative to base model (2013) output run over a 50-year period.

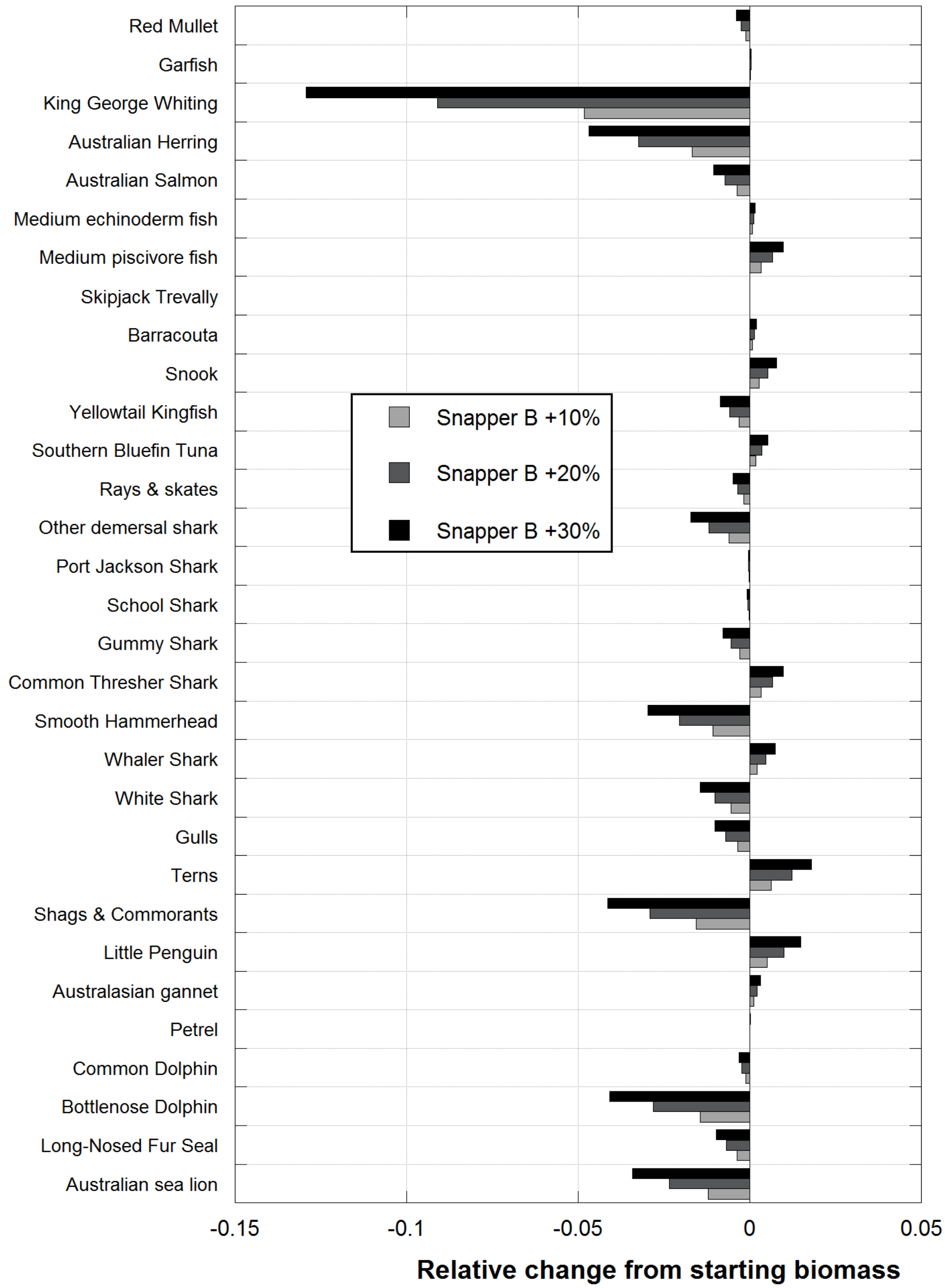


Figure 3.11 continued.

3.3 Optimisation of future ecological and economic performance

In this study, the specific context for optimisation of ecological and economic performance in the GSV ecosystem is centred on the potential that increased Snapper biomass has directly impacted the available biomass of prawns and Blue Crabs, and resulted in reduced catches in recent years in these fisheries. A key aim of the study was to examine the flow-on ecosystem effects of changes in Snapper biomass on Western King Prawns, Blue Crabs and other species; and if these interactions were significant, what balanced exploitation scenarios would optimise production (and value) of Snapper, prawns and Blue Crabs and minimise ecosystem impacts? The previous section identified the impacts of increased biomass of Snapper on key fisheries and the ecosystem, here we examine the impacts of reducing Snapper biomass on Blue Crab and prawn fisheries and other high value commercially fished groups and the broader ecosystem, and the potential benefits from balanced exploitation of Snapper in the GSV ecosystem.

Impact of reducing Snapper biomass on high value commercial fisheries and on the GSV ecosystem

Results from *Ecosim* scenarios that examined potential changes to the GSV ecosystem resulting from reductions in Snapper biomass by 10%, 20% and 30% of 2013 levels (base model) are presented in Figure 3.12. The majority of groups were relatively insensitive to changes in Snapper biomass. For example, under a scenario with a 10% decrease in Snapper biomass, the greatest increase in biomass was 5.4% (King George Whiting), and decrease in biomass was -1.4% (small polychaete feeding fish). With a 10% decrease in Snapper biomass, the mean and median biomass changes were 0.6% and 0.3%, respectively for positively impacted groups, and -0.3% and -0.1%, respectively, for negatively impacted groups (Figure 3.12). Under a scenario with a 30% decrease in Snapper biomass, the greatest increase in biomass was 18.8% (King George Whiting), and decrease in biomass was -4.5% (small polychaete feeding fish) and the mean and median biomass changes were just 2.0% and 1.0%, respectively for positively impacted groups, and -0.9% and -0.4%, respectively, for negatively impacted groups (Figure 3.12). Other than King George Whiting, the main groups that responded positively to reduced Snapper biomass were Giant Australian Cuttlefish, Australian Herring, Shags and Cormorants, Bottlenose Dolphin, Australian Sea Lions, Smooth Hammerhead, Blue Swimmer Crab, Octopus, and Rock Lobster, but all by ≤2% under a 10% decrease in Snapper biomass scenario (Figure 3.12). Other than small polychaete feeding fish, groups that responded negatively to decreases in Snapper biomass included medium-sized crustacean eating fish, sand crabs, sprats, terns, Western King Prawns and Little Penguins, but all by ≤1% under a 10% decrease in Snapper biomass scenario (Figure 3.12).

The estimated impact of reducing Snapper biomass by 10%, 20% and 30% of 2013 levels on the biomass of the other five high-value commercially fished groups in the GSV ecosystem is summarised in Table 3.1. Although not strictly linear, these results indicate that for every 10% reduction in Snapper biomass, there is about a 0.6% decline in prawn biomass, and 0.04% decline in Garfish biomass, a 0.3% increase in Calarmari biomass, a 1.2% increase in Blue Crab biomass and a 5.4% increase in King George Whiting biomass (Table 3.1). These estimates assume no change in fleet fishing effort or other environmental factors throughout the scenario period.

Table 3.1. The estimated impact of reducing Snapper biomass by 10%, 20% and 30% of 2013 levels on the percentage biomass of the other five high value commercially fished species in the GSV Ecosystem.

Snapper	-10%	-20%	-30%
Western King Prawn	-0.6%	-1.2%	-1.8%
Garfish	-0.04%	-0.1%	-0.2%
Calarmari	0.3%	0.5%	0.8%
Blue Swimmer Crab	1.2%	2.4%	3.7%
King George Whiting	5.4%	11.7%	18.8%

The potential impact of Snapper on other high-value, commercially-fished groups was also explored by running the GSV ecosystem *Ecosim* model under different scenarios of Snapper biomass (while maintaining all fleet fishing efforts at 2013 levels). By running scenarios from near zero to high biomass levels (0.7 t km^{-2}), the potential impact of different possible Snapper biomasses (potentially brought about by targeted reductions in biomass) on other high-value commercial-fished groups (while not forcing biomass change in any other group), was examined. Results are presented in Figure 3.13. As detailed above, the results suggest that as the biomass of Snapper increases, the biomass of Blue Crabs, King George Whiting, and Southern Calarmari decline, while the biomass of Western King Prawns and Garfish increase (Figure 3.13). The relationship between Snapper biomass and other group biomasses is close to linear for Garfish, Southern Calarmari and Western King Prawns, but is more curvilinear for Blue Crab and King George Whiting.

Between 1994 and 2013, estimates of the fishable biomass of Snapper in GSV almost trebled (Fowler *et al.* 2013). Results from *Ecosim* scenarios examining the direction and magnitude of biomass change in the key fished species that would result if Snapper biomass was reduced from this peak by two-thirds (from 0.3998 to 0.1333 t.km^{-2}) to 1994 levels are presented in Table 3.2. The estimated change in the biomass in groups (while not forcing biomass change in any other group and holding all fleet fishing efforts constant), when Snapper biomass is reduced by 66.7% was a decline in prawn biomass of 4.9%, an increase in Blue Crab biomass of 10.9%; a decrease in Garfish biomass of 0.6%; an increase in King George Whiting biomass of 64.5%; and an increase in Calamari biomass of 2.4% (Table 3.2). These modelled results are different to the observed and estimated biomasses from fisheries data and the balanced *Ecopath* model of the key fished taxa in 1994, especially for King George Whiting (-41.5% difference), Calarmari (-25.1% difference), prawns (24.1% difference) and Garfish (-16.6% difference) (Table 3.2).

Table 3.2. *Ecosim* scenarios examining the direction and magnitude of biomass change in the key fished species that would result if Snapper biomass was reduced from its peak in 2010 by two-thirds (from 0.3998 to 0.1333 t km^{-2}) to 1994 levels. Comparison of actual biomass at 1994 level in the *Ecopath* model are presented for comparison and the magnitude of difference is presented as a percent (in parentheses).

	Estimated biomass of groups at peak Snapper biomass (~2010)	Modelled change in biomass if Snapper reduced to 1994 levels	Actual biomass at 1994 levels (% difference between modelled and actual biomass)
Snapper	0.3998	0.1333 (-66.7%)	0.1333 (0.0%)
Prawns	0.0821	0.0780 (-4.9%)	0.0968 (24.1%)
Blue Crab	0.4625	0.5130 (10.9%)	0.5298 (3.3%)
Garfish	0.1813	0.1802 (-0.6%)	0.1504 (-16.6%)
Kin George Whiting	0.0846	0.1391 (64.5%)	0.0813 (-41.5%)
Calarmari	0.1165	0.1194 (2.4%)	0.0893 (-25.1%)

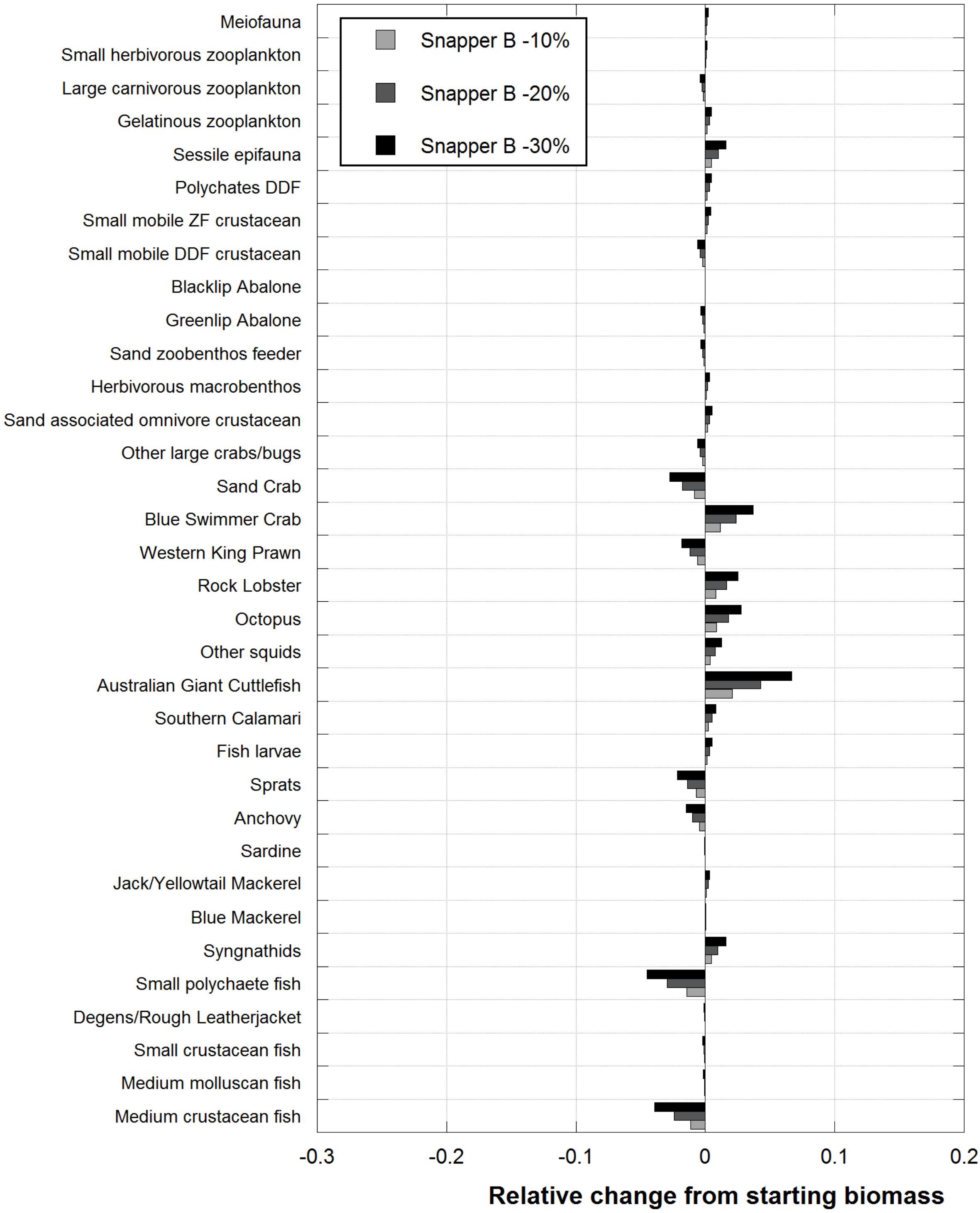


Figure 3.12 Estimated relative change in the biomass of functional groups following scenarios that examined the potential impacts of decreasing Snapper biomass by 10%, 20% and 30% of 2013 levels. Biomass change under each scenario is plotted relative to base model (2013) output run over a 50-year period.

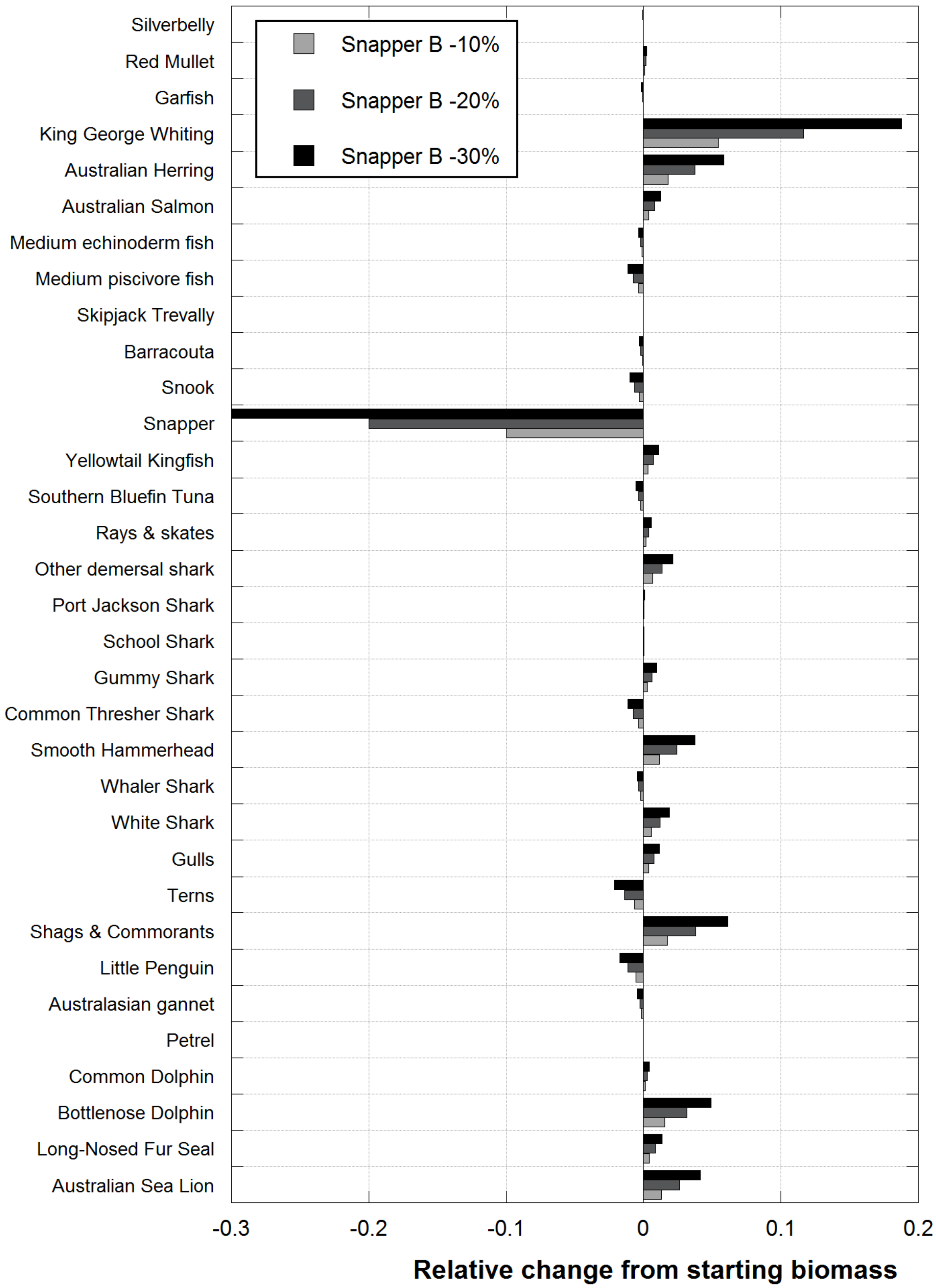


Figure 3.12 continued.

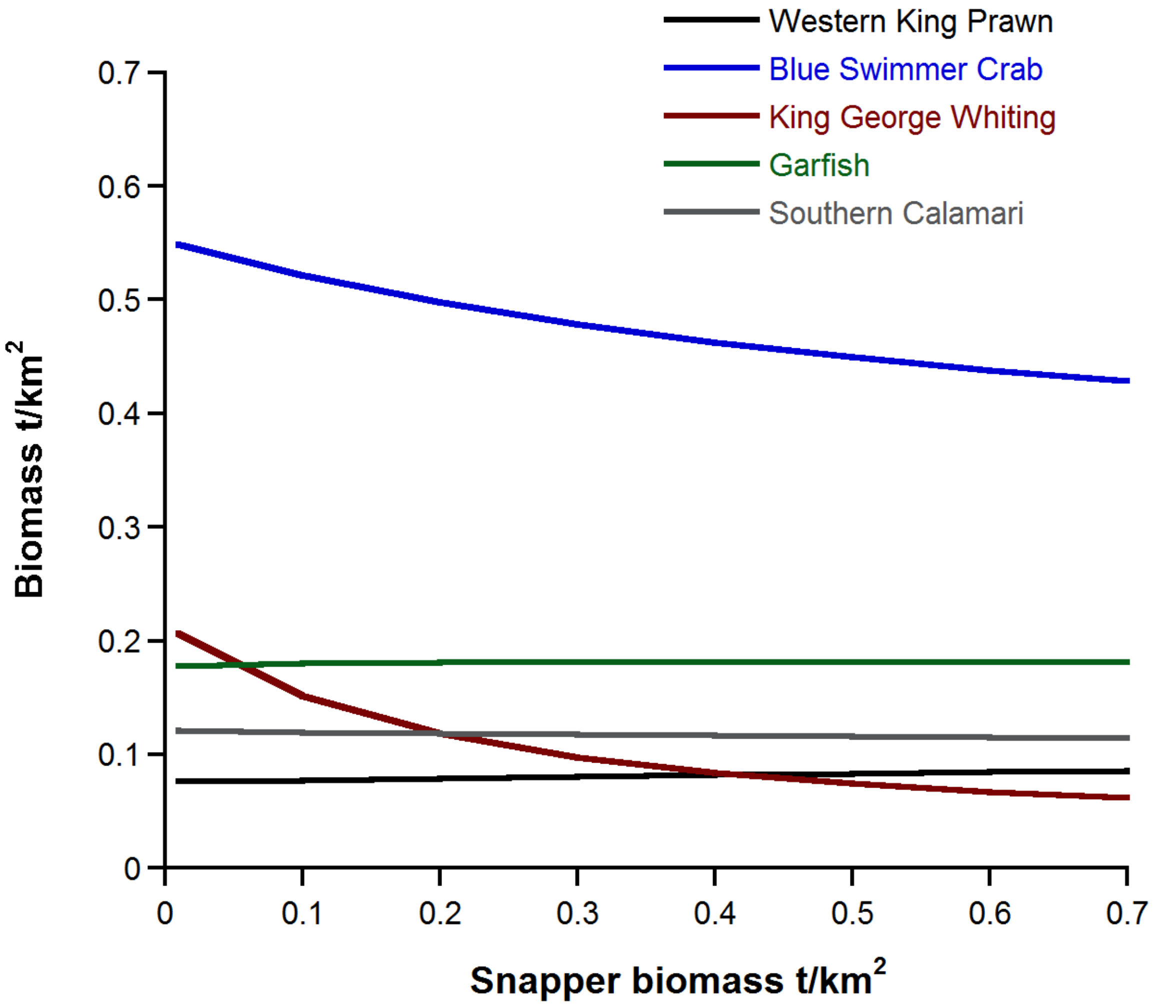


Figure 3.13 Estimated relative change in the biomass of high value commercially fished groups under different scenarios of Snapper biomass (while maintaining all fleet fishing efforts at 2013 levels).

4 Discussion

To our understanding, this is the first quantitative ecosystem model to be developed for the GSV ecosystem, and the first attempt to integrate all fisheries time-series on catch and effort and examine them in an ecosystem context. A key feature of the GSV ecosystem is the biomass dominance of crustaceans (Figure 3.1). The prevalence of omnivorous, scavenger and detritivore crustaceans shows marked similarity to that of Spencer Gulf ecosystem (Gillanders *et al.* 2015). Blue crabs and Western King Prawns are important taxa in this community.

Dynamic management arrangements in the GSV ecosystem

This section outlines the complex management changes that have occurred in the GSV ecosystem for each key commercial species over the 20-year study period (1994-2013). The ecosystem model focused on the fishery changes over this period, the most profound being the marked decline in catch and effort. Depending on the metrics used, this amounted to a reduction by more than half (54%) based on catches in the first and final year, or by about one-third (32%) in mean catches between the first and last five years of the study period (excluding the episodic catches of sardine, which by volume regularly exceed all other combined catches in the region). Part of the declines in catch in the GSV ecosystem is attributed to major reductions in effort targeting Australian Salmon (~90% reduction in catch) following the ending of purse seining; a major reduction in large mesh net effort directed to gummy and school shark that resulted in a ~60% decline in combined catch of these species following transfer of management from State to Commonwealth in 2000, and net closures to this fishery in GSV.

The six main commercially targeted species in GSV (prawn, Blue Crabs, King George Whiting, Garfish, Snapper, Calarmari) have also seen major changes in catch over the 20-year study period. These amounted to 25% and 37% declines in Blue Crab and prawn catch (respectively), and 54% and 43% declines in King George Whiting and Garfish catches (respectively), between the first and last five years of the study period. In contrast, Calarmari catch has remained relatively stable, while the catch of Snapper increased almost 6-fold (Figure 3.3). Declines in catch and effort have occurred in response to changes in the biomass of stocks, and in response to marked changes in fisheries management arrangements and fishing methods (gear and efficiencies). These changes for each of the key GSV fisheries over the last 20 years, are summarised below, with reference to GSV ecosystem model outputs where relevant.

Snapper

The main fishing gears used to target Snapper in GSV by commercial fishers in the Marine Scalefish Fishery (MSF) are handlines and longlines, which account for most of the 81% of total catch taken by the commercial sector. Handlines are used by recreational fishers, accounting for about 19% of the total catch (Fowler *et al.* 2013). The Snapper fishery in SA is geographically extensive and assessed in regions: Northern GSV(NGSV); Southern GSV (SGSV); Northern Spencer Gulf (NSG); Southern Spencer Gulf (SSG); the South East (SE); and the West Coast (WC). The Snapper fishery is managed through a combination of input and output controls, with limited entries (licence holders) for commercial fishers, and a legal minimum total length of 38 cm. Gear restrictions include a ban on targeting Snapper using fish traps or nets (net ban introduced in 1993), and the number of hooks on set lines was reduced from 400 to 200 (for fishers in Spencer Gulf and GSV) in 2012 (Fowler 2008). A daily commercial catch limit of 500 kg was also introduced in 2012. State-wide seasonal closures were introduced in 2000, initially two three-week closure periods in August and November, with the August closure being removed in 2003 and the November closure extended over the entire month (Fowler 2008). A review of management arrangements in 2011 and 2012 identified the benefits of seasonal closures in reducing effort and protecting spawning aggregations (to optimise spawning and recruitment), and in 2012 the seasonal closure was extended to 15 December. Furthermore, from 2013 specific spatial closures were extended through to the end of January for parts of NGSV and NSG to protect important spawning aggregations (Fowler *et al.* 2013). For recreational fishers, in addition to seasonal closures, there are bag and boat limits of 5 and 15, respectively for GSV (including

Investigator Strait and Backstairs Passage) and a size limit of 38-60 cm total length (Fowler *et al.* 2013).

Historically, the catch of Snapper from NGSV and SGSV has been low (~8% of South Australian catch between 2000 to 2004), with most of the catch originating from both regions in Spencer Gulf (Fowler 2008). Commercial catches of Snapper in GSV were relatively high in the late 1970s and early 1980s, peaking at 117 t in 1984/85, before declining to their lowest minimum of 23 t in 1995/96 (Fowler 2008). However, since 2006 there has been a marked increase in the catch of Snapper, especially in NGSV, leading to commercial catches peaking in 2009 and 2010 (Fowler *et al.* 2013). This increase was associated with a major shift in fishing effort away from the traditional areas of Spencer Gulf, where catch rates declined significantly, to NGSV. It was also associated with a major shift from handline to more efficient longline fishing methods, where targeted catches, effort and catch rates all increased exponentially to record levels, seeing the NGSV contribution to South Australian Snapper catches increase from 9% in 2007 to 65% in 2012 (Fowler *et al.* 2013). Assessment of the Snapper fishery in 2013 identified that all fishery regions (with the exception of NGSV) demonstrated declines in catch and catch rates consistent with decreases in fishable biomass, resulting in the NSG, SSG, WC, SE and SGSV Snapper stocks all being assigned a status of 'transitional depleting'. In contrast, the NGSV stock was assigned a status of 'sustainable' based on it producing record catches and catch rates that reflected high biomass and strong recruitment throughout the 1990s and 2000s (Fowler *et al.* 2013). Assessment in 2016 has demonstrated no recovery for NSG, SSG, WC, SE, while high catches have persisted in NGSV (Fowler 2016).

There has been uncertainty in whether the regional shift in Snapper catch, and specifically the simultaneous decline in catches from Spencer Gulf as catch in GSV increased, was coincidental or related (Fowler *et al.* 2013). In particular, if the declines in biomass of Snapper in Spencer Gulf were related to poor recruitment or if there was some large-scale movement of Spencer Gulf fish into GSV. A recent otolith study that examined age-related increment width and elemental concentrations identified three primary nursery areas for South Australian Snapper: NSG, NGSV and Port Phillip Bay in Victoria (Fowler 2016). Each of these populations was found to be self-sustaining but also to provide the source fish that replenish regional populations. The study showed that recent declines in Snapper biomass in Spencer Gulf were primarily due to persistent high exploitation rates as well as poor recruitment throughout the 2000s; while the growth in biomass of the GSV population from the late 2000s was due to local reproduction and recruitment, and not the movement of fish into the region from other populations (Fowler 2016). The GSV ecosystem model fit to Snapper time series data predicted a gradual increase in Snapper biomass in GSV throughout the 2000s and did not match the low level of catch throughout the 2000s, with the marked increase in catch from 2008, instead projecting a more gradual increase in catch (Figure 3.2). Part of the marked increase in catch and effort for Snapper in GSV was related to the switch from handline to longline fishing that increased efficiency and attracted more fishers to targeting Snapper and increased the targeted fishing effort, much of which switched from Spencer Gulf (where catch rates had declined) to GSV (Fowler 2016).

GSV Prawn fishery

High exploitation of the spawning biomass of Western King Prawns in GSV in the 1970s and 1980s resulted in reduced recruitment and stock depletion (Dixon *et al.* 2012, Kangas and Dixon 2008). A range of management measures were introduced including major reductions in fishing effort; an increase in the size of targeted prawns; restricting the area of the fishery to protect the spawning stock; the introduction of a licence buy-back scheme followed by a total closure of the fishery between June 1991 and March 1994 following a parliamentary review (Kangas and Dixon 2008), and no pre-Christmas fishing in the 1994/95 season (Dixon *et al.* 2012). The GSV ecosystem model time-series commenced in the year the GSV Prawn fishery was reopened in 1994, and from then to 2000, prawn catch and CPUE increased, suggesting that the closure period over four spawning seasons enabled the stock biomass to increase (Dixon *et al.* 2006). However, increased catches and a harvest strategy based solely on harvest size precipitated further stock declines between 2000 and 2004 (Kangas and Dixon 2008), resulting in a new management strategy being implemented for the fishery that included the introduction of independent stock assessment and the current harvest strategy. The harvest strategy aimed to control exploitation rates using spatial and temporal closures, restricting the fleet to targeting limited areas with appropriately sized prawns, with the intent to protect a sufficient portion of the stock to enable stock recovery (Dixon *et al.* 2012). These measures appeared to enable both catch and biomass of prawns to increase until 2008/09, but following declines in the catch rates of adult prawns and poor economic performance through to the 2011/12 season, the fishery was closed for two consecutive fishing seasons (2012/13 to 2013/14) (Beckmann *et al.* 2015). Part of the most recent declines in biomass and catch has been attributed to a shift in size structure in the population, particularly an increase in the abundance of smaller prawns that made it difficult for fishers to locate suitable areas with appropriate size prawns (Dixon *et al.* 2013a). This led industry to consider approaches to improve the size selectivity of their trawl gear (Dixon *et al.* 2013a). As a consequence, modified T90 cod-end mesh and BRDs were introduced into the GSV prawn fishery in March 2012 (Beckmann *et al.* 2015). Fishing trials demonstrated that these devices improved gear selectivity, specifically the escapement of small pawns, enabling increased tow durations and trawl effort efficiencies (Dixon *et al.* 2013a). The fishery was reopened in November 2014, and although subsequent stock assessment surveys have noted an increase in adult biomass following the recent closures, low catch rates of recruits suggest limited recruitment to the harvestable biomass. As such the GSV Prawn fishery has recently been classified as 'transitional-depleting' (Beckmann *et al.* 2015). The GSV ecosystem model provided a reasonable fit to the prawn biomass and catch time-series, although clearly anticipated a gradual recovery in prawn biomass from the late 1990s through to 2013 (Fig 3.2, 3.3), which is not reflected in the CPUE data for the fishery (Beckmann *et al.* 2015).

GSV Blue Crab fishery

Historically, Blue Crabs were mainly taken as bycatch in the Marine Scalefish and Prawn fisheries, although the provision to sell Blue Crabs caught in the Prawn fishery was revoked in 1986. The modern fishery was established in 1996 comprising two fishing zones (Spencer Gulf and GSV), each with a separate quota (TACC), and consisting of commercial 'pot fisheries' and MSF licence holders, with most of the quota now allocated to pot fishers (Beckmann and Hooper 2015). Blue Crabs are widely caught among fishing sectors including the recreational fishery, where recent catches are estimated to represent about 30% of the total combined commercial and recreational catches (Beckmann and Hooper 2015). There are some differences in the reported catch data presented in stock assessment reports, which do not include discards, and those used in the GSV ecosystem model that includes both landings and discards. Based on the later data, Blue Crab catch was reasonably stable (~300 t year^{-1}) throughout the late 1990s to the mid-2000s, but then declined from about 2006 to the end of the time-series in 2013. CPUE of legal size and pre-recruit crabs in the GSV fishery declined markedly between 2006 and 2012, and the stock was classified then as 'transitional-depleting' (Dixon *et al.* 2013b). The management initiatives to promote stock recovery then included 50% voluntary reduction in catch in 2012/13 (through effort reduction), a quota reduction of 20% in 2013/14 and 2014/15, a six month voluntary closure in July 2013, and a halving of recreational bag and boat limits in 2013/14 (Beckmann and Hooper 2016). The low 2012 and 2013 Blue Crab catches for the GSV ecosystem (Figure 3.3), reflect these management measures. The GSV ecosystem model provided a reasonable fit to the Blue Crab biomass (generally increasing) and catch (generally declining) throughout the time-series, but anticipated generally lower catches of Blue Crab than those observed from the early 2000s to 2013 (Figure 3.2).

Garfish

The commercial fishery for Garfish in GSV has undergone significant changes in operation and management over the last 40 years that include: restructuring; gear limitations, configurations and restrictions; licencing; spatial and temporal closures; and size limits. These changes, summarised by Steer *et al.* (2016), include netting restrictions introduced in the 1970s (restricting fishers to net in coastal waters <5 m deep), and netting closures introduced in 1983, 1994, 1995, 1997, 2005 and 2006 that have resulted in a continued contraction of the areas open to haul-netting in northern GSV. Although occurring just outside the time series of this study, the introduction of marine parks in 2014 further restricted the area available to haul-netting in northern GSV. Other important management measures include: an increase in the minimum total legal size of Garfish from 210 mm to 230 mm and reductions in recreational bag limits in 2001; a voluntary net buy-back scheme in 2005 to reduce fishing effort (reduced haul-netting licence holders by 54%); introduction of seasonal closures in 2012, 2013 and 2014 (increased from 20 to 38 to 40 days, respectively); increases to the minimum regulated mesh size of haul-net pockets (to 32 mm in 2013, and 35 mm in 2015); and further increases to the minimum legal size of Garfish in 2015 (250 mm) and 2016 (260 mm) (Steer *et al.* 2016).

Most of the Garfish catch is taken by commercial MSF licence holders (~90% haul-net, ~10% dab net), with recreational catch increasing from 18% in 2000/01 to 23% in 2013/14 (Steer *et al.* 2016). The GSV region contains two Garfish management zones (NGSV and SGSV). The NGSV zone is the second most productive Garfish region in SA, accounting for about 35% of the State's annual catch (Steer *et al.* 2016). Between 1993 and 2013, catch in the SGSV zone steadily declined from ~70 t to <10 t per year due to a steady decline in haul and dab net fishing effort (Steer *et al.* 2016). In the NGSV zone, annual catches have exceeded 200 t twice in the last 20 years (2000 and 2005). Annual catch declined to 97 t in 2007 with a concomitant 22% decline in haul-net effort and a 35% reduction in CPUE. Catch and effort declined further to their lowest values in 2012 (82 t in NGSV), when winter fishing closures were first implemented. Estimates of fishable biomass and recruitment in NGSV have both trended downward since 2000, reaching their lowest recorded levels in 30 years in 2014 (Steer *et al.* 2016). On this basis, a recent assessment classified the NGSV Garfish stock as 'overfished' (Steer *et al.* 2016). In contrast, the SGSV Garfish stock, which accounts for a negligible component of the GSV Garfish fishery, was classed as 'sustainable'(Steer *et al.* 2016). The GSV ecosystem model provided a good fit to Garfish catch throughout the time-series, but contrary to observed declining trends in biomass, anticipated increasing biomass over the time-series (Figure 3.2). The estimated fishable biomass of Garfish is based on that available in the areas open to the fishery, which has contracted over time, and as such may not reflect the biomass of the entire stock.

King George Whiting

Commercial targeting of King George Whiting (KGW) is principally undertaken with three gear types: handlines, haul nets and gillnets. Recreational fishers principally use hook and line gear from boats. The principal areas of the fishery are the West Coast bays, Northern Spencer Gulf and GSV. The KGW fishery in GSV is a 'gauntlet' fishery, with juvenile fish (~3 years of age) being targeted when they move from shallow, protected nursery areas to adjacent deeper waters of GSV (Fowler and Jones 2008). The predominant take in NGSV is therefore of juvenile fish that are close to the minimum legal size, while fish in the southern spawning grounds tend to be larger and older. Changes in the management and performance of the fishery have been detailed by Fowler *et al.* (2014). In 1994, reductions in daily recreational bag (30 to 20 fish per person) and boat limits (90 to 60 fish per boat) were introduced, and the minimum legal size of fish was increased from 28 to 30 cm total length in 1995. Concern about the status of the fishery in 2003 resulted in significant management changes introduced in 2004 that included: an increase in the minimum legal length (30 to 31 cm); a reduction in the daily recreational bag and boat limit (from 20 to 12 legal-sized fish per person; boat limit reduced from 60 to 36 fish); enhancement of the existing licence amalgamation scheme; and possession limits for non-licence holders. The MSF net-buy back scheme in 2005 (as above for Garfish), resulted in significant (~45%) reduction in haul net and gillnet fishing effort. Commercial catch of KGW in GSV declined steadily over the study period from ~145 t in 1994 to 45 t in 2013. Handlines and haul nets account for most of the catch in NGSV, gillnets now account for a small portion of the total catch, and most of the catch off Kangaroo Island in Investigator Strait is with handlines. As commercial catch has declined, the relative contribution of the recreational catch has increased and now accounts for more than 60% of the total catch in GSV (Fowler *et al.* 2014). Recent assessment of the fishery has noted persistent declining catch and catch rates over the last 20 years, with the shift in effort away from KGW suggesting a decline in fishable biomass. Effort creep (vessel speed and navigational equipment) may mean that declines in fishable biomass over the last decade have been greater than suggested by reductions in CPUE. Furthermore, recreational catches have only been estimated for three seasons (2000/01, 2007/08, 2013/14) and if the recreational catches have increased since 2007/08, then the decline in fishable biomass would be greater than estimated. Based on declining trends in commercial catch, effort, CPUE and estimated fishable biomass to 2013, the GSV stock was classified as 'transitional-depleting' (Fowler *et al.* 2014). The GSV ecosystem model provided a good fit to the KGW biomass and marked decline in catch throughout the time-series, (Figure 3.2).

Calarmari
In GSV, southern Calarmari are landed by commercial MSF fishers (~62% of catch), by recreational fishers (~30%) and incidentally in the GSV prawn fishery (~8% of catch) (Steer *et al.* 2007). Commercial catches in GSV fluctuate from year to year, from a low in 2001 (133 t) to a high in 2004 (258 t) (Triantafillos 2008). Calarmari are targeted with haul nets and jigs, with hauling net catch focused in NGSV and jigging in SGSV (Triantafillos 2008). In the early 1990s, catch was evenly distributed between these two sectors, but with the decrease in haul net effort the jig sector catch has grown. In 1995, recreational bag and boat limits were introduced (15 per bag and 45 per boat day). The same spatial and temporal closures detailed above for net fishers in GSV, and gear restrictions (mesh size) apply, as well as the generic licence amalgamations and voluntary buyback schemes, but are not specific to Calarmari (Steer *et al.* 2007). For south and central GSV which accounts for the largest Calarmari catch in SA (28% in 2006), CPUE for the jig sector has shown a general increase between 1993 to 2006 (Steer *et al.* 2007). The last assessment of the fishery within GSV was undertaken in 2007 (Steer *et al.* 2007). The stock status of the GSV fishery has not undergone a formal assessment since then. The GSV Ecosystem model provided a reasonable fit to the time series data for Calarmari biomass, with the fit to catch data better since the mid-2000s (Figure 3.2).

Trophodynamic indicators of fisheries status and impact

Trophodynamic indicators are broadly used in marine ecology and can be useful in identifying the expansion or contraction of fisheries, and quantifying the effects of fishing on the trophic structure and functioning of marine ecosystems (Tsikliras *et al.* 2015). Two key indicators used extensively are the mean trophic level of the catch (or marine trophic index, MTI) and the Fishing In Balance (FIB) index, which have proven robust in tracking fishing effects (see MTI and FIB trends for all maritime countries and Large Marine Ecosystems of the world at www.seaaroundus.org), and in assessing ecosystem changes (Kleisner and Pauly 2011, Tsikliras *et al.* 2015). Typically the two measures are assessed together, with the interpretation of one facilitating the other (Kleisner and Pauly 2011).

In the 20-year study of the GSV ecosystem, the mean trophic level of the catch averaged ~3.6 and remained relatively stable, but could be broadly divided into three periods: 1994 to 2001 (increasing slightly to ~3.8); 2003 to 2007 (decreasing slightly to ~3.4) and 2007 to 2013 (increasing to 3.8). The increase in mean trophic level of the catch between 2007 and 2013 occurred as a consequence of the marked increase in catches of Snapper and concomitant declines in the catches at lower trophic levels (Blue Crab, Prawns and Garfish), and steady catches of Calamari (TL>5).

As a consequence of the marked reduction in fishing effort and landed and discarded catches between 1993 and 2014, the biomass of most trophic groups in the model was estimated to have increased over the study period. However, this does not imply that over the study period, these groups were not subject to the effects of fishing, or other ecosystem changes, and the FIB index can provide some insight into potential ecological change and impacts of fishing on the GSV ecosystem. In absolute terms, the FIB index for the 20-year study period in GSV has remained stable, noting that the scale of change in FIB index reported for other fisheries-impacted ecosystems globally are generally in the order of 0.1 to 1.0 or more over time (Large Marine Ecosystems of the world at www.seaaroundus.org, Kleisner and Pauly 2011, Tsikliras *et al.* 2015). In contrast, the range of FIB index values for the 20-year study period in GSV was two to three orders of magnitude lower. The FIB index is set to 0 for the first year of the time-series and will remain stable when the trophic level of the catch and catches change in opposite directions (i.e. where trophic level changes are matched by 'ecologically equivalent' changes in catch, Kleisner and Pauly 2011, Tsikliras *et al.* 2015). For example, with transfer efficiencies of ~10% between trophic levels there should be a ten-fold increase in catches when fishing shifts one trophic level down (Christensen 2000). If this occurs, then the FIB index should remain constant, and fishing is assessed to be 'in balance' (Christensen 2000). The FIB index will increase when catches increase more than expected when fisheries move to a lower trophic level, and when the geographic area expands beyond its traditional fishing area or ecosystem (Coll *et al.*

2009, Pauly and Palomares 2005). In contrast, the FIB index will decline when catches fail to increase as much as anticipated when a fishery moves to a lower trophic level, if the geographic area of the fishery contracts, or if the underlying food web is collapsing through the excessive removal of biomass (Kleisner and Pauly 2011, Pauly *et al.* 2000).

The relative stability of the ecosystem indictors (FIB index and mean trophic level of catch) for GSV is challenging to reconcile against the marked changes in fisheries catch and effort and management dynamics over the last two decades. The GSV Prawn, Blue Crab and KGW fisheries were all classified as transitionally depleting between 2012 and 2014, with the main GSV Garfish stock classified as overfished, indicating the deteriorating performance of the main fisheries in GSV towards the latter stage of the study period. The considerable management adjustments in these fisheries over the last decade, which have resulted in a contraction in their spatial extent and reductions in fishing effort and catch, could be expected to have resulted in a marked decline in the FIB index. In contrast, an increase in the FIB index may have been expected following the marked increase in Snapper catches between 2007 and 2013, a concomitant increase in mean trophic level of the catch, and with the periodic marked increases in overall catches in GSV due to sardine in the second decade of the time-series. Despite these marked changes the FIB index and mean trophic level of the catch have remained relatively stable. How do we reconcile these observations?

As indicated above, the FIB index will remain stable when the trophic level of the catch and catches change in opposite directions, or are stable. For fishing to have remained in balance with stable trophic level of the catch while catches have declined, suggests that the productive capacity of the GSV ecosystem and/or its resilience to fisheries exploitation has diminished over the 20-year study period. Loss of productivity could have occurred as a consequence of environmental or ecological change (e.g. reduced primary productivity, seagrass loss) or other factors that may have impacted production of the ecosystem (e.g. pollution). Loss of production may also reflect additional loss of biomass (catch) to the GSV ecosystem that has not been adequately estimated by the model. This could include an under-estimation of discarding (not reported as catch), or recreational fishing catches. The available data for discards and recreational catches are both poor for the GSV ecosystem. As no time-series of recreational fishing catch and effort are available, the model undoubtedly under-estimated the increasing level of recreational fishing catch in GSV, as indicated by recent recreational fishing surveys (Giri and Hall 2015). Any loss of production to the ecosystem over time, either through environmental change, underestimated catches (discards or recreational fishing), or other impacting factors or processes, will potentially contribute to the situation where catches fail to remain stable when fishing is in balance and occurring at the same trophic level.

In summary, the broad picture of the GSV ecosystem over the 20-year period of the study is of persistent declines in total catch (by about one-third), reduced performance (declines in catches and fishable biomass) in four of the six key fisheries over the last decade and the classification of their main GSV stocks as either transitional-depleting (Prawn, Blue Crab, KGW) or overfished (Garfish), and significant fisheries management intervention. The overall performance of the key commercial fisheries in GSV, despite contractions in effort and catch, is at odds with the key ecosystem indicators of a general increase in the biomass of most non-targeted trophic groups, and stable FIB index and mean trophic level of the catch. These inconsistencies in fishery performance and ecosystem indicators suggest that the ecosystem is not performing as well as expected, and is less resilient to change because there has been a net loss in production or productivity over time. This net loss of production may be real (environmental change and/or degradation), an artefact of underestimated catches (discards/recreational fishing), and/or data limitations and assumptions of the modelling approach.

Ecosystem implications of greater gear selectivity in the GSV prawn fishery

A major advance in increasing the selectivity of fishing methods in the GSV Prawn fishery was the introduction of rigid-grid BRDs and T90-mesh cod-ends in 2012. Experimental trials identified that the use of BRDs with T90 nets substantially reduced sponge, elasmobranch and fish bycatch, improving fishing efficiency, and excluded juvenile prawns, small fish and crustaceans from the catch (Dixon *et al.* 2013a). Other benefits included improved catch quality (fewer damaged prawns), trawling efficiency (due to longer trawl shot duration) and fuel efficiency (total fuel consumption per hour of trawling). The expectation was that the reduced bycatch would also lead to good environmental outcomes. We used the GSV ecosystem model to examine the likely ecological changes to the GSV ecosystem from reduced bycatch levels in the prawn fishery.

Using the levels of bycatch reduction reported for sponges, elasmobranchs and fish when using the BRD/T90 net (Dixon *et al.* 2013a), we compared the ecosystem response by comparing the estimated biomasses of groups under pre- and post-BRD/T90 fishing scenarios, run over a 50-year period (2014 to 2063). Results indicated a ~43% reduction in discards under post-BRD/T90 fishing scenarios, as well as an increase in the biomass of demersal chondrichthyians, particularly Port Jackson sharks (~22%), rays and skates (~10%) and other demersal sharks (~6%). All other group biomass changes estimated under a post-BRD/T90 fishing scenario were relatively minor, and there were no projected negative impacts on any of the key fished species. These results therefore, support the expectation that the introduction of more selective fishing methods in the GSV Prawn fishery were likely to have positive ecological benefits, and importantly here we also identify that they were unlikely to have negative impacts on the biomasses of key commercial taxa. As we did not develop any multi-stanza models, which incorporate both juvenile and adult age-classes for key species, we were not able to examine the ecological change and stock biomass implications of greater escapement of undersized prawns that result from improved gear selectivity, but this could be examined with further development of the GSV ecosystem model.

Trophic relationships – role of Snapper in GSV ecosystem

The importance of Snapper as a predator in the GSV ecosystem, and its potential impacts on key crustacean fisheries were the central questions for this study. An understanding of the diet of Snapper and the importance of its predation and competitive interactions on other taxonomic groups in the ecosystem were fundamental to addressing these questions, which we examined by developing the GSV ecosystem model. Underpinning much of this assessment are data from Lloyd (2010), who undertook a comprehensive analysis of diet by examining the stomach contents of 1,068 Snapper collected between 2008 and 2010 in Spencer Gulf and GSV. Of these 735 stomach samples were obtained within the GSV ecosystem domain, across all seasons in both SGSV and NGSV. Data from these samples were used and integrated into the ecosystem model. Lloyd (2010) found significant differences in the diet of Snapper by region, season and fish size. He found that Blue Crabs were the dominant prey in NGSV, followed by mussel *Modilus areolatus*, and Red Swimmer Crabs (*Nectocarcinus integrifrons*). Other important prey included Snapping Prawn (*Alpheus villosus*), Mantis Shrimp (*Erugosquilla graham*), Facetted Crabs (*Actea calculosa*), Sea Slugs (*Philine angasi*), Western King Prawns, Blood Worms (*Glycera americana*) and Razor Clams (*Pinna bicolor*). Blue Swimmer Crabs were absent from the diet of Snapper in SGSV, and crustaceans of much less importance than in NGSV with the top three species being Doughboy Scallops (*Mimachlamys asperrrima*), File Clam (*Limatula strangei*) and Thorny Sea Urchins (*Goniocidaris tubaria*). Other important prey species included Goose Barnacles (*Ibla quadrivalvis*), Wrinkled Swimcrabs (*Liocarcinus corrugatus*), Facetted Crab (*Actea calculosa*), Spider Crabs (*Schizophrys rufescens*) and Common Hermit Crabs (*Paguristes frontalis*). Seasonal differences were also marked. In NGSV, the consumption of Blue Crab and Western King Prawn was highly seasonal and largely restricted to the warmer months of summer and autumn, the mussel *Modilus areolatus* featured most in spring and summer, and Red Swimmer Crabs in winter and spring. In SGSV, doughboy scallops peaked in spring and the File Clam peaked in autumn while thorny sea urchins were present in similar amounts throughout the year. In NGSV, medium and large sized Snapper consumed more Blue Crab and mussel *Modiolus areolatus*, and large Snapper consumed more red swimmer crab and mantis shrimp, while small Snapper consumed more Western King Prawns and Blood Worms. In contrast, no significant ontogenetic difference in diet was detected for Snapper of different size in SGSV (Lloyd 2010).

Integration of the diet of Snapper from the study of Lloyd (2010), along with dietary information on Blue Crab, Western King Prawn and other taxonomic groups into a trophic model of the GSV ecosystem, enabled the impact of predation by Snapper on key fished species to be examined. Within this context, the diet of Snapper was extremely broad, including taxa across four trophic levels. The trophic model also enabled the relative importance of Snapper predation on key taxa, relative to other predators of these groups to be determined. Results from analysis of the GSV ecosystem model estimates that Snapper accounted for 11% of Blue Crab predation, and about 1% of king prawn predation. Other crustaceans were the major consumer of Blue Crab (~76% by sand associated omnivorous crustaceans), while medium (31%) and small (16%) crustacean feeding fish taxa, medium piscivorous fish taxa (14%), gummy shark (13%), giant cuttlefish (11%), and other squids (8%) were the major consumer of Western King Prawns.

Across the taxonomic groups in the GSV ecosystem, most were relatively insensitive to changes in Snapper biomass. A 10% increase or decrease in Snapper biomass resulted in negligible changes (usually a fraction of 1%) in the biomasses of impacted groups. *Ecosim* modelling indicated that as Snapper biomass increased, Blue Crab biomass decreased, but these declines were relatively small (i.e. <2% decline in Blue Crab biomass for every 10% increase in Snapper biomass). Contrary to expectation, increases in Snapper biomass resulted in an increase in Prawn biomass, although the gain was relatively minor (i.e. <1% increase for every 10% increase in Snapper biomass). Although Snapper predate on Prawns, this predation impact was less than the predation and/or competitive impacts that Snapper apply to other Prawn consumers, particularly small crustacean feeding fish, Giant Cuttlefish and other squids that collectively are estimated to consume 35% of King Prawn biomass. Snapper may therefore play an ecological role in the regulation of mesopredator populations. Mesopredators are medium-sized, middle-trophic level predators that both predate and are predated upon. Many of the species that predate on crustacean populations in GSV, including crustacean feeding fish, cuttlefish and other squids, could be described as mesopredators. Snapper may form part of the guild of apex predators within the GSV ecosystem that are important in regulating mesopredator populations. The 'mesopredator release' hypothesis is an important ecological theory that describes the population dynamics that occur as a consequence of the trophic interactions between apex predators and mesopredators, where major declines in the former may lead to eruptions ('mesopredator release') of the latter (Baum and Worm 2009, Ritchie and Johnson 2009). There are many examples of such patterns in marine ecosystems where reductions in apex predators such as marine mammals, sharks and piscivorous fish have led to increases in mesopredator populations (Baum and Worm 2009). Although Snapper also predate on Garfish, the *Ecosim* models indicated that Snapper have a minor positive impact on Garfish biomass, potentially through similar mesopredator control, here through preferential predation on Australian Salmon, Calarmari and Giant Australian Cuttlefish.

The most notable negative impact of increasing Snapper biomass identified through the *Ecosim* models, was the interaction with KGW. Even though KGW formed a relatively small component of Snapper diet in the GSV ecosystem model (2.9%), the *Ecosim* analysis identified that for every 10% increase in Snapper biomass, the biomass of KGW declined by 4.8%.

In summary, ecological modelling has shown that the impact of Snapper on Blue Crab and prawn biomass in the GSV ecosystem is relatively minor and that increasing biomass of Snapper is unlikely to have been significant in explaining decreases in the biomass of Blue Crabs and prawns. Although declines in Blue Crab and prawn catch rates have occurred over the same period as Snapper biomass has increased, ecological modelling suggest these declines have principally arisen due to other factors.

Could a directed fishery for Snapper be used to optimise ecological and economic performance of Blue Crab and prawn fisheries?

Ecosystem based fishery management strives to maintain healthy marine ecosystems and the fisheries they support by mitigating many of the unintended consequences of fishing, including habitat destruction, bycatch of non-target species and impacts to ecosystem structure and function (Pikitch *et al.* 2004). It has been recognised that to achieve EBFM, there is a need to address both fishing intensity and selectivity, the latter through one or more of 6-S selection strategies of: species, stock, size, sex, season and space. It has been argued that the focus on the 6-S selection may lead to unintended consequences that ultimately increase, and not decrease ecosystem-level effects of fishing, and reduce the productive capacity of ecosystems to sustain fishery catches (Zhou *et al.* 2010). Less selective fishing, where stocks are harvested relative to their productivity while maintaining species, stocks, sexes and sizes above certain thresholds, has been suggested as a better approach to achieve EBFM. This concept of 'balanced exploitation' encompasses reduced fishing effort with less selective fishing and better utilisation of catch to maximise sustainable yields and minimise ecological impacts (Zhou *et al.* 2010). A key aim of this study was to examine if a balanced exploitation approach in GSV, productivity dependent exploitation of Snapper biomass through a directed fishery, could deliver benefits to other fisheries, in particular whether this approach could enhance the sustainable yield of Blue Crab and prawn fisheries, and what balanced exploitation scenarios would optimise production and value across fisheries, while minimising ecosystem impacts.

This study has shown that although Snapper predate on Blue Crab and King Pawn, the trophic linkages between these groups are not particularly strong (e.g. slight negative effect on Blue Crabs, slight positive effect on King Prawns), and the benefits to Blue Crab and Prawn biomass (and catch rates) from a directed Snapper fishery to reduce its biomass are unlikely to bring about significant benefits to fisheries targeting these species. *Ecosim* scenarios where Snapper biomass was reduced by two-thirds from its peak biomass in 2010 to 1994 levels (while not forcing biomass change in any other group and maintaining constant fishing effort across all fleets), resulted in a decline in prawn biomass of ~5%, and an increase in Blue Crab biomass of ~11%. These model results suggest that the declining catches of prawns and Blue Crabs that occurred concurrent with a marked increase in Snapper biomass, were likely to have been caused by factors other than changes in the biomass of Snapper.

The study focused on the potential of top-down (predation) control by Snapper on Prawn and Blue Crab biomass. It was unable to examine the range of other, bottom-up factors that could have contributed changes in Prawn and Blue Crab biomass in the GSV ecosystem. There is almost no data on the diet of Prawns and Blue Crabs in SA's Gulfs, and our understanding of the trophic interactions between the significant crustacean communities in these ecosystems and the key environmental factors that drive recruitment and regulate their populations is poor. Addressing such critical gaps in knowledge should be prioritised in future studies, because such data deficiencies and model assumptions limit the range of scenarios that can be examined and could impact model results.

5 Conclusion

This project has developed the first whole of ecosystem model for GSV that describes the key components of the ecosystem, providing an integrated assessment of the status, performance and impact of its key fisheries between 1993 and 2014.

This integrated assessment identified temporal changes in the GSV ecosystem. In terms of fisheries impacts, major change occurred as a consequence of declines in fisheries catch and effort, which in general has resulted in an increase in biomass of most trophic groups over the 20-year period.

Analyses of key ecosystem indicators identified a relatively stable mean trophic level of the catch and a stable Fishing in Balance (FIB) index, against a backdrop of declining fisheries catches. At the end of the study time-series, four of the five main fisheries that had undergone recent stock assessments in the region were classified as either transitional-depleting or overfished, with only Snapper classed as sustainable.

The poor performance of key commercial fisheries, despite marked contractions in effort and catch, is at odds with the key ecosystem indicators of a general increase in the biomass of most taxonomic groups, and stable FIB index and mean trophic level of the catch. These inconsistencies suggest that the ecosystem is not performing as well as expected, and that there may have been a net loss in production or productivity over time. Some of this could be explained by an underestimation of fishing catch (commercial discards and/or recreational catch), through natural and/or anthropogenic environmental change (e.g. climate change, pollution, habitat loss) or model limitations.

Scenarios comparing the ecosystem response pre- and post-introduction of BRD/T90 fishing gear in the GSV Prawn fishery, supported the expectations that the introduction of more selective fishing methods would bring about positive ecological benefits, such as increases in the biomass of demersal sharks and rays, with no negative impacts on other commercially targeted species (namely Western King Prawns and Blue Crabs).

The ecological role of Snapper and its impact on crustacean and other fisheries was assessed. Snapper are an important predator in the GSV ecosystem, consuming a diverse range of prey across four trophic levels, with much of their predation directed to crustaceans and molluscs. Blue Crabs were important in Snapper diet (~24%), especially in northern GSV, but other commercially targeted species were consumed at low levels (King George Whiting 2.9%, Western King Prawns 1.1%, Calarmari 0.4%, Garfish 0.04%). Ecosystem model results estimated that Snapper accounted for ~11% of the total consumption of Blue Crabs, and ~1% of Western King Prawns, with most of the consumption of Blue Crabs and prawns being accounted for by omnivorous crustaceans and a range of crustacean and piscivorous feeding fish, respectively.

Sensitivity and scenario analysis of ecosystem responses to changes in Snapper biomass, suggests that most taxonomic groups are relatively insensitive to changes in Snapper biomass (<1% change in biomass for each 10% change in Snapper biomass). Increasing Snapper biomass had a negative effect on Blue Crab biomass (<2% decline for each 10% increase in Snapper biomass), and a slight positive effect on Western King Prawn biomass (~1% increase for each 10% increase in Snapper biomass). Snapper may form an important predator of mesopredators in the GSV ecosystem, providing a positive benefit to Prawns and Garfish through predation and competitive interactions with their key predators (Australian salmon, Calarmari and Giant Australian Cuttlefish). Results from the ecosystem model indicate that the increase in Snapper biomass in GSV is unlikely to have contributed significantly to the observed reduction in biomass of Blue Crab and Prawn.

Scenarios examining the potential ecological and economic benefits (increased biomasses of commercially targeted taxa) that may result from a reduction in Snapper biomass achieved through a directed Snapper fishery, identified that such an approach was unlikely to deliver significant benefits across fisheries. In particular, the benefits to Blue Crabs (slight positive effect) and Prawns (slight negative effect) were small or non-existent.

6 Implications

The key outputs of the project included:

- the development of the first whole of ecosystem model for GSV that describes the key components of the ecosystem and provides important ecological context for its multi-species fisheries;
- an integrated assessment of the status and impact of the key fisheries in GSV over a 20-year period, and an ecosystem-based assessment of their performance;
- an assessment of the ecological implications of the introduction of more selective fishing gear in the GSV Prawn fishery;
- an assessment of the role of Snapper predation in changing production of Prawn and Blue Crab fisheries; and
- an assessment of the potential ecological and economic benefits that may result from a reduction in Snapper biomass achieved through a directed Snapper fishery.

The model results were able to determine that the increase in GSV Snapper biomass in the late 2000s was unlikely to have contributed significantly to the observed biomass reductions in Blue Crab and Western King Prawn.

An important outcome of the study was an ecosystem based assessment of the performance of the key GSV fisheries over a 20-year period. In the framework of the existing model, the ecosystem performance measures were difficult to interpret, but suggested that the declining performance of many of the key fisheries in GSV in recent decades has been due in part to a decline and/or loss of production to the ecosystem over time. The important implication of this finding is the need to identify the source(s) and cause(s) for the potential decline in productivity, and whether or not these could be managed and mitigated. Potential sources (and causes) include environmental (e.g. temperature, salinity and nutrients changes) and anthropogenic factors (e.g. pollution, seagrass/habitat loss) that have resulted in a loss of primary productivity (e.g. reduced phytoplankton, macroalgae and/or seagrass production). The loss of production to the ecosystem may also be an artefact of model limitations, including underestimated catches. This possibility cannot be ruled out, as although the fishery data and time-series for landed catch in commercial fisheries in GSV are relatively good, those for discarded catch across most fisheries, and for landed and discarded catch in recreational fisheries are poor, and may well have been underestimated in the model. All of these factors could impact on the ecological sustainability of the GSV ecosystem and its fisheries, and could be assessed through additional model development (see Section 8).

This study is a first step towards a tool to assist with ecosystem based management of GSV and its fisheries. Further development of the model including a spatially explicit ecosystem model and consideration of other (non-fishery) sectors would be beneficial.

7 Recommendations

This project has developed the first whole of ecosystem model for GSV that describes the key components of the ecosystem, and the first ecosystem based assessment of the status, performance and impact of its key fisheries over a 20-year period. The *Ecopath with Ecosim* model developed has incorporated as much data available on the key taxa and trophic structure of groups within the GSV ecosystem, or where absent from similar systems (including Spencer Gulf) to inform key parameters including diet, biomass and production. However, given that much of the data used in the model does not have high provenance to the GSV ecosystem, and/or there is a general lack of data and knowledge about some of the key parameters for taxonomic groups in general, there is some level of uncertainty about whether the model outputs adequately reflect the real processes and trophodynamic relationships in GSV. For example, in this study we were fortunate to have comprehensive data on the diet of Snapper in the GSV ecosystem from the study of Lloyd (2010); however, for other key predators of prawns, Blue Crab and other fished species in GSV, the information is either poor or absent and has had to be inferred from other studies in other ecosystems. Furthermore, information on the diet of many of the key fished species, and food webs that underpin their populations in GSV, is extremely limited. The absence of good quantitative data with high provenance to GSV for key species and interactions ultimately means that results from the model need to be interpreted with caution and cognisant of the limitations of some of the data that inform it. It also means that there is a limitation on the types of questions that the model can adequately investigate, a critical one being (for example) the extent to which ecological processes may have limited the production of key fished species in GSV.

The GSV ecosystem model developed for this project is a combination of a static, mass-balance model (*Ecopath*) combined with temporal (times-series) model (*Ecosim*). This *Ecopath with Ecosim* model does not capture the critical spatial dimension that underpins distribution of species and their habitat, nor the spatial and temporal variability in fisheries catch and effort, and other human activities. The spatial dimension is important to improving model utility and confidence, given the marked environmental gradients in GSV that influence the distribution of taxa and fishing effort. *Ecospace* replicates *Ecosim* dynamics over a spatial grid and can include habitat, habitat preference and habitat capacity parameters, physical circulation patterns (advection model) that can capture larval transport and taxa movement dynamics, environmental data (e.g. depth, temperature, salinity), marine protected areas, and changes in the spatial distribution of fishing effort and catch (Walters *et al.* 1999). Given the significant and fine-scales of spatial management of fisheries and other activities including defence and marine parks that constrain and/or limit the influence of activities in GSV, future development of ecosystem models should include spatial and temporal dynamics. Their inclusion will improve our understanding of the GSV ecosystem, and ensure that the models for the ecosystem are relevant and have improved application.

The GSV ecosystem model developed for this project did not attempt to include environmental time series data. Improving *Ecosim* model fits to environmental time series data greatly aids in the assessment of whether changes in the production of the ecosystem and biomasses of taxonomic groups can be explained by environmental change. This is clearly relevant to the GSV ecosystem given the findings of this study infer that recent declines in the performance of some GSV fisheries may be due to a loss of ecosystem production, which could be due to environmental change. Given the marked north-south environmental gradients in GSV, including environmental time-series in a spatial context within *Ecospace* would be most relevant.

8 Further development

All of the key recommendations identified above require further development of the GSV ecosystem model to improve its application, utility and confidence as a decision support tool for ecosystem management.

Further development is needed to improve the basic information and provenance of core data that underpin confidence in the GSV ecosystem model and ensure that its outputs are robust. Information on the diet and biomass of many important trophic groups, including many commercially targeted species is poor, and represents a key data gap.

A spatially explicit trophodynamic model should be developed that includes habitat layers and ascribes trophic groups to habitats, spatially allocates fishing catch and effort, and incorporates other spatial layers in relation to the full range of activities in GSV. The spatial dimension is important to improving model utility and confidence, given the marked environmental gradients in GSV that influence the distribution of taxa and fishing effort. The addition of dynamic spatial components will provide a critical decision support tool to evaluate alternate management scenarios.

Future models need to incorporate environmental time series as these will provide an important tool for examining the potential impacts of environmental change (including climate change) on the GSV ecosystem, and the industries and activities it supports.

9 Extension and Adoption

Project findings were presented to relevant fisheries managers and other key staff at PIRSA Fisheries and Aquaculture in November 2016, and to key industry stakeholders, including Jim Raptis (GSV Prawn Fishery), Nathan Bicknell (Executive Officer, Marine Fishers Association), Dennis Holder (Blue Crab fishery) and Neil McDonald (Executive Officer, Saint Vincent Gulf Prawn Boat Owners Association). A presentation on the project was given to the Spencer Gulf and West Coast Prawn Fisherman’s Association board on 23 February 2016, to present potential applications of the GSV and Spencer Gulf (FRDC Project 2011/205) ecosystem models to assist in fisheries management and the potential applications of the model in the future.

10 Appendices

Appendix 1. Research staff and intellectual property

- List of researchers and project staff (boat skippers, technicians, consultants)

Simon Goldsworthy PI, SARDI Aquatic Sciences
Maylene Loo, SARDI Aquatic Sciences
Tony Fowler, CI SARDI Aquatic Sciences
Mike Steer, CI SARDI Aquatic Sciences
Craig Noell, CI SARDI Aquatic Sciences
Sean Sloan, CI PIRSA Fisheries and Aquaculture

- Intellectual Property

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

Appendix 2. Description of functional groups, data sources, methods and assumptions in estimating parameters used in the Gulf St Vincent Ecosystem model

Pinnipeds

Australian Sea Lion

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). The nearest and most significant ASL breeding colony to GSV, is The Pages Islands, which lie in the Backstairs Passage just east of the GSV ecosystem domain location. Trend data are available for this population between 1990 and 2009, with pup numbers per breeding season ranging from 381 to 609 (Shaughnessy *et al.* 2013). The most recent survey conducted during a breeding was in 2009 when 478 pups were counted (Goldsworthy *et al.* 2015, Shaughnessy *et al.* 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 Australian sea lions 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 GSV of B = 0.00627 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 $ER_{at\text{-}sea}$ is MJd^{-1} and M is the mean mass of each age-class/sex. The average daily energy requirement of otariid seals is a function of the proportion of time spent at sea and on-shore (Costa and Gales 2000, Winship *et al.* 2002), with daily energy requirements at-sea being about 1.8 times greater than those on-shore ($ER_{on\text{-}shore}$) (Costa and Gentry 1986). As such the *ER* of each age-class/sex was estimated following Mecenero et al. (2006) as:

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

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 (Mecenero *et al.* 2006, Winship *et al.* 2002). 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:

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

Using this approach prey consumption for the Australian Sea Lion population in the GSV areas was estimated (*Q* = 1,939 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)).

No definitive diet study of Australian sea lion in the GSV area 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); and from faecal prey DNA studies (K. Peters, unpublished data).

Long-Nosed Fur Seal

Biomass and consumption: Long-Nosed Fur Seal (*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 along the south coast of Kangaroo Island and south of Spencer Gulf at North and South Neptune and Liguanea Islands (Shaughnessy *et al.* 2015). Estimates for the abundance of long-nosed fur seals in the GSV ecosystem model area were based on pup production estimates obtained from these locations between the 1993/94 and 2013/14 breeding seasons (Shaughnessy *et al.* 2015).

Adult female fur seals were not considered to forage within the GSV model area based on satellite tracking studies, 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 and off the north coast of Kangaroo Island indicate that most of 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, south of Tasmania and into Bass Strait (B. Page and S. Goldsworthy unpublished data). For the purposes of the GSV Ecosystem 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 $S = 0.627 - 0.073a + 0.003a^2 - (5.91 \times 10^{-5})a^3$; males $S = 0.627 - 0.097a + 0.006a^2 - (0.140 \times 10^{-3})a^3$, where *S* is survival and *a* is age in years. Maximum ages were 23.4 and 16.7 for females and males, respectively (McKenzie 2006, McKenzie *et al.* 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 sub-adult 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 Long-Nosed Fur Seals within GSV ecosystem area was estimated (*Q* = 1,044 t yr^{-1}); with *Q/B* = 46.707 and *P/B* = 1.184. Production per Biomass estimates (*P/B*) was estimated for the entire population as: ((current biomass live + dead)/(previous year annual biomass alive)).

Diet data for the long-nosed fur seal were based on 333 scat samples collected from five sites containing juveniles and sub-adult males (B. Page, unpublished data) within Spencer Gulf

(Donnington Rock, Sibsey, Althorpe, Thistle and Liguanea Islands), assuming these are similar to prey species targeted in the GSV ecosystem area. 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 long-nosed fur seals, it is unlikely that all of the foraging undertaken by animals is restricted to the GSV ecosystem model area, and it is also possible that some animals originating from other key population centres outside GSV ecosystem model domain also spend part of their time foraging within the region. We have estimated the amount of dietary input from regions outside of the GSV ecosystem model area as 40%.

Cetaceans

Within GSV, 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*), the latter of which are predominantly distributed in shelf and oceanic waters further offshore than former (Gibbs *et al.* 2011, Kemper and Ling 1991, Kemper 2004). More recently, a third species (the Southern Australian bottlenose dolphin) has been described from coastal regions of Victoria, South Australia and Tasmania (Charlton-Robb *et al.* 2011, Moller *et al.* 2008). There may also be hybridisation between species (Kemper 2004). Marked population structure has been found between coastal *Tursiops* spp. 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 GSV ecosystem model area. Using their strata 3 (Upper GSV) and strata 4 (Lower GSV and Investigator Strait) surveys areas, the mean estimate was 3,462 dolphins (1,712 – 7,239 ±95% confidence limits, uncorrected for availability bias) giving a mean density of 0.2799 dolphins km^{-2}. The mean mass of *Delphinus* was estimated to be 79 kg (C. Kemper pers. comm.) giving overall estimates of biomass of 273 t or B = 0.0221 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 weight) and average daily metabolic requirements ($ADMR$ in kJ d^{-1}) as follows:

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

where:

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

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 β = 2.5 (Hooker *et al.* 2002, Kenney *et al.* 1997, 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 yearly ration (365 x *R*) and abundance (Barlow et al. 2008). Following this, we estimated the annual prey consumption of common dolphins to be 5,625 t yr^{-1}. This provides a *Q/B* estimate of 20.578. *P/B* was estimated at 0.09 for Common Dolphins based on Barlow and Boveng (1991).

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. In Spencer Gulf they are known to feed on bycatch and discards from the Prawn fishery (Svane 2005). We have assumed some feeding also occurs in association with the GSV Prawn fishery, although given the fishery is much smaller, the contribution to the diet from discards is likely to be considerably less.

Bottlenose Dolphin

Biomass and consumption: Bottlenose Dolphin abundance was based on the mean density of dolphins (0.0325 km^{-2}) from 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).

The diets of bottlenose dolphins were assessed from stomach contents and stable isotope analyses of individuals that were found dead in southern Australia (Gibbs *et al.* 2011, Kemper and Gibbs 2001), 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 GSV ecosystem model area. Estimates used here were based on available summaries and estimates (Copley 1996, Wiebkin 2011a). 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 2,348 breeding pairs based on surveys of Troughbridge and Kangaroo Island (Wiebkin 2011a), the total population of little penguins in the GSV ecosystem model area is estimated to be 6,476. Assuming a mean mass of 1.2 kg per bird, the total biomass in the habitat area of the population is estimated to be 7.8 t (B = 0.00148 t km^{-2}), assuming 50% of the GSV domain area is suitable habitat for Little Penguins. 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 GSV model area of 665.5 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).

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

Petrels



Biomass and consumption: The dominant petrel species in the GSV ecosystem region are the abundant short-tailed shearwater (*Puffinus tenuirostris*) which breeds across many of the islands in lower Spencer Gulf and off the southern and western Eyre Peninsula (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 are only available for the short-tailed shearwater, but only for areas south of Spencer Gulf outside of the GSV ecosystem. 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 (Hunter *et al.* 2000, Wooller *et al.* 1990). 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 and adjacent to the GSV ecosystem was 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 waters adjacent to the GSV ecosystem, 55 days incubating the egg (incubation shared equally between the sexes) and 90 days rearing chicks (Einoder 2010, Einoder and Goldsworthy 2005, Weimerskirch and Cherel 1998). 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 GSV ecosystem). *Q/B* is estimated to be 147.3. *P/B* was estimated to be 1.0 and 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 region adjacent to the GSV ecosystem 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). 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 (0.00514 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 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 based on their proportion of prey biomass consumed in the habitat area. Import of prey consumption from outside the GSV ecosystem 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 GSV 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 10% of the Margaret Brock Reef population and about 2% of the Victoria and Tasmania populations may be foraging within the GSV ecosystem area at any time. With individual gannets weighing approximately 2.5 kg (Daunt *et al.* 2008), GSV gannet biomass is estimated to be 0.9 t (0.00003 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 125.3, and a *P/B* estimate of 1.0 was used based on Sakshaug (1997). 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 GSV ecosystem, 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 GSV (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 (0.0000265 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 90.2. A *P/B* estimate of 1.0 was used based on Sakshaug (1997). 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 GSV ecosystem (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 GSV ecosystem region was based on data presented in Copley (1996), (20,000 black-faced shags; 21,216 pied cormorants). Assuming a mean mass of 1.6 kg (Riordan and Johnston 2013), the estimated biomass is 58.0 t (B = 0.00051 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 4,119.2 t, and a *Q/B* estimate of 77.4. A *P/B* estimate of 1.0 was used based on Sakshaug (1997).

There are no published data on the diets of black-faced shags or pied cormorants from South Australia or the GSV region. Information was taken instead from limited data available for Black-Faced Shags (samples from SA and Victoria, Marchant and Higgins 1990), and Pied Cormorants (samples from WA and Queensland, Blaber and Wassenberg 1989, Humphries *et al.* 1992). The proportions of prey taxa were weighted for each species by their estimated biomass.

Gulls


Biomass and consumption: There are two species of gull that occur in the GSV ecosystem 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). Earlier estimates were used to derive estimates for 1993 of 35,840 pairs of silver gulls, and 7 pairs of Pacific gulls in the GSV 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 21.5 t (or 0.00205 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 2,760 t yr^{-1}. Based on these estimates, *Q/B* is 130.1. A *P/B* estimate of 1.0 was used based on Sakshaug (1997).

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 GSV region. Proportion of prey taxa was weighted for each species by estimated biomass.

Pelagic sharks

The pelagic shark community of the GSV region is made up of five main species, the White Shark (*Carcharodon carcharias*), Smooth Hammerhead (*Sphyrna zygaena*), Common Thresher (*Alopias vulpinus*) and Bronze (*Carcharhinus brachyurus*) and Dusky Whaler (*C. obscurus*). There is scant catch data for pelagic sharks in the GSV 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 whaler sharks and smooth hammerhead, mainly caught 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):

$$M = K^{0.65} L_{\infty}^{-0.279} T^{0.463}$$

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

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

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

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 broader SA region outside of the GSV ecosystem, 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 total length, occur in areas where Snapper are abundant. Smaller 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, Dewar *et al.* 2004, Klimley *et al.* 2002, Weng *et al.* 2005). *Q/B* was estimated using Christensen and Pauly's (1992b) 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.

The diet of white shark was based primary on the study from South Africa by Hussey et al. (2012), using their largest size class (> 2.85 m) as most sharks in SA watres 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 know elasmobranch taxa.

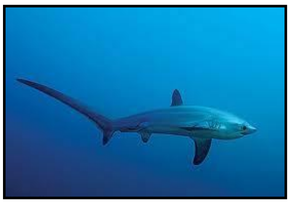
Smooth hammerhead

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 GSV 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 (1992b) 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.

Dietary information was taken from the study of Rogers et al. (2012). A total of 39 stomachs (95% containing 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 (1992b) 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.

Dietary information was taken from the study of Rogers et al. (2012). A total of 27 stomachs (63% containing 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 GSV 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 (1992b) equation with $W\infty$ 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.

Dietary information was taken from the study of Rogers et al. (2012). A total of 250 bronze whaler stomachs (65% containing prey items) and 49 dusky shark stomachs (65% containing prey items) were examined from samples collected in the Spencer Gulf, GSV and GAB, between 2007 and 2010.

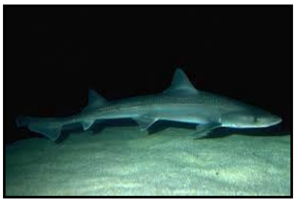
Demersal sharks

Demersal sharks were represented by three model groups within the GSV 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 a common bycatch in many commercial and recreational fisheries in southern Australia. Biomass estimates were based on standardised surveys undertaken in the GSV Prawn fishery, using the upper standard error estimates (0.02529 t km^{-2}). This was calculated following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). 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). 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 and school shark



Gummy (*Mustelus antarcticus*) and School Shark (*Galeorhinus galeus*) biomass estimates were based on mean annual catch of both species in GSV between 1993 and 2013 and an estimated fishing mortality (*F*) during this period of 0.3 giving a biomass of 0.07472 t km^{-2} for gummy shark while *F* for school shark was 0.7, giving a biomass of 0.00071 t km^{-2}. Estimates of *P/B* (0.55) and *Q/B* (2.6) were based on estimates of *F* (0.3) and *M (0.25)* (Froese and Pauly 2009) for gummy shark while school shark estimates of *P/B* is 8855 and *Q/B* is 2.5 (*F* = 0.7 and *M* = 0.25) (Froese and Pauly 2009). 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 several species including the Rusty Catshark (*Parascyllium ferrugineum*), Elephant Fish (*Callorhinchus milii*), Angel Shark (*Squatina australis*), Cobbler Carpet Shark (*Sutorectus tentaculatus*), Piked Dogfish (*Squalus megalops*), Southern Sawshark (*Pristiophorus nudipinnis*) and Common Sawshark (*Pristiophorus cirratus*).

Biomass estimates were based on standardised surveys undertaken in the GSV Prawn fishery, using the upper standard error estimates for 3 species, Rusty Catshark, Elephant Fish and Angel Shark (0.03622 t km^{-2}). This was calculated following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates of 0.351 and 2.6 were based on Froese and Pauly (2009) and Currie (2009b) respectively. Diet 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






Biomass estimates were based on standardised surveys undertaken in the GSV Prawn fishery, using the upper standard error estimate for eight species (0.16777 t km^{-2}) and the calculation followed the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). The species included Melbourne Skate (*Spiniraja whitleyi*), Southern Fiddler Ray (*Trygonorrhina dumerilii*), Southern Shovelnose Ray (*Aptychotrema vincentiana*), Smooth Stingray (*Dasyatis brevicaudata*), Coastal Stingaree (*Urolophus orarius*), Sparsely-Spotted Stingaree (*Urolophus paucimaculatus*), Western Shovelnose stingaree (*Trygonoptera mucosa*), White Spotted Skate (*Dipturus cerva*). *P/B* and *Q/B* estimates of 0.234 and 1.757 were based on Froese and Pauly (2009) and Currie (2009b), respectively.

Dietary data were based on the analysis 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 Dasyatidaes 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 (Gunn and Young 1999, Young *et al.* 1996). Most (99.6%) of the Australian SBT catch is taken in the eastern GAB region (Wilson *et al.* 2009). The estimated spawning stock biomass has declined over the past three decades and there is no sign that the spawning stock is rebuilding, hence it is listed as Critically Endangered in the Red List (Collette *et al.* 2011). 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

No biomass data were available for Yellowtail Kingfish (*Seriola lalandi*) and it 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*) is an abundant, inshore, demersal fish species that occurs throughout temperate and sub-tropical waters of the Indo-Pacific region (Kailola *et al.* 1993, Paulin 1990). 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 models developed to facilitate management of this fishery (Fowler *et al.* 2013). Biomass was estimated to be 0.08698 t km^{-2}. *P/B* and *Q/B* estimates were 0.493 and 3.8, respectively (Froese and Pauly 2009, Fulton and Smith 2004). Diet data were sourced from 735 stomach samples obtained across multiple seasons within the GSV region (Lloyd 2010).

Snook

No Snook (*Sphyraena novaehollandiae*) were caught in the bycatch surveys undertaken in the GSV fishery, hence the biomass estimates were based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, using the upper standard error estimate (0.00357 t km^{-2}) (Currie *et al.* 2009a). *P/B* and *Q/B* estimates were 0.411 and 3.51, respectively (Currie and Sorokin 2010, Fulton and Smith 2004). Diet data were sourced from Caines (2005) and Page et al. (2011) based on 181 stomach samples.

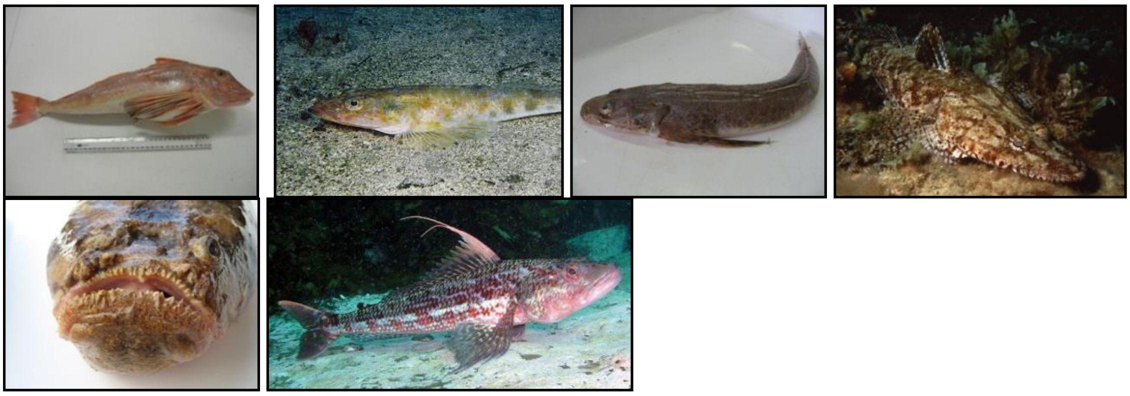
Barracouta

Biomass estimates of Barracouta (*Thyrsites atun*) were based on standardised surveys undertaken in the GSV Prawn fishery, using upper standard error estimate (0.00830 t.km^{-2}) and the calculation followed the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 0.411 and 3.64, respectively (Currie and Sorokin 2010, Fulton and Smith 2004). Diet data were sourced from Caines (2005) and Page et al. (2011) based on 71 stomach samples.

Skipjack Trevally

Skipjack Trevally (*Pseudocaranx wrighti*) was the most abundant fish caught in the bycatch surveys for the Gulf St Vincent Prawn fishery. The biomass estimates based on these standardised surveys using the upper SE estimate was 0.21971 t km^{-2}, using the calculation methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 0.480 and 4.17, respectively (Currie and Sorokin 2010, Fulton and Smith 2004). Diet data were sourced from Currie and Sorokin (2010) based on 42 stomach samples collected from Spencer Gulf.

Medium demersal piscivores

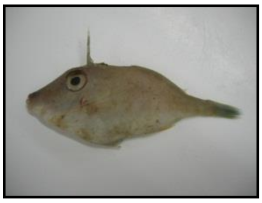


The medium demersal piscivore group consists of six main species that primarily consumed fish. This piscivorous fish group included Small Tooth Flounder (*Pseudorhombus jenynsii*), Common Stargazer (*Kathetostoma laeve*), Tiger Flathead (*Platycephalus richardsoni*), Red Cod (*Pseudophycis bachus*), Tasselled Anglerfish (*Rhycherus filamentosus*) and an unknown Flathead species (*Platycephalus* sp.). These species were grouped due to dietary similarities identified by Currie and Sorokin (2010). Biomass estimates were based on standardised surveys undertaken in the Gulf St Vincent Prawn fishery, summing the upper standard error estimate for all species (0.06785 t km^{-2}) using the calculation methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 0.636 and 1.58 (mean of eight species from Spencer Gulf), respectively (Currie and Sorokin 2010, Fulton and Smith 2004). 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.

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). However, these species were not sampled in the GSV Prawn Fishery surveys, but were included as a functional group in the model as the species do occur in GSV. Biomass estimates were therefore based on standardised surveys undertaken in the Spencer Gulf Prawn fishery, summing the upper SE estimate for each species (0.00703 t km^{-2}) (Currie *et al.* 2009b, Currie and Sorokin 2010). *P/B* and *Q/B* estimates were 0.625 and 2.34, respectively (Currie and Sorokin 2010, Fulton and Smith 2004). Diet data were sourced from Currie and Sorokin (2010) based on only 3 stomach samples (blue morwong =1, short boarfish = 2), weighted for each species in proportion to their estimated biomass.

Degens/Rough Leatherjackets




Degens Leatherjacket (*Thamnaconus degeni*) was the next most abundant species after Skipjack Trevally collected during the trawl surveys in the GSV Prawn Fishery. The Rough Leatherjacket (*Scobinichthys granulatus*) in this group was not as abundant. 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 Gulf St Vincent Prawn fishery, summing the upper standard error estimate for each species (0.12785 t km^{-2}), calculated using the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 0.900 and 2.26, respectively (Currie and Sorokin 2010, Fulton and Smith 2004). Diet data were sourced from Currie and Sorokin (2010) based on the analyses of 62 stomach samples (Degens leatherjackets = 27, rough leatherjacket = 35), weighted for each species in proportion to their estimated biomass.

King George Whiting

King George Whiting (*Sillaginodes punctatus*) are an important commercially and recreationally caught species in South Australia. Key estimates of biomass, exploitation rate and recruitment are available for this species as part of adynamic, spatial age-length structure model (WhiteEst), developed to facilitate management of this fishery (Fowler *et al.* 2014, McGarvey and Fowler. 2002). Legal-size population biomass was used as a starting estimate overall biomass for the GSV region as 0.04568 t km^{-2}. *P/B* and *Q/B* estimates were 0.548 and 2.29, respectively (Currie and Sorokin 2010, Fulton and Smith 2004). Diet data are limited for this species, and were sourced from Currie and Sorokin (2010) based on the analysis of 19 stomach samples from Spencer Gulf.

Southern Sea Garfish

Southern Sea Garfish (*Hyporhamphus melanochir*) are an important commercially and recreationally caught species in South Australia. 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 *et al.* 2007, Steer *et al.* 2012). 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 as a starting estimate overall biomass for the GSV region as 0.03164 t km^{-2}. *P/B* and *Q/B* estimates were 0.329 and 4.73, respectively (Currie and Sorokin 2010, Fulton and Smith 2004).

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

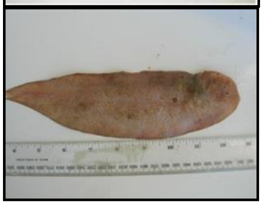
Red Mullet

Red Mullet (*Upeneichthys vlamingii*) was among the more dominant fish sampled during standardised trawl surveys in the Gulf St Vincent Prawn fishery. Biomass estimates were calculated following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015), using the upper standard error estimate of 0.02008 t km^{-2}. *P/B* and *Q/B* estimates were 0.790 and 4.4, respectively (Currie and Sorokin 2010, Froese and Pauly 2009). Diet data were sourced from Currie and Sorokin (2010) based on the analysis of 55 stomach samples collected from Spencer Gulf.

Medium crustacean teleosts

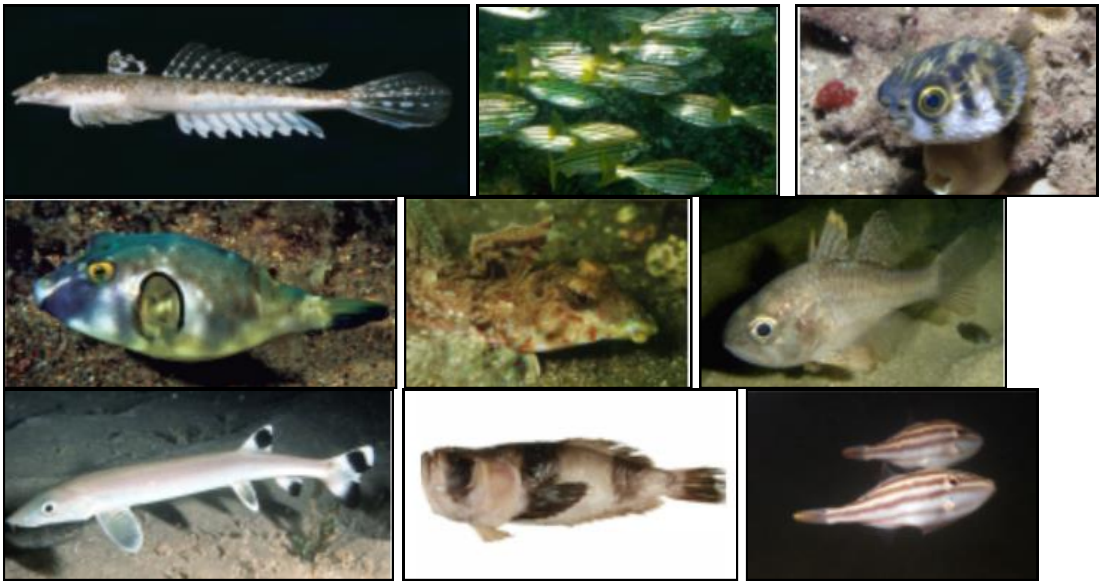






Medium crustacean teleosts are composed of a group of medium sized fishes whose diet is mostly composed of crustaceans, and include Southern Tongue Sole (*Cynoglossus broadhursti*), Gurnard Perch (*Neosebastes pandus*) and Blue Warehou (*Seriolella brama*). Biomass estimates were based on standardised surveys undertaken in the GSV Prawn fishery, summing the upper SE estimate for each species (0.00849 t km^{-2}), calculated using the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 0.546 and 2.97, respectively (Currie and Sorokin 2010, Froese and Pauly 2009). Diet data were sourced from Currie and Sorokin (2010) based on the analysis of 44 stomach collected from Spencer Gulf, weighted for each species in proportion to their estimated biomass.

Medium molluscan teleosts



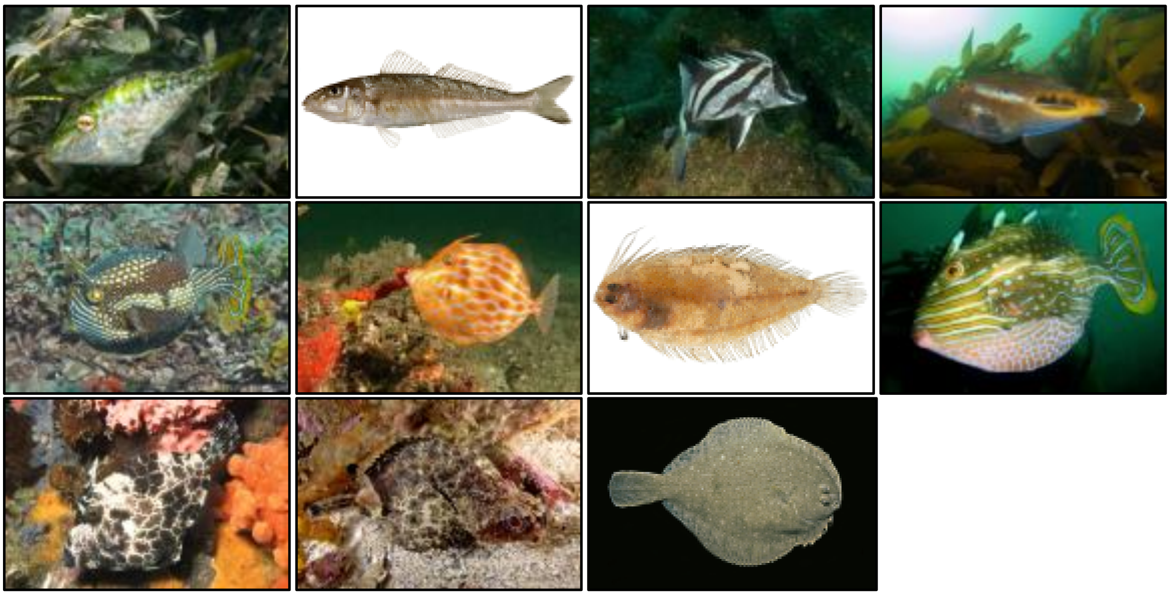
Medium molluscan teleosts are composed of a group of medium sized fishes whose diet was mostly composed of molluscs, and included Spotted Stinkfish (*Repomucenus calcaratus*), Striped Perch (*Pelates* octolineatus), Spikey Globefish (*Diodon nicthemerus*), Ringed Toadfish (*Omegophora* armilla), Common Stink Fish (*Foetorepus calauropomus*), Southern Gobbleguts (*Vincentia conspersa*), Beaked Salmon (*Gonorynchus greyi*), Fringed Stargazer (*Ichthyscopus barbatus*) and Chinaman Leather Jacket (*Nelusetta ayraud*). Biomass estimates were based on standardised surveys undertaken in the GSV Prawn fishery, summing the upper standard error estimate for each species (0.07149 t km^{-2}) following the calculation methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 0.869 and 2.26, respectively (Currie and Sorokin 2010, Froese and Pauly 2009). Diet data were sourced from Currie and Sorokin (2010) based on the analysis of 54 stomach samples collected from Spencer Gulf and weighted for each species in proportion to their estimated biomass.

Small crustacean teleosts

Small crustacean teleosts are composed of several species of small sized fish (< 30 cm) whose diet was mostly composed of crustaceans (Currie and Sorokin 2010). This group includes Little Scorpionfish (*Maxillicosta scabriceps*), Slender Bullseye (*Parapriacanthus elongatus*), Spiny Gurnard

(*Lepidotrigla papilio*), Spotted Grubfish (*Parapercis ramsayi*), Wavy Grubfish (*Parapercis haackei*), Gulf Gurnard Perch (*Neosebastes bougainvillii*), Soldier Fish (*Gymnapistes marmoratus*), Southern Pygmy Leatherjacket (*Brachaluteres jacksonianus*), Many Banded Sole (*Zebrias scalaris*), Sculptured Seamoth (*Pegasus lancifer*) and Lachet (*Pterygotrigla polyommata*). Biomass estimates were based on standardised surveys undertaken in the GSV Prawn fishery, summing the upper standard error estimate for each species (0.06808 t km^{-2}) after calculation following the calculation methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). P/B and Q/B estimates were 1.315 and 3.32, respectively((Currie and Sorokin 2010, Froese and Pauly 2009). Diet data were sourced from Currie and Sorokin (2010) based on the analysis of 230 stomach samples from the Spencer Gulf; with prey biomass weighted for each species in proportion to their estimated biomass.

Small annelid teleosts



Small annelid teleosts comprised several species of small sized fish (<30 cm) whose diet is mostly composed of polychaetes (Currie and Sorokin 2010). This group included bridled leatherjacket (*Acanthaluteres spilomelanurus*), silver whiting (*Sillago bassensis*), longsnout boarfish (*Pentaceropsis recurvirostris*), toothbrush leatherjacket (*Acanthaluteres vittiger*), ornate cowfish (*Aracana ornate*), mosaic leatherjacket (*Eubalichthys mosaicus*), crested flounder (*Lophonectes gallus*), shaws cowfish (*Aracana aurita*), Gunn's leatherjacket (*Eubalichthys gunnii*), goblin fish (*Glyptauchen panduratus*), longsnout flounder (*Ammotretis rostratus*). Biomass estimates were based on standardised surveys undertaken in the Gulf St Vincent Prawn fishery, summing the upper standard error estimate for each species (0.02571 t km^{-2}) after calculation following the calculation methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 0.992 and 2.82, respectively (Currie and Sorokin 2010, Froese and Pauly 2009). Diet data were sourced from Currie and Sorokin (2010) based on the analyses of 133 stomach samples collected in Spencer Gulf; with prey biomass weighted for each species in proportion to their estimated biomass.

Syngnathids

Only one syngnathid species (*Hippocampus abdominalis*) was sampled in the trawl surveys undertaken in the Gulf St Vincent Prawn fishery. The biomass was calculated following the calculation methods of Currie *et al.* (2009a) and Burnell *et al.* (2015), but is noted to be an underestimate. *P/B* and *Q/B* estimates were 1.000 and 4.70, respectively (Currie and Sorokin 2010, Froese and Pauly 2009). Diet data were sourced from Currie and Sorokin (2010), based on the analysis of 10 stomach samples from several species. Most prey species were small crustaceans and the prey biomass were 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. No biomass data were available, and this parameter was estimated by the model. *P/B* and *Q/B* estimates were 0.45 and 4.7, respectively (Currie and Sorokin 2010, Froese and Pauly 2009). Diet data for *A. truttaceus* were sourced from Caines (2005) and Page et al. (2011). Historically, the commercial catch has mostly been extracted 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 1994 and 2013.

Australian Herring

Australian Herring or Tommy Rough (*Arripis georgianus*) are predominantly found in the Gulfs, inshore areas in shelf waters and around offshore islands. Historically, the commercial catch has mostly been extracted from State waters using purse seine nets, gill-nets, haul-nets and hand-lines, and is used for bait and human consumption. No biomass data were available, and this parameter was estimated by the model. *P/B* and *Q/B* estimates were 1.64 and 6.32, respectively (Currie and Sorokin 2010, Froese and Pauly 2009). 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.).

Small pelagic fish

Jack and Yellowtail Mackerel

Jack (*Trachurus declivis*) and Yellowtail (*Trachurus novaezelandiae*) Mackerel are common small pelagic fish in the GSV region. Biomass estimates were based on standardised surveys undertaken in the Gulf St Vincent Prawn fishery, summing the upper SE estimate for each species (0.07560 t km^{-2}) after calculations following the calculation methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 0.52 and 5.37, respectively (Currie and Sorokin 2010, Froese and Pauly 2009). 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 australasi*cus) are a common small pelagic fish in the GSV region. Biomass of the species within the GSV region was estimated by the model. P/B and Q/B estimates were 0.52 and 5.37, respectively (Currie and Sorokin 2010, Froese and Pauly 2009). 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 found in the Gulfs. The Australian sardine was the largest (by volume) single species caught in the South Australian wild-catch sector in 2012-13, constituting ~79% of total catch and 11% of total value (ABARES 2014). The South Australian Sardine Fishery predominantly takes sardine (*Sardinops sagax*), but other small pelagics are also captured including anchovy (*Engraulis australis*), jack and yellowtail 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 using the Daily Egg Production Method (DEPM) on an annual or bi-annual basis (Ward *et al.* 2015). The average spawning stock biomass from 2000 to 2014 was used as an estimate for 1994 (1.79589 t km^{-2}). *P/B* and *Q/B* estimates were 1.6 and 5.04, respectively (Currie and Sorokin 2010, Froese and Pauly 2009). Diet of sardines was based on the analysis 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 Gulfs. The DEPM based estimates of anchovy spawning biomass in the 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 (Currie and Sorokin 2010, Froese and Pauly 2009). Diet information was based on the analysis 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*). No biomass data were available; hence this parameter was estimated by the model. *P/B* and *Q/B* estimates were 1.80 and 5.76, respectively (Currie and Sorokin 2010, Froese and Pauly 2009). 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).

Cephalopods

The cephalopod group consists of Southern Calarmari, Giant Australian Cuttlefish, ‘other squids’, and ‘octopuses’. All of these groups are commercially harvested. Calarmari (*Sepioteuthis australis*) and Giant Australian Cuttlefish (*Sepia apama*) are targeted in the MSF fishery. *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 Calarmar

Calarmari (*Sepioteuthis australis*) are a common and commercially harvested cephalopod in the Gulfs. Biomass estimates were based on standardised surveys undertaken in the Gulf St Vincent Prawn fishery using the upper SE estimate (0.03613 t.km^{-2}). Calculations followed the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 1.83 and 18.25, respectively (Officer & Parry 1996 in Fulton and Smith 2004). Diet of Calarmari was based on 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.* 2009b, Currie and Sorokin 2010) as none were caught in the surveys carried out in GSV. *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 of all other remaining squid taxa found in the GSV region. These included the Nova Cuttlefish (*Sepia novaehollandiae*), Southern Bottletail Squid (*Sepiadarium austrinum*), Striped Pyjama Squid (*Sepioloidea lineolata*) and Braggi's Cuttlefish (*Sepia braggi*). Biomass estimates for these species were based on standardised surveys undertaken in the Gulf St Vincent Prawn fishery, with biomass data for all species summed (0.01357 t km^{-2}) following the calculation methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *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 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 several species including southern keeled octopus (*Octopus berrima*), southern sand octopus (*Octopus kaurna*) and pale octopus (*Octopus pallidus*). Biomass estimates for these species were based on standardised surveys undertaken in the Gulf St Vincent Prawn Fishery, summing the upper SE estimate for each species (0.01615 t km^{-2}) after calculations following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *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 GSV as part of the Northern Zone management region for the fishery. Biomass for 1994 was estimated from the catch within the GSV (92.6 t) and assuming fishing mortality of 0.3, giving a biomass estimate of 0.05881 t km^{-2} with a 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. A biomass estimate of 0.08225 t km^{-2} for 1994 was obtained from a recently completed project that developed a bio-economic model for South Australia's prawn trawl fisheries (Noell *et al.* 2015). *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 (*Pelagicus armatus*) are distributed very widely from tropical coastal waters of the western Indian Ocean to the eastern Pacific Ocean. In relatively colder waters of temperate Australia, the Blue Swimmer Crab has adapted by increasing growth and reproduction during the warmer months when temperatures are similar to the tropical regions. Biomass estimates were based on standardised surveys undertaken in the Gulf St Vincent Prawn fishery using the upper SE estimate (0.21620 t km^{-2}) after calculations following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 2.80 and 8.50, respectively (Loneragan *et al.* 2010). Blue swimmer crabs have a predatory/scavenging lifestyle, feeding mainly on molluscs, crustaceans and polychaetes. However, diet of this species has not been studied in South Australia, therefore the diet data was based on the study by Edgar (1990) and descriptions in Bryars and Svane (2008).

Sand Crabs

Biomass estimates were based on standardised surveys undertaken in the Gulf St Vincent Prawn fishery using the upper SE estimate, giving a value of 0.00055 t km^{-2}, after calculations following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 2.80 and 8.50, respectively (Loneragan *et al.* 2010). Little is known of the diet of Sand Crabs (*Ovalipes australiensis*). 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).

Large crabs and bugs

The large crab and bugs group comprises at least ten species from seven families (Diogenidae, Dromiidae, Majidae, Pilumnidae, Portunidae, Scyllaridae and Xanthidae) and include (from most to least biomass contribution) the Balmain Bug (*Ibacus peroneii*), Giant Spider Crab (*Leptomithraz gaimardii*), Common Hermit Crab (*Paguristes frontalis*), Bristled Sponge Crab (*Austrodromidia octodentata*), Facetted Crab (*Actaea calculosa*), Shaggy Sponge Crab (*Lamarckdromia globosa*), Hairy Shore Crab (*Pilumnus* sp.), Rock Crab (*Nectocarcinus integrifrons*) and Southern Sponge Crab (*Austrodromidia australis*). Biomass estimates for these species were based on standardised surveys undertaken in the Gulf St Vincent Prawn fishery, with the upper SE biomass data for all species combined (0.01615 t km^{-2}), after calculations following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *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, strawberry prawns (Pandalidae) and shrimps. Biomass estimates for these crustaceans were based on standardised surveys undertaken in the GSV Prawn fishery, with the upper SE biomass data for all species combined (0.00864 t km^{-2}), after calculations following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015) *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 several species of echinoderms (seastars, sea urchins and holothurians) and molluscs (chitons, sea slugs, gastropods and nudibranchs). The biomass estimate for this group was based on standardised surveys undertaken in the GSV Prawn fishery, by summing the upper standard error biomass of all species (0.04353 t km^{-2}), after calculations following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 2.80 and 14.00 respectively (Loneragan *et al.* 2010). Diet was based on the reef associated herbivore diet detailed in Loneragan et al. (2010), from the Jurien Bay *Ecopath* model.

Sand zoobenthos feeders

Sand zoobenthos feeders comprised a broad group of bivalves including cockles (*Acrosterigma cygnorum*), commercial scallops (*Pecten fumatus*), doughboy scallop (*Mimachlamys asperrima*), queen scallop (*Equichlamys bifrons*), grooved cardita (*Cardita incrassiata*), lima lima (*Lima vulgaris*) razor fish (*Atrina (Servatrina) tasmanica*), rock shell (*Cleidothaerus albidus*), mud oyster (*Ostrea angasi*) and southern hammer oyster (*Malleus meridianus*). Biomass estimate for this group was based on standardised surveys undertaken in the Gulf St Vincent Prawn fishery, by summing the upper standard error biomass of all species (0.01135 t km^{-2}), after calculations following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015) *P/B and Q/B* estimates were 0.65 and 7.50 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 GSV 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. The fishery in GSV falls into the Central Zone management region for the fishery.

Greenlip abalone
No biomass estimate was available for this group. *P/B*, *Q/B* and *EE* estimates were 0.73, 12.41 and 0.90, respectively (Fulton and Smith 2004). 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). 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

Biomass estimates for the small mobile crustacean (deposit detritovore feeders, DDF), which included small crustaceans such as snapping shrimps (*Alpheus villosus*), pistol shrimp (*Alpheus lottini*), mantis shrimp (*Erugosquilla graham*) and long-writes shrimp (*Processa gracilis*), were based on standardised surveys undertaken in the GSV Prawn fishery. The upper standard error biomass of all species were combined (0.00235 t km^{-2}), after calculations following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B and Q/B* estimates were 7.01 and 27.14 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 comprise deposit detritovore feeding polychaetes which can include trumpet worms (Pectinariidae), bobbit worms (Eunicidae), sea mice (Aphroditidae), rag worms (Nereidae), spaghetti worms (Terebellidae), Capitellidae, Cirratulidae, peanut worms (Sipuncula), spoon worms (Echiuroidea), Oligochaeta, horse-shoe worms (Phoronid) and acorn worms (Hemichordata). Biomass estimate was based on standardised surveys undertaken in the GSV Prawn fishery, using the combined upper SE biomass of all species (0.00240 t km^{-2}), after calculations following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 1.60 and 6.00 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 various species of ascidians (*Pyura gibbosa, Ascidia sydneiensis, Herdmania momus, Polycarpa pedunculata, Trididemnum cerebriforme*), bryozoans (*Celleporaria fusca, Adeona grisea, Steginoporella chartacea, Triphyllozoon moniliferum*), Goose Barnacle (Pedunculata), hydroids, molluscs such as Hairy Mussel (*Trichomya hirsute*), octocorals (*Carijoa multiflora*), Sea Pen (*Sarcoptilus grandis*) and Porifera. The biomass estimate was based on standardised surveys undertaken in the GSV Prawn Fishery, using the combined upper SE biomass of all species (0.95748 t km^{-2}), after calculations following the methods of Currie *et al.* (2009a) and Burnell *et al.* (2015). *P/B* and *Q/B* estimates were 2.80 and 11.80 respectively (Fulton and Smith 2004). Diet was based on Mackinson and Daskalov 2008).

Gelatinous zooplankton

Gelatinous zooplankton consisted of all the jellies, salps and ctenophores. Very little is known about the taxa within these groups. The biomass estimate was based on the SGE *Ecopath* model (Gillanders *et al.* 2015) while estimates of *P/B* (16.50) and *Q/B* (80.00) were 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. A biomass estimate of 0.63158 t km^{-2} was based on values reported for GSV in (2008, 2012). 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 values reported for GSV (0.16842 t km^{-2}) in van Ruth (2008, 2012). 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 retained on a 45 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 also based on Mackinson and Daskalov (2008).

Primary Producers

Phytoplankton
Estimates of phytoplankton biomass and primary productivity (*P/B*) in GSV were based on values reported in van Ruth (2008, 2012). Phytoplankton biomass was estimated to be 7.2 t km^{-2}, while primary productivity (*P/B*) was estimated to be 114.2.

Macroalgae and Seagrass
Macroalgae biomass in GSV (1675.3 t km^{-2}) was calculated using the dry weight estimated for the SGE *Ecopath* model and converted to wet weight using macroalgae moisture content of 77% (SARDI, unpublished data) and the habitat area estimate from Department for Environment and Heritage (2008).

Biomass estimate for seagrass in GSV (3306.3 t km^{-2}) was calculated using the dry weights reported in Nayar (2012) and converted to wet weight using seagrass moisture content of 67% (Nayar *et al.* 2009) and the habitat area reported in Department for Environment and Heritage (2008). *P/B* was then estimated from this biomass and the production value from the SGE *Ecopath* model (Gillanders *et al.* 2015).

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 GSV was based on that developed for the North Sea by Mackinson & Daskalov (2008). Essentially, organic matter produced from phytoplankton is aportioned 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 GSV 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; 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

As there were no data for biomass of benthic microflora in GSV, the estimate was based on the value estimated for the SGE *Ecopath* model, 0.5 t km^{-2} (Gillanders *et al.* 2015). Estimates of *P/B* and *Q/B* were 29,200 and 18,940, respectively, from Mackinson & Daskalov (2008). Diet was also 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), similarly for the diet data.

Detritus

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

Detritus – DOM in water column

As no data were available for the biomass of DOM in the water column for GSV, biomass was based on the value estimated for the SGE *Ecopath* model, 20.4 t km^{-2} (Gillanders *et al.* 2015).

Detritus – POM in sediment

The biomass of POM in the sediment was based on the calculation used for the SGE *Ecopath* model (Gillanders *et al.* 2015). Lauer et al. (2007) showed that ~2.7% of sediment organic carbon was microphytobenthos. Assuming the remaining fraction is benthic detritus (benthic POM), then POM biomass is 18.5 t km^{-2}.

Discards

Discards were calculated as the total fishery discards 0.10490 t km^{-2}.

Appendix 3. Diet matrix Gulf St Vincent Ecosystem model

Group name	Group	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
Australian Sea Lion	1											0.1267					
Long-Nosed Fur Seal	2											0.1267					
Bottlenose Dolphin	3											0.1569	0.0001				
Common Dolphin	4											0.0169	0.0001				
Petrel	5		0.0030								0.0100						
Australian Gannet	6																
Little Penguin	7	0.0010	0.0020										0.0064				
Shags & Cormorants	8																
Terns	9										0.0001						
Gulls	10																
White Shark	11																
Whaler Shark	12											0.0735					
Smooth Hammerhead	13											0.1152					
Common Thresher Shark	14											0.0735					
Gummy Shark	15											0.0735					
School Shark	16											0.0735					
Port Jackson Shark	17												0.0208				
Other Demersal Shark	18	0.0025								0.0009							
Ray & Skate	19	0.0042											0.0534				
Southern Bluefin Tuna	20											0.0134	0.0059				
Yellowtail Kingfish	21											0.0134	0.0059				
Snapper	22				0.0019							0.0134	0.0710	0.0164			
Snook	23											0.0134	0.0111				
Barracouta	24	0.0005	0.0069			0.0059	0.3775			0.0571				0.2767			
Skipjack Trevally	25			0.0087	0.0637		0.0075			0.0005							
Medium piscivore fish	26	0.0427	0.0165	0.0118	0.0070						0.0200		0.1004	0.0130			
Medium echinoderm fish	27																
Australian Salmon	28	0.0020		0.0013	0.0170	0.0001	0.0200			0.0249		0.0134	0.0059				
Australian Herring	29		0.0079					0.0086			0.0009		0.0147	0.0313	0.0719		
King George Whiting	30	0.0010	0.0010	0.0065					0.0780				0.0059				
Garfish	31		0.0016		0.0030	0.0001	0.1000	0.0500	0.0431	0.0489	0.0067		0.0106	0.0051			
Red Mullet	32	0.0151					0.0575		0.0862	0.0007	0.0112		0.0036				
Silverbelly	33	0.0161	0.0072	0.0140	0.0019			0.0011	0.0862	0.0006	0.0093						
Medium crustacean fish	34	0.0962	0.0442	0.0201	0.0139				0.2755				0.0263	0.0033	0.0001		0.2000
Medium molluscan fish	35	0.0448					0.0300			0.0010	0.1561		0.0056				
Small crustacean fish	36	0.0024	0.1006	0.2357	0.0209			0.1202	0.1079	0.0357			0.0060	0.0009			

Group name	**Group**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**
Degens/Rough Leatherjacket	37		0.0727						0.2155	0.1294	0.2956		0.0002				
Small polychaete feeding fish	38	0.0684	0.0051	0.0436				0.0043	0.0862	0.0571	0.0039		0.0138	0.0165			
Syngnathids	39			0.0022						0.0014				0.0020			
Blue Mackerel	40			0.0004	0.0032	0.0007	0.1100			0.0043			0.0318				
Jack/Yellowtail Mackerel	41	0.0080	0.1042	0.0153	0.1140	0.0134				0.0016			0.0620		0.0012		0.2000
Sardine	42		0.0223	0.0209	0.3825	0.0014	0.1600	0.0429		0.2444	0.0355		0.2353		0.1712		
Anchovy	43		0.0019		0.2516	0.0177	0.0900	0.7407	0.0188	0.3072			0.0011	0.0073	0.7556		0.2000
Sprats	44				0.0063	0.0007	0.0075	0.0086		0.0755	0.0009		0.0006				
Fish larvae	45																
Southern Calamari	46	0.1465	0.0010	0.1337	0.0604		0.0200			0.0037	0.0009	0.0322	0.0943	0.3071			0.2000
Giant Australian Cuttlefish	47	0.1817	0.0001	0.2004	0.0249					0.0001	0.0215	0.0322	0.0767	0.1307			
Other squids	48	0.0532	0.1901	0.0004	0.0124	0.0088	0.0200	0.0225		0.0022	0.0019		0.0520	0.1593			
Octopus	49	0.2900	0.0115	0.2849	0.0029						0.0047	0.0322	0.0715			0.2000	0.2000
Rock lobster	50	0.0200															
Western King Prawn	51	0.0036						0.0011		0.0029			0.0010	0.0179		0.2500	
Blue Swimmer Crab	52										0.0039					0.2500	
Sand Crab	53																
Other large crabs/bugs	54										0.0039		0.0018	0.0009		0.3000	
Sand associated omnivore crustacean	55					0.0136			0.0001	0.0000	0.0078		0.0001	0.0115			
Herbivorous macrobenthos	56								0.0024		0.0353						
Sand zoobenthos feeder	57										0.0505						
Greenlip Abalone	58																
Blacklip Abalone	59																
Small mobile DDF crustacean	60								0.0001								
Small mobile ZF crustacean	61					0.1216											
Polychates DDF	62																
Sessile epifauna	63										0.0039		0.0039				
Gelatinous zooplankton	64																
Large carnivorous zooplankton	65																
Small herbivorous zooplankton	66																
Meiofauna	67																
Microphytobenthos	68																
Planktonic microflora	69																
Macroalgae	70																
Seagrass	71										0.0879						
Phytoplankton	72																
Detritus DOM water column	73																
Detritus POM sediment	74																
Discards	75				0.0127												
Import			0.4000			0.8160					0.2274						

Group name	Group	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33
Australian Sea Lion	1																	
Long-Nosed Fur Seal	2																	
Bottlenose Dolphin	3																	
Common Dolphin	4																	
Petrel	5																	
Australian Gannet	6																	
Little Penguin	7																	
Shags & Cormorants	8																	
Terns	9																	
Gulls	10																	
White Shark	11																	
Whaler Shark	12																	
Smooth Hammerhead	13																	
Common Thresher Shark	14																	
Gummy Shark	15																	
School Shark	16																	
Port Jackson Shark	17																	
Other demersal shark	18						0.0005											
Ray & skate	19																	
Southern Bluefin Tuna	20																	
Yellowtail Kingfish	21																	
Snapper	22						0.0000											
Snook	23																	
Barracouta	24						0.0010											
Skipjack Trevally	25					0.2050												
Medium piscivore fish	26						0.0048				0.0193							
Medium echinoderm fish	27																	
Australian Salmon	28				0.0046		0.0118											
Australian Herring	29					0.0174	0.1061						0.1056					
King George Whiting	30		0.0200				0.0290											
Garfish	31		0.0200				0.0004						0.0010					
Red Mullet	32		0.0200				0.0027											
Silverbelly	33		0.0200										0.0624					
Medium crustacean fish	34	0.0040	0.0200		0.0006	0.0110												
Medium molluscan fish	35	0.0040	0.0500								0.0660		0.0643					
Small crustacean fish	36	0.0040	0.1033				0.0469	0.5189			0.0277		0.0416				0.0147	

Group name	Group	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33
Degens/Rough Leatherjacket	37		0.0700				0.0067											
Small polychaete fish	38	0.0023	0.0400				0.0078				0.2959							
Syngnathids	39		0.0200				0.0001											
Blue Mackerel	40				0.0041	0.0694			0.0785				0.1040					
Jack/Yellowtail Mackerel	41				0.0196	0.1640	0.0002		0.0926		0.3329		0.0832					
Sardine	42				0.2292	0.0752	0.0085	0.2689	0.5505				0.1768	0.0800				
Anchovy	43				0.1316			0.1387	0.0965				0.3573	0.0800				
Sprats	44				0.0183		0.0002							0.0900				
Fish larvae	45																	
Southern Calarmari	46	0.2015					0.0041				0.0210							
Giant Australian Cuttlefish	47				0.0003	0.0110	0.0127											
Other squids	48	0.0731			0.0916	0.4470	0.0144	0.0723	0.1820		0.0183		0.0038					
Octopus	49	0.0119					0.0095											
Rock Lobster	50						0.0003											
Western King Prawn	51	0.1048	0.1315				0.0107				0.0868				0.0037			
Blue Swimmer Crab	52						0.2345											
Sand Crab	53	0.0403					0.0033											
Other large crabs/bugs	54						0.1829				0.0339				0.0408		0.1031	
Sand associated omnivore crustacean	55	0.2834	0.2825	0.0897			0.0237			0.3529	0.0577			0.3500	0.0111		0.5302	0.0150
Herbivorous macrobenthos	56	0.0023					0.0685					0.9977						0.0371
Sand zoobenthos feeder	57		0.0602				0.0456								0.0222			0.0106
Greenlip Abalone	58																	
Blacklip Abalone	59																	
Small mobile DDF crustacean	60	0.1209		0.0368			0.0414			0.1538	0.0036			0.4000	0.0865	0.0078		
Small mobile ZF crustacean	61			0.0488				0.0013		0.1584	0.0017	0.0023			0.0012	0.2870	0.1768	0.0247
Polychaetes DDF	62	0.1474	0.0381	0.8220			0.0212			0.3349	0.0161				0.8345	0.1337	0.1752	0.8879
Sessile epifauna	63						0.0957				0.0184							0.0247
Gelatinous zooplankton	64																	
Large carnivorous zooplankton	65																	
Small herbivorous zooplankton	66																	
Meiofauna	67																	
Microphytobenthos	68																	
Planktonic microflora	69																	
Macroalgae	70	0.0002	0.0159				0.0010				0.0007							
Seagrass	71		0.0886	0.0028			0.0037				0.0001					0.5699		
Phytoplankton	72																	
Detritus DOM water column	73																	
Detritus POM sediment	74																	
Discards	75																	
Import					0.5000											0.0016		

Group name	Group	34	35	36	37	38	39	40	41	42	43	44	45	46	47	48	49
Australian Sea Lion	1																
Long-Nosed Fur Seal	2																
Bottlenose Dolphin	3																
Common Dolphin	4																
Petrel	5																
Australian Gannet	6																
Little Penguin	7																
Shags & Cormorants	8																
Terns	9																
Gulls	10																
White Shark	11																
Whaler Shark	12																
Smooth Hammerhead	13																
Common Thresher Shark	14																
Gummy Shark	15																
School Shark	16																
Port Jackson Shark	17																
Other demersal shark	18																
Ray & skate	19																
Southern Bluefin Tuna	20																
Yellowtail Kingfish	21																
Snapper	22															0.0150	
Snook	23															0.0025	
Barracouta	24															0.0255	
Skipjack Trevally	25															0.0150	
Medium piscivore fish	26													0.1009		0.0033	
Medium echinoderm fish	27															0.0025	
Australian Salmon	28													0.1070		0.0085	
Australian Herring	29															0.0200	
King George Whiting	30																
Garfish	31													0.0010		0.0005	
Red Mullet	32															0.0124	
Silverbelly	33															0.0015	0.0500
Medium crustacean fish	34																
Medium molluscan fish	35																
Small crustacean fish	36	0.0982												0.0510	0.1000	0.0085	0.1500

Group name	Group	34	35	36	37	38	39	40	41	42	43	44	45	46	47	48	49
Degens/Rough Leatherjacket	37															0.1959	
Small polychaete fish	38			0.0038										0.0143	0.1000	0.0010	
Syngnathids	39	0.0051															
Blue Mackerel	40															0.0600	
Jack/Yellowtail Mackerel	41													0.4016		0.1002	
Sardine	42													0.0510		0.0400	
Anchovy	43													0.1142		0.0100	
Sprats	44													0.0510			
Fish Larvae	45							0.1875	0.0241	0.0161	0.0678						
Southern Calarmari	46															0.0109	
Giant Australian Cuttlefish	47		0.0070													0.0092	0.0500
Other squids	48	0.0008	0.0024	0.0006										0.0051		0.0050	
Octopus	49	0.0494				0.0002								0.0163			
Rock Lobster	50																
Western King Prawn	51	0.3936		0.0354											0.1000	0.0100	
Blue Swimmer Crab	52	0.1196													0.1500		
Sand crab	53														0.1500	0.0100	
Other large crabs/bugs	54	0.0110		0.2068		0.0113									0.1500	0.0100	
Sand associated omnivore crustacean	55	0.1558	0.1403	0.4351	0.0485	0.0478	0.6127	0.1490	0.1168	0.2837	0.4719	0.8650		0.0866	0.1500	0.0844	0.6500
Herbivorous macrobenthos	56	0.0030	0.1917	0.0055	0.0635	0.0454											
Sand zoobenthos feeder	57		0.4416	0.0068	0.0221	0.0140		0.0237	0.0062	0.0048					0.1000	0.0012	0.1000
Greenlip Abalone	58																
Blacklip Abalone	59																
Small mobile DDF crustacean	60	0.0249	0.0380	0.1488	0.1658	0.0408	0.1960										
Small mobile ZF crustacean	61	0.0580	0.0339	0.0632	0.0553	0.1486	0.1652	0.6178	0.7260	0.6641	0.4604					0.2920	
Polychaetes DDF	62	0.0210	0.0786	0.0567	0.1041	0.2855			0.1136	0.0000						0.0453	
Sessile epifauna	63	0.0552	0.0387	0.0309	0.2460	0.1108	0.0024										
Gelatinous zooplankton	64							0.0062	0.0132	0.0290							
Large carnivorous zooplankton	65																
Small herbivorous zooplankton	66												0.8000				
Meiofauna	67																
Microphytobenthos	68																
Planktonic microflora	69																
Macroalgae	70	0.0025	0.0193	0.0053	0.2718	0.2624	0.0237					0.1350					
Seagrass	71	0.0019	0.0084	0.0011	0.0229	0.0334	0.0000	0.0158	0.0000	0.0023							
Phytoplankton	72												0.2000				
Detritus DOM water column	73																
Detritus POM sediment	74																
Discards	75																
Import																	

Group name	Group	50	51	52	53	54	55	56	57	58	59	60	61	62	63	64	65
Degens/Rough leatherjacket	37																
Small polychaete fish	38																
Syngnathids	39																
Blue Mackerel	40																
Jack/Yellowtail Mackerel	41																
Sardine	42																
Anchovy	43																
Sprats	44																
Fish larvae	45															0.2570	
Southern Calarmari	46																
Giant Australian Cuttlefish	47																
Other squids	48																
Octopus	49																
Rock Lobster	50																
Western King Prawn	51																
Blue Swimmer Crab	52						0.0025										
Sand Crab	53						0.0075										
Other large crabs/bugs	54						0.1800		0.0010								
Sand associated omnivore crustacean	55	0.2393		0.1000	0.1500												
Herbivorous macrobenthos	56	0.3742		0.0500	0.0300	0.0010			0.1000								
Sand zoobenthos feeder	57			0.1700	0.2500	0.0011	0.2000										
Greenlip Abalone	58																
Blacklip Abalone	59	0.0123															
Small mobile DDF crustacean	60			0.1000	0.1000		0.0100										
Small mobile ZF crustacean	61								0.0050							0.2280	
Polychaetes DDF	62		0.1000	0.2500	0.2000									0.0500			
Sessile epifauna	63																
Gelatinous zooplankton	64																
Large carnivorous zooplankton	65												0.6100				
Small herbivorous zooplankton	66												0.1000			0.2570	0.8000
Meiofauna	67		0.1000						0.0200			0.1000		0.1000			
Microphytobenthos	68		0.1000			0.0061	0.0510	0.5000	0.1500			0.5440	0.2900	0.5500	0.1500		
Planktonic microflora	69													0.0500	0.6000	0.1290	0.2000
Macroalgae	70	0.3742	0.5000	0.0800	0.0600	0.1630	0.3000	0.5000		0.8100	0.6200	0.0010					
Seagrass	71					0.1260				0.1500	0.3300						
Phytoplankton	72														0.1000	0.1290	
Detritus DOM water column	73											0.0050		0.1250			
Detritus POM sediment	74		0.2000	0.2500	0.2100	0.5518	0.2490	0.0000	0.7240	0.0400	0.0500	0.3500		0.1250	0.1500		
Discards	75																
Import						0.1510											

Group name	Group	66	67	68	69
Degens/Rough Leatherjacket	37				
Small polychaete fish	38				
Syngnathids	39				
Blue Mackerel	40				
Jack/Yellowtail Mackerel	41				
Sardine	42				
Anchovy	43				
Sprats	44				
Fish larvae	45				
Southern Calarmari	46				
Giant Australian Cuttlefish	47				
Other squids	48				
Octopus	49				
Rock Lobster	50				
Western King Prawn	51				
Blue Swimmer Crab	52				
Sand Crab	53				
Other large crabs/bugs	54				
Sand associated omnivore crustacean	55				
Herbivorous macrobenthos	56				
Sand zoobenthos feeder	57				
Greenlip Abalone	58				
Blacklip Abalone	59				
Small mobile DDF crustacean	60				
Small mobile ZF crustacean	61				
Polychates DDF	62				
Sessile epifauna	63				
Gelatinous zooplankton	64				
Large carnivorous zooplankton	65				
Small herbivorous zooplankton	66				
Meiofauna	67		0.0500		
Microphytobenthos	68		0.7500	0.1000	0.0200
Planktonic microflora	69	0.0500		0.1000	0.1000
Macroalgae	70				
Seagrass	71				
Phytoplankton	72	0.9000			
Detritus DOM water column	73	0.0500	0.2000	0.0400	0.6000
Detritus POM sediment	74			0.7600	0.2800
Discards	75				
Import					

Appendix 4. Biomass estimates for bycatch species from the GSV Prawn fishery

Group name	Common name	Scientific name	Mean biomass (t km²)	Standard error
Port Jackson Shark	Port Jackson Shark	*Heterodontus portusjacksoni*	0.0205225	0.0047718
Other demersal shark	Angel Shark	*Squatina australis*	0.0083797	0.0219242
Other demersal shark	Elephant Fish	*Callorhinchus milii*	0.0021284	0.0037914
Other demersal shark	Rusty Catshark	*Parascyllium ferrugineum*	0.0004921	
Ray & skate	Coastal Stingaree	*Urolophus orarius*	0.0001625	0.0003151
Ray & skate	Sparsely-Spotted Stingaree	*Urolophus paucimaculatus*	0.0016398	0.0011308
Ray & skate	Southern Fiddler Ray	*Trygonorrhina fasciata*	0.0150147	0.0136650
Ray & skate	Melbourne Skate	*Dipturus whitleyi*	0.0125391	0.0035931
Ray & skate	Southern Shovelnose Ray	*Aptychotrema vincentiana*	0.0005858	0.0013267
Ray & skate	Smooth Stingray	*Dasyatis brevicaudata*	0.0832937	0.0145346
Ray & skate	White spotted skate		0.0000958	
Ray & skate	Western Shovelnose Stingaree	*Trygonoptera mucosa*	0.0086252	0.0113413
Snapper	Snapper	*Pagrus auratus*	0.0090813	0.0096661
Barracouta	Barracouta	*Thyrsites atun*	0.0033708	0.0049318
Skipjack trevally	Skipjack Trevally	*Pseudocaranx wrighti*	0.1929389	0.0267710
Medium piscivore fish	Small Tooth Flounder	*Pseudorhombus jenynsii*	0.0127388	0.0057050
Medium piscivore fish	Common Stargazer	*Kathetostoma laeve*	0.0015927	0.0045385
Medium piscivore fish	Tiger Flathead	*Neoplatycephalus richardsoni*	0.0348707	0.0068153
Medium piscivore fish	Red Cod	*Pseudophycis bachus*	0.0000212	0.0000303
Medium piscivore fish	Tasselled Anglerfish	*Rhycherus filamentosus*	0.0000055	
Medium piscivore fish	Flathead no. 2		0.0010880	0.0004470
King George whiting	King George Whiting	*Sillaginodes punctata*	0.0010170	0.0010099
Garfish	Southern Garfish	*Hyporhamphus melanochir*	0.0001323	0.0005070
Red mullet	Red Mullet (Bluespotted Goatfish)	*Upeneichthys vlamingii*	0.0158407	0.0042436
Silverbelly	Silverbelly	*Parequula melbournensis*	0.0111045	0.0029806
Medium crustacean fish	Southern Tongue Sole	*Cynoglossus broadhursti*	0.0002514	0.0001855
Medium crustacean fish	Gurnard Perch	*Neosebastes pandus*	0.0027954	0.0040953
Medium crustacean fish	Blue Warehou		0.0003717	0.0007858
Medium molluscan fish	Spotted Stinkfish (Spotted Dragonet)	*Repomucenus calcaratus*	0.0271482	0.0051664
Medium molluscan fish	Striped Perch (Western Striped Grunter)	*Pelates octolineatus*	0.0010009	0.0002388
Medium molluscan fish	Spikey Globefish	*Diodon nicthemerus*	0.0108863	0.0032668
Medium molluscan fish	Ringed Toadfish	*Omegophora armilla*	0.0006556	0.0014712
Medium molluscan fish	Common Stink Fish	*Foetorepus calauropomus*	0.0157991	0.0036588
Medium molluscan fish	Southern Gobbleguts (Southern cardinalfish)	*Vincentia conspersa*	0.0011009	0.0004065
Medium molluscan fish	Beaked Salmon	*Gonorynchus greyi*	0.0003901	0.0003036
Medium molluscan fish	Fringed Stargazer	*Ichthyscopus barbatus*	0.0003005	
Medium molluscan fish	Chinaman Leather Jacket	*Nelusetta ayraudi*	0.0000283	
Small crustacean fish	Little Scorpion Fish (Little Gurnard Perch)	*Maxillicosta scabriceps*	0.0068448	0.0015934
Small crustacean fish	Slender Bullseye (Elongate Bullseye)	*Parapriacanthus elongatus*	0.0069022	0.0077917
Small crustacean fish	Spiny Gurnard	*Lepidotrigla papilio*	0.0249933	0.0033120
Small crustacean fish	Spotted Grubfish	*Parapercis ramsayi*	0.0016453	0.0024021
Small crustacean fish	Wavy Grubfish	*Parapercis haackei*	0.0001764	0.0002450
Small crustacean fish	Gulf Gurnard Perch	*Neosebastes bougainvillii*	0.0052396	0.0034937
Small crustacean fish	Soldier Fish	*Gymnapistes marmoratus*	0.0005935	0.0003145
Small crustacean fish	Sthn. Pygmy Leatherjacket	*Brachaluteres jacksonianus*	0.0002338	0.0001529
Small crustacean fish	Many Banded Sole	*Zebrias scalaris*	0.0001912	0.0001843
Small crustacean fish	Sculptured Seamoth	*Pegasus lancifer*	0.0000357	0.0000142
Small crustacean fish	Latchet	*Pterygotrigla polyommata*	0.0007091	0.0010159

Group name	**Common name**	**Scientific name**	**Mean biomass (t km^2)**	**Standard error**
Degens/Rough leatherjacket	Rough Leatherjacket	*Scobinichthys granulatus*	0.0143624	0.0045946
Degens/Rough leatherjacket	Degens Leatherjacket (Bluefin)	*Thamnaconus degeni*	0.0875597	0.0213356
Small polychaete fish	Bridled Leatherjacket	*Acanthaluteres spilomelanurus*	0.0052929	0.0010623
Small polychaete fish	Silver Whiting (Sthn. School Whiting)	*Sillago bassensis*	0.0059240	0.0025304
Small polychaete fish	Longsnout Boarfish	*Pentaceropsis recurvirostris*	0.0000955	
Small polychaete fish	Toothbrush Leatherjacket	*Acanthaluteres vittiger*	0.0011906	0.0004862
Small polychaete fish	Ornate Cowfish	*Aracana ornata*	0.0031631	0.0012518
Small polychaete fish	Mosaic Leatherjacket	*Eubalichthys mosaicus*	0.0022220	0.0003785
Small polychaete fish	Crested Flounder	*Lophonectes gallus*	0.0004273	0.0002774
Small polychaete fish	Shaws Cowfish	*Aracana aurita*	0.0005512	0.0002754
Small polychaete fish	Gunn's Leatherjacket	*Eubalichthys gunnii*	0.0000088	
Small polychaete fish	Goblin Fish	*Glyptauchen panduratus*	0.0000712	0.0000229
Small polychaete fish	Longsnout Flounder		0.0002302	0.0003547
Syngnathids	Bigbelly Seahorse	*Hippocampus abdominalis*	0.0000012	
Jack/yellowtail mackerel	Jack Mackerel	*Trachurus declivis*	0.0460397	0.0295558
Anchovy	Australian Anchovy	*Engraulis australis*	0.0001960	0.0002380
Sprats	Sandy Spratt	*Hyperlophus vittatus*	0.0000021	
Southern calamari	Southern Calamary	*Sepioteuthis australis*	0.0300369	0.0060971
Other squids	Nova Cuttlefish	*Sepia novaehollandae*	0.0093409	0.0039442
Other squids	Southern Bottletail Squid	*Sepiadarium austrinum*	0.0000040	
Other squids	Striped Pyjama Squid	*Sepioloidea lineolata*	0.0000713	
Other squids	Braggi's Cuttle	*Sepia braggi*	0.0001666	0.0001270
Octopus	Southern Keeled Octopus	*Octopus berrima*	0.0001679	
Octopus	Pale Octopus	*Octopus pallidus*	0.0000449	0.0000708
Octopus	Southern Sand Octopus	*Octopus kaurna*	0.0000239	
Western king prawn	Western King Prawn	*Melicertus latisulcatus*	0.3385099	0.0564801
Blue swimmer crab	Blue Swimmer crab	*Portunus (Portunus) pelagicus*	0.1593908	0.0568070
Sand crab	Sand Crab	*Ovalipes australiensis*	0.0003838	0.0001687
Other large crabs/bugs	Balmain Bug (Eastern Balmain Bug)	*Ibacus peronii*	0.0050855	0.0038968
Other large crabs/bugs	Hairy Shore Crab	*Pilumnidae sp.*	0.0001877	0.0000642
Other large crabs/bugs	Bristled Sponge Crab	*Austrodromidia octodentata*	0.0001698	0.0002955
Other large crabs/bugs	Common Hermit crab	*Paguristes frontalis*	0.0006789	0.0002192
Other large crabs/bugs	Shaggy Sponge Crab	*Lamarckdromia globosa*	0.0001457	0.0001334
Other large crabs/bugs	Rock Crab (Rough Rock Crab)	*Nectocarcinus integrifrons*	0.0000613	0.0001787
Other large crabs/bugs	Great Spider Crab	*Leptomithrax gaimardii*	0.0003235	0.0002214
Other large crabs/bugs	Facetted Crab	*Actaea calculosa*	0.0002127	0.0000851
Other large crabs/bugs	Southern Sponge Crab	*Austrodromidia australis*	0.0000658	0.0000400
Other large crabs/bugs	Pilumnidae crab		0.0001810	0.0000840
Other large crabs/bugs	Giant Spider Crab		0.0016981	0.0021211
SAO crustaceans	Strawberry Prawn	*Metapenaeopsis sp.*	0.0067050	0.0016544
SAO crustaceans	Small hermit crab	`	0.0000171	0.0000242
SAO crustaceans	Small crabs		0.0001328	0.0001074
Herbivorous macrobenthos	Ophiothrix caespitosa	*Ophiothrix (Ophiothrix) caespitosa*	0.0002178	0.0001951
Herbivorous macrobenthos	Thorny Sea Urchin	*Goniocidaris tubaria*	0.0000477	0.0000115
Herbivorous macrobenthos	Schayer's brittlestar	*Ophionereis schayeri*	0.0000107	0.0000424
Herbivorous macrobenthos	Handsome Sea Cucumber	*Holothuria (Thymiosycia) hartmeyeri*	0.0030076	0.0041986
Herbivorous macrobenthos	Southern Sand Star	*Luidia australiae*	0.0012118	0.0010923
Herbivorous macrobenthos	Goniodiscaster	*Goniodiscaster seriatus*	0.0007867	0.0006614
Herbivorous macrobenthos	Eleven-armed seastar	*Coscinasterius muricata*	0.0114474	0.0169571
Herbivorous macrobenthos	Sea cucumber		0.0006303	0.0007366
Herbivorous macrobenthos	Wavyvolute	*Amoria undulata*	0.0000169	0.0000084

Group name	Common name	Scientific name	Mean biomass (t km²)	Standard error
Herbivorous macrobenthos	Nudibranch	*Ceratosoma brevicaudatum*	0.0004147	0.0014060
Herbivorous macrobenthos	Spindle Shell	*Fusinus (Fusinus) australis*	0.0000197	
Herbivorous macrobenthos	Fluted Murex	*Pterynotus triformis*	0.0000799	0.0000287
Herbivorous macrobenthos	New Holland spindle	*Fusinus novaehollandiae*	0.0000921	0.0002276
Herbivorous macrobenthos	Pear Helmet	*Semicassis pyrum*	0.0002907	
Sand zoobenthos feeders	Lima Lima	*Lima vulgaris*	0.0002034	
Sand zoobenthos feeders	Southern Hammer Oyster	*Malleus (Malleus) meridianus*	0.0010881	0.0008008
Sand zoobenthos feeders	Doughboy Scallop (Sponge Scallop)	*Mimachlamys asperrima*	0.0007035	0.0005696
Sand zoobenthos feeders	Cockle	*Acrosterigma cygnorum*	0.0000284	
Sand zoobenthos feeders	Mud Oyster (Native Oyster)	*Ostrea (Eostrea) angasi*	0.0000918	0.0001392
Sand zoobenthos feeders	Rock Shell	*Cleidothaerus albidus*	0.0003485	0.0005649
Sand zoobenthos feeders	Commercial Scallop	*Pecten fumatus*	0.0015081	0.0009030
Sand zoobenthos feeders	Queen Scallop	*Equichlamys bifrons*	0.0007443	0.0009217
Sand zoobenthos feeders	Razor Fish sp. 2	*Atrina (Servatrina) tasmanica*	0.0009513	0.0020108
Sand zoobenthos feeders	Grooved cardita	*Cardita incrassiata*	0.0000325	
Small mobile DDF crustaceans	Snapping Prawn (Hairy Pistol Prawn)	*Alpheus villosus*	0.0000133	0.0000138
Small mobile DDF crustaceans	Pistol Shrimp (Coral Snapping Shrimp)	*Alpheus lottini*	0.0000013	0.0000009
Small mobile DDF crustaceans	Mantis Shrimp	*Erugosquilla grahami*	0.0018812	0.0004318
Small mobile DDF crustaceans	Long-Wristed Shrimp	*Processa gracilis*	0.0000011	0.0000024
Polychaete DDF	Polychaeta		0.0008972	0.0015012
Sessile epifauna	Sea Tulip	*Pyura gibbosa*	0.0000176	
Sessile epifauna	Blue Ascidian	*Ascidia sydneiensis*	0.0018091	0.0022040
Sessile epifauna	Spined Ascidian	*Herdmania momus*	0.0158165	0.0118911
Sessile epifauna	Polycarpa	*Polycarpa pedunculata*	0.0002483	0.0003934
Sessile epifauna	Trididemnum	*Trididemnum cerebriforme*	0.0001388	0.0003212
Sessile epifauna	Pyura sp. 2	*Pyura molguloides*	0.0000468	
Sessile epifauna	Christmas Tree Ascidian	*Halocynthia dumosa*	0.0001899	0.0001345
Sessile epifauna	Cnemidocarpa	*Cnemidocarpa radicosa*	0.0000977	0.0003026
Sessile epifauna	Cystodytes	*Cystodytes dellachiajei*	0.0001995	0.0003611
Sessile epifauna	Polysyncraton	*Polysyncraton aspiculatum*	0.0000389	
Sessile epifauna	Phallusia	*Phallusia obesa*	0.0013156	0.0010685
Sessile epifauna	Cunjuvoi	*Pyura stolonifera*	0.0068124	0.0046619
Sessile epifauna	Didemnum	*Didemnum augusti*	0.0000735	0.0000861
Sessile epifauna	Holozoid	*Sycozoa cerebriformis*	0.0000048	
Sessile epifauna	White compound ascidian		0.0006791	0.0002296
Sessile epifauna	Speckled compound ascidian		0.0008949	0.0017106
Sessile epifauna	Strawberry compound ascidian		0.0000511	
Sessile epifauna	Grey sandy ascidian		0.0000443	
Sessile epifauna	Ascidian		0.0004768	
Sessile epifauna	Large leathery solitary ascidian		0.0003089	0.0002049
Sessile epifauna	Celleporaria	*Celleporaria fusca*	0.0100612	0.0057777
Sessile epifauna	Adeona	*Adeona grisea*	0.0003642	0.0004282
Sessile epifauna	Steginoporella	*Steginoporella chartacea*	0.0336179	0.0740309
Sessile epifauna	Lace Bryozoan	*Triphyllozoon moniliferum*	0.0009497	0.0004086
Sessile epifauna	Purple Bryozoan	*Iodictyum phoniceum*	0.0000952	0.0001129
Sessile epifauna	Black encrusting	*Celleporaria sp.*	0.0041142	0.0100421
Sessile epifauna	Black vane		0.0028348	0.0009472
Sessile epifauna	Orange fine vanes		0.0000940	
Sessile epifauna	Orange/pink fenestrate	*Triphyllozoon sp.*	0.0003688	0.0001554
Sessile epifauna	Adenopsis 1	*Adenopsis 1*	0.0001158	

Group name	**Common name**	***Scientific name***	**Mean biomass (t km²)**	**Standard error**
Sessile epifauna	Canda	*Canda*	0.0000274	0.0000298
Sessile epifauna	Adenopsis 2	*Adenopsis 2*	0.0000313	0.0001219
Sessile epifauna	Goose barnacle		0.0001487	0.0006195
Sessile epifauna	Halopteris sp. 1	*Halopteris campanula*	0.0000009	
Sessile epifauna	Hydroid sp. 3		0.0000017	
Sessile epifauna	Hairy Mussel	*Trichomya hirsuta*	0.0000855	
Sessile epifauna	Carijoa	*Carijoa multiflora*	0.0030041	0.0031872
Sessile epifauna	Sea Pen	*Sarcoptilus grandis*	0.0001211	
Sessile epifauna	Clathria sp. 1	*Clathria sp. 1*	0.0104367	0.0050851
Sessile epifauna	Cannon Ball Sponge	*Ecionemia sp. 1*	0.0113191	0.0406232
Sessile epifauna	Ircinia sp. 1	*Ircinia sp.*	0.0303736	0.0441204
Sessile epifauna	Poecilosclerid sp. 1	*Poecilosclerid sp. 1*	0.0910334	0.0409196
Sessile epifauna	Dictyoceratid sp. 1	*Dictyoceratid sp. 1*	0.0008555	0.0036731
Sessile epifauna	Chondropsid sp. 1	*Chondropsid sp. 1*	0.0341257	0.0491163
Sessile epifauna	Honey Comb Sponge	*Holopsamma laminaefavosa*	0.0122127	0.0081319
Sessile epifauna	Demosponge sp. 1	*Demosponge sp. 1*	0.0007676	
Sessile epifauna	Demosponge sp. 4	*Demosponge sp. 4*	0.0001766	
Sessile epifauna	Haplosclerid sp. 2	*Haplosclerid sp. 2*	0.0001877	0.0001906
Sessile epifauna	Dictyoceratid sp. 2	*Dictyoceratid sp. 2*	0.0127890	0.0247498
Sessile epifauna	Dictyoceratid sp. 3	*Dictyoceratid sp. 3*	0.0016704	
Sessile epifauna	Clathria sp. 2	*Clathria sp. 2*	0.0008637	0.0010361
Sessile epifauna	Thorectid sp. 1	*Thorectid sp.*	0.0020787	0.0075007
Sessile epifauna	Dictyoceratid sp. 4	*Dictyoceratid sp. 4*	0.0003893	0.0004932
Sessile epifauna	Demosponge sp. 64	*Demosponge sp. 64*	0.0002501	
Sessile epifauna	Demosponge sp. 6	*Demosponge sp. 6*	0.0002795	0.0003675
Sessile epifauna	Ecionemia sp. 2	*Ecionemia sp. 2*	0.0002547	
Sessile epifauna	Bath Sponge	*Spongiid sp. 2*	0.0003108	0.0005078
Sessile epifauna	Demosponge sp. 7	*Demosponge sp. 7*	0.0003124	
Sessile epifauna	Demosponge sp. 8	*Demosponge sp. 8*	0.0010382	
Sessile epifauna	Demosponge sp. 9	*Demosponge sp. 9*	0.0000388	
Sessile epifauna	Sponge sp. 27		0.0016883	0.0020694
Sessile epifauna	Dictyoceratid sp. 6	*Dictyoceratid sp. 6*	0.0004311	
Sessile epifauna	Dictyoceratid sp. 7	*Dictyoceratid sp. 7*	0.0075781	0.0108355
Sessile epifauna	Holopsamma sp. 3	*Holopsamma sp. 3*	0.0000830	0.0001671
Sessile epifauna	Sponge sp. 33		0.0020233	0.0030933
Sessile epifauna	Demosponge sp. 12	*Demosponge sp. 12*	0.0333159	0.0154918
Sessile epifauna	Demosponge sp. 14	*Demosponge sp. 14*	0.0002235	
Sessile epifauna	Dictyoceratid sp. 8	*Dictyoceratid sp. 8*	0.0302949	0.0702309
Sessile epifauna	Siphonochalina sp. 1	*Siphonochalina sp.*	0.0000705	
Sessile epifauna	Demosponge sp. 17	*Demosponge sp. 17*	0.0000094	
Sessile epifauna	Sponge sp. 43		0.0000316	
Sessile epifauna	Demosponge sp. 20	*Demosponge sp. 20*	0.0000863	0.0001866
Sessile epifauna	Spheciospongia	*Shpeciospongia papillosa*	0.0022248	
Sessile epifauna	Demosponge sp. 22	*Demosponge sp. 22*	0.0004476	
Sessile epifauna	Demosponge sp. 29	*Demosponge sp. 29*	0.0001222	0.0004751
Sessile epifauna	Demosponge sp. 30	*Demosponge sp. 30*	0.0049127	0.0097258
Sessile epifauna	Demosponge sp. 31	*Demosponge sp. 31*	0.0063318	0.0027655
Sessile epifauna	Demosponge sp. 36	*Demosponge sp. 36*	0.0003978	
Sessile epifauna	Demosponge sp. 37	*Demosponge sp. 37*	0.0008135	
Sessile epifauna	Haplosclerid sp. 3	*Haplosclerid sp. 3*	0.0002114	0.0007645
Sessile epifauna	Demosponge sp. 43	*Demosponge sp. 43*	0.0030417	
Sessile epifauna	Demosponge sp. 44	*Demosponge sp. 44*	0.0018143	

Group name	Common name	*Scientific name*	Mean biomass (t km²)	Standard error
Sessile epifauna	Chondrilla sp. 1	*Chondrilla sp.*	0.0002329	
Sessile epifauna	Demosponge sp. 47	*Demosponge sp. 47*	0.0001196	
Sessile epifauna	Dictyoceratid sp. 9	*Dictyoceratid sp. 9*	0.0003883	0.0008470
Sessile epifauna	Sponge sp. 82		0.0009921	
Sessile epifauna	Verongid sp. 2	*Verongid sp. 2*	0.0006493	0.0027231
Sessile epifauna	Demosponge sp. 50	*Demosponge sp. 50*	0.0002634	
Sessile epifauna	Demosponge sp. 51	*Demosponge sp. 51*	0.0009692	0.0012162
Sessile epifauna	Demosponge sp. 52	*Demosponge sp. 52*	0.0002713	0.0003840
Sessile epifauna	Sponge sp. 87		0.0066621	0.0223972
Sessile epifauna	Demosponge sp. 54	*Demosponge sp. 54*	0.0010009	0.0015196
Sessile epifauna	Demosponge sp. 55	*Demosponge sp. 55*	0.0001717	
Sessile epifauna	Demosponge sp. 56	*Demosponge sp. 56*	0.0001168	0.0000997
Sessile epifauna	Dictyoceratid sp. 10	*Dictyoceratid sp. 10*	0.0002237	0.0002629
Sessile epifauna	Demosponge sp. 63	*Demosponge sp. 63*	0.0003822	
Sessile epifauna	Arenochalina sp. 1	*Arenochalina sp.*	0.0000255	
Sessile epifauna	Tethya	*Tethya sp. 1*	0.0000717	
Sessile epifauna	Dendrilla	*Dendrilla rosea*	0.0004028	0.0011007
Sessile epifauna	Honeycomb sponge	*Holopsamma sp.*	0.0198089	0.0073897
Sessile epifauna	Demosponge sp. 64	*Demosponge sp. 64*	0.0001311	0.0002218
Sessile epifauna	Demosponge sp. 65	*Demosponge sp. 65*	0.0009086	0.0011965
Sessile epifauna	Demosponge sp. 66	*Demosponge sp. 66*	0.0026794	0.0055606
Sessile epifauna	Demosponge sp. 68	*Demosponge sp. 68*	0.0010598	
Sessile epifauna	Demosponge sp. 69	*Demosponge sp. 69*	0.0001133	
Sessile epifauna	Holopsamma sp.	*Holopsamma sp.*	0.0000942	0.0002560
Sessile epifauna	Demosponge sp. 70	*Demosponge sp. 70*	0.0002136	0.0003057
Sessile epifauna	Thorectid.	*Thorectid.*	0.0000505	
Sessile epifauna	Haplosclerid	*Haplosclerid*	0.0010060	0.0010826
Sessile epifauna	Calcarea	*Calcarea*	0.0002025	0.0006950
Sessile epifauna	Demosponge sp. 71	*Demosponge sp. 71*	0.0034026	0.0066776
Sessile epifauna	Demosponge sp. 72	*Demosponge sp. 72*	0.0008221	
Sessile epifauna	Hadromerida	*Hadromerida*	0.0046903	0.0198638
Sessile epifauna	Caulospongia sp.	*Caulospongia sp.*	0.0003509	
Macroalgae	Caulerpa	*Caulerpa cactoides*	0.0000010	
Macroalgae	Zonaria sp. 1	*Zonaria angustata*	0.0051579	0.0221774
Macroalgae	Gracilaria sp. 2	*Gracilaria flageliformis*	0.0071489	0.0237312
Macroalgae	Popcorn	*Sporolithon durum*	0.0000750	0.0002705
Macroalgae	Spongoclonium	*Spongoclonium conspicuum*	0.0000041	
Macroalgae	Zonaria sp. 2	*Zonaria turneriana*	0.0001126	
Macroalgae	Phacelocarpus	*Phacelocarpus peperocarpus*	0.0000188	
Macroalgae	Dictyota	*Dictyota ciliolata*	0.0000017	
Macroalgae	Botryocladia	*Botryocladia sonderi*	0.0034679	0.0048424
Macroalgae	Gelidium sp. 1	*Gelidium asperum*	0.0032390	0.0137105
Macroalgae	Cystophora sp. 1	*Cystophora sp. 1*	0.0000043	
Macroalgae	Hormosira	*Hormosira banksii*	0.0001334	
Macroalgae	Sargassum sp. 1	*Sargassum sp. 1*	0.0000334	0.0001196
Macroalgae	Cystophora sp. 3	*Cystophora sp. 3*	0.0000195	0.0000103
Macroalgae	Dasya	*Dasya extensa*	0.0000015	
Macroalgae	Scabera	*Scabera agardhii*	0.0000825	0.0000341
Macroalgae	Sargassum sp.	*Sargassum sp.*	0.0000361	0.0000533
Seagrass	Strapweed	*Posidonia sp.*	0.0000664	0.0000845
Seagrass	Amphibolis	*Amphibolis antartica*	0.0000344	0.0000575
Seagrass	Halophila	*Halophila australis*	0.0000025	

Appendix 5. Time series

Table showing time series of taxa used for the *Ecosim* model. For some commercial taxa data for multiple gear types were used. Abbreviations: EF, effort; 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; LNFS, long-nosed fur seal; ASL, Australian sea lion.

No.	**Name**
1	Sardine-PS EF
2	Prawn EF
3	Australian Salmon-SA EF
4	Blue Crab-CP EF
5	Rock Lobster-LP EF
6	Blacklip Abalone EF
7	Greenlip Abalone EF
8	Blue Crab-CN EF
9	Sand Crab-CN EF
10	Australian Herring-HN EF
11	Australian Salmon-HN EF
12	Blue Crab-HN EF
13	Garfish-HN EF
14	King George Whiting-HN EF
15	Other-HN EF
16	King George Whiting-HL EF
17	Snapper-HL EF
18	Other-HL EF
19	Shark-LL EF
20	Garfish-DN EF
21	Small Mesh NET EF
22	Large Mesh Set Net EF
23	Cockle-CR EF
24	Squid Jig EF
25	Troll Line EF
26	Drop Line EF
27	Fish Trap EF
28	Other EF
29	C Whaler sharks
30	C Smooth hammerhead
31	C Common thresher shark
32	C Gummy shark
33	C School shark
34	C Port Jackson shark
35	C Other demersal sharks
36	C Rays & skates
37	C Yellowtail kingfish
38	C Snapper
39	B Snapper
40	F Snapper
41	C Snook
42	C Barracouta
43	C Skipjack trevally
44	C Medium piscivore fish
45	C Medium echinoderm fish
46	C Australian salmon
47	C Australian herring

48	C King George Whiting
49	B King George Whiting
50	C Garfish
51	B Garfish
52	C Red mullet
53	C Silverbelly
54	C Medium crustacean fish
55	C Medium molluscan fish
56	C Small crustacean fish
57	C Degens/Rough leatherjacket
58	C Small polychaete fish
59	C Blue mackerel
60	C Jack/yellowtail mackerel
61	C Sardine
62	C Anchovy
63	C Southern Calarmari
64	C Giant cuttlefish
65	C Other squids
66	C Octopus
67	C Rock lobster
68	C Western king prawn
69	B Western king prawn
70	C Blue swimmer crab
71	C Sand crab
72	C Other large crabs/bugs
73	C SAO crustaceans
74	C Hebivorous macrobenthos
75	C Sand-zoobenthos feeders
76	C Greenlip abalone
77	C Black abalone
78	C Small mobile DDF crustaceans
79	C Polychaetes DDF
80	C Sessile epifauna
81	C Macroalgae
82	Sardine-PS CPUE
83	Blue Crab-CP CPUE
84	Rock Lobster-LP CPUE
85	Blacklip Abalone CPUE
86	Greenlip Abalone CPUE
87	Sand Crab-Cn CPUE
88	Australian Herring-HN CPUE
89	Australian Salmon-HN CPUE
90	Shark-LLCPUE
91	Cockle-Cr CPUE
92	Squid Jig CPUE
93	B LNFS
94	B ASL

11 References

ABARES (2014). Australian fisheries and aquaculture statistics 2013. Australian Bureau of Agricultural and Resource Economics and Sciences. Canberra. 115pp.

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.

Baum, J. K. and Worm, B. (2009). Cascading top-down effects of changing oceanic predator abundances. Journal of Animal Ecology, 78(4): 699-714.

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.

Beckmann, C. L. and Hooper, G. E. (2015). Blue Crab (*Portunus armatus*) Fishery 2013/14. Fishery Assessment Report to PIRSA Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2007/000729-11. SARDI Research Report Series No. 837. 70pp.

Beckmann, C. L. and Hooper, G. E. (2016). Blue Crab (*Portunus armatus*) Fishery 2014/15. Fishery Assessment Report to PIRSA Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2007/000729-12. SARDI Research Report Series No. 897. 57pp.

Beckmann, C. L., J., N. C. and Hooper, G. E. (2015). Status of the Gulf St Vincent Prawn *Penaeus* (*Melicertus) latisulcatus* Fishery in 2014/15. Fishery Status Report to PIRSA Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2007/00174-4. SARDI Research Report Series No. 870. 25pp.

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.

Burnell, O. W., Barrett, S. L., Hooper, G. E., Beckmann, C. L., Sorokin, S. J. and Noell, C. J. (2015). Spatial and temporal reassessment of by-catch in the Spencer Gulf Prawn Fishery. Report to PIRSA Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences). Adelaide. SARDI Publication No. F2015/00414-1. SARDI Research Report Series No. 854. 128pp.

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. (2000). Indicators for marine ecosystems affected by fisheries. Marine and Freshwater Research, 51: 447–450.

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

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

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

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

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.

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

Collette, B., Chang, S.-K., Di Natale, A., Fox, W., Juan Jorda, M., Miyabe, N., Nelson, R., Uozumi, Y. and Wang, S. (2011). *Thunnus maccoyii*. The IUCN Red List of Threatened Species 2011: e.T21858A9328286. http://dx.doi.org/10.2305/IUCN.UK.2011-2.RLTS.T21858A9328286.en. Available at http://www.iucnredlist.org/ [accessed 16 November 2015].

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

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.

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.

Cowan Jr, J. H., Rice, J. C., Walters, C. J., Hilborn, R., Essington, T. E., Day Jr, J. W. and Boswell, K. M. (2012). Challenges for implementing an ecosystem approach to fisheries management. Marine and Coastal Fisheries, 4(1): 496-510.

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

Currie, D. R., Dixson, C. D., Roberts, S. D., Hooper, G. E., Sorokin, S. J. and Ward, T. M. (2009b). 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. 121pp.

Currie, D. R., Hooper, G. E. and Ward, T. M. (2007). Blue Crab (*Portunus pelagicus*) Fishery 2005/06 Stock Assessment Report to PIRSA Fisheries. South Australian Research and Development Institute (Aquatic Sciences). Adelaide. RD -3/0274-4. 65pp.

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. 179pp.

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., 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.

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.

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.

Dixon, C., Raptis, J., Gorman, D., Roberts, S., Hooper, G., Bicknell, N., Sorokin, S., Newman, R., Noell, C., Benediktsson, T., Saint, J. and Hill, W. (2013a). A collaborative approach to novel by-catch research for rapid development, extension and adoption into a commercial trawl fishery. South Australian Research and

Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2012/000250-1. SARDI Research Report Series No. 643. 47pp.

Dixon, C., Roberts, S., Hooper, G., Steer, M. A. and Ward, T. M. (2006). Gulf St Vincent Prawn (*Melicertus latisulcatus*) Fishery. Fishery Assessment Report to PIRSA Fisheries. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. RD03/0063-4. SARDI Research Report Series No. 159. 97pp.

Dixon, C. D., Hooper, G. E. and Burch, P. (2012). Spencer Gulf Prawn Penaeus (*Melicertus latisulcatus*) Fishery 2010/11. Fishery Assessment Report to PIRSA Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2007/000770-5. SARDI Research Report Series No. 603. 118pp.

Dixon, C. D., Noell, C. J., Hooper, G. E. and Ward, T. M. (2013b). Blue Crab (*Portunus armatus*) Fishery 2011/12. Fishery Assessment Report to PIRSA Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2007/000729-9. SARDI Research Report Series No. 689. 70pp.

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.

Fowler, A. J. (2008). Chapter 28. The population biology of snapper (*Pagrus auratus*) in Gulf St Vincent. 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. 385-398.

Fowler, A. J. (2016). The influence of fish movement on regional fishery production and stock structure for South Australia's snapper (*Chrysophrys auratus*) fishery. Final Report to Fisheries Research and Development Corporation (FRDC) Project No. 2012/020. South Australian Research and Development Institute (Aquatic Sciences). Adelaide. 181pp.

Fowler, A. J. and Jones, G. K. (2008). Chpter 29. The population biology of King George whiting (*Sillaginodes punctata*) in Gulf St Vincent. 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. 399-414.

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. 79pp.

Fowler, A. J., McGarvey, R., Burch, P., Feenstra, J. E., Jackson, W. B. and Lloyd, M. T. (2013). Snapper (*Chrysophrys auratus*) Fishery. Fishery Assessment. Report to PIRSA Fisheries and Aquaculture. . South Australian Research and Development Institute (Aquatic Sciences). Adelaide. F2007/000523-3. SARDI Research Report Series No. 713. 103pp.

Fowler, A. J., McGarvey, R., Carroll, J. and Feenstra, J. E. (2014). King George Whiting (*Sillaginodes punctatus*) Fishery. Fishery Assessment Report to PIRSA Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences). Adelaide. F2007/000843-4. SARDI Research Report Series No. 801. 85pp.

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.

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.

Gillanders, B. M., Goldsworthy, S. D., Prowse, T., Doubell, M., Middleton, J., Rogers, P., Tanner, J., Clisby, N. A., James, C., Luick, J., Van Ruth, P., Bradshaw, C. J. A. and Ward, T. M. (2015). Spencer Gulf Research Initiative: Development of an Ecosystem Model for Fisheries and Aquaculture. University of Adelaide and SARDI Aquatic Sciences. Adelaide. 116pp.

Giri, K. and Hall, K. (2015). South Australian Recreational Fishing Survey. Fisheries Victoria Internal Report Series No. 62.

Gislason, H., Sinclair, M., Sainsbury, K. and O'boyle, R. (2000). Symposium overview: incorporating ecosystem objectives within fisheries management. ICES Journal of Marine Science: Journal du Conseil, 57(3): 468-475.

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., 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.

Goldsworthy, S. D., Mackay, A. I., Shaughnessy, P. D., Bailleul, F. and Holman, D. (2015). Maintaining the monitoring of pup production at key Australian sea lion colonies in South Australia (2014/15). Final Report to the Australian Marine Mammal Centre. South Australian Research and Development Institute (Aquatic Sciences). Adelaide. SARDI Publication No. F2010/000665-5. SARDI Research Report Series No. 871. . 73pp.

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. 213pp.

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.

Griffiths, S. P., Young, J. W., Lansdell, M. J., Campbell, R. A., Hamption, J., Hoyle, S. D., Langley, A., Bromhead, D. and Hinton, M. G. (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.

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.

Heritage, D. f. E. a. (2008). Marine Habitats in the Adelaide and Mount Lofty Ranges NRM Region. Final Report to the Adelaide and Mount Lofty Ranges Natural Resources Management Board for the program: Facilitate Coast, Marine and Estuarine Planning and Management by Establishing Regional Baselines. . Department for Environment and Heritage, Coast and Marine Conservation Branch Adelaide. 84pp.

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.

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.

Kangas, M. and Dixon, C. (2008). Chpter 34. Prawns of Gulf St Vincent. 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. 456-463.

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). 41pp.

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

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.

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.

Kleisner, K. and Pauly, D. (2011). The Marine Trophic Index (MTI), the Fishing in Balance (FiB) Index and the spatial expansion of fisheries. In: . In: Christensen, V., Lai, S., Palomares, M. L. D., Zeller, D. and Pauly, D. (Eds). The State of Biodiversity and Fisheries in Regional SeasFisheries Centre, University of British Columbia, vol. 19(3), pp. 41-44.

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.

Lloyd, M. (2010). Spatial variation in growth of snapper (*Chrysophrys auratus*) in South Australia gulf waters: An investigation of diet as a contributing factor. Honours Thesis. The University of Adelaide. 66 pp.

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

McGarvey, R. and Fowler., A. J. (2002). Seasonal growth of King George whiting (*Sillaginodes punctata)* from length-at-age samples truncated by legal minimum size. Fisheries Bulletin, 100: 545-558.

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

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.

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

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.

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.

Nayar, S., Collings, G., Pfening, P. and Royal, M. (2012). Managing nitrogen inputs into seagrass meadows near a coastal city: Flow-on from research to environmental improvement plans. Marine Pollution Bulletin, 64: 932-940.

Nayar, S., Collings, G. J., Miller, D. J., Bryars, S. and Cheshire, A. C. (2009). Uptake and resource allocation of inorganic carbon by the temperate seagrasses Posidonia and Amphibolis. Journal of Experimental Marine Biology and Ecology, 373(2): 87-95.

Noell, C. J., O'Neill, M. F., Carroll, J. D. and Dixon, C. D. (2015). A bio-economic model for South Australia's prawn trawl fisheries. Final Report. South Australian Research and Development Institute (Aquatic Sciences). Adelaide. CRC Project No. 2011/750. 115pp.

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.

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.

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.

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

Pauly, D., Christensen, V. and Walters, C. (2000). Ecopath, Ecosim, and Ecospace as tools for evaluating ecosystem impact of fisheries. ICES Journal of Marine Science, 57: 697–706.

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

Pikitch, E., Santora, C., Babcock, E., Bakun, A., Bonfil, R., Conover, D., Dayton, P., et al., Doukakis, P., Fluharty, D. and Heneman, B. (2004). Ecosystem-based fishery management. Science, 305(5682): 346-347.

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

Poiner, I. R., Glaister, J., Pitcher, R., Burridge, C., Wassenberg, T. J., Gribble, N., Hill, B., Blaber, S., Milton, D., Brewer, D. and Ellis, N. (1998). Final report on effects of trawling in the Far Northern Section of the Great Barrier Reef: 1991-1996. CSIRO Division of Marine Research, Cleveland.

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.

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

Ritchie, E. G. and Johnson, C. N. (2009). Predator interactions, mesopredator release and biodiversity conservation. Ecology letters, 12(9): 982-998.

Roberts, S. D. and Steer, M. A. (2010). By-product assessment in the Spencer Gulf Prawn Fishery with an emphasis on developing managment 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. 48pp.

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.

Shannon, L. J., Neira, S. and Taylor, M. (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., Goldsworthy, S. D., Burch, P. and Dennis, T. E. (2013). Pup numbers of the Australian sea lion (*Neophoca cinerea*) at The Pages Islands, South Australia, over two decades. Australian Journal of Zoology, 61: 112-118.

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.

Shaughnessy, P. D., Goldsworthy, S. D. and Mackay, A. I. (2015). The long-nosed fur seal in South Australia in 2013-14; abundance, status and trends. Australian Journal of Zoology, 63: 101-110.

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.

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.

Steer, M. A., Lloyd, M. T. and Jackson, W. B. (2007). Southern Calamary (*Sepioteuthis australis*) Fishery. Fishery Assessment Report to PIRSA. South Australian Research and Development Institute (Aquatic Sciences). Adelaide. SARDI Publication No. F2007/000528-2. SARDI Research Report Series No. 229. 82pp.

Steer, M. A., McGarvey, R., Carroll, J., Jackson, W. B., Lloyd, M. T. and Feenstra, J. E. (2016). Southern Garfish (*Hyporhamphus melanochir*) Fishery. Fishery Assessment Report to PIRSA Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences). Adelaide. SARDI Publication No. F2007/000720-4. SARDI Research Report Series No. 891. 75pp.

Steer, M. A., McGarvey, R., Fowler, A. J., Burch, P., Feenstra, J. E., Jackson, W. B. and Lloyd, M. T. (2012). Southern Garfish (*Hyporhamphus melanochir*) Fishery. Report to PIRSA Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences). Adelaide. SARDI Publication No. F2007/000720-3. SARDI Research Report Series No. 658. 76pp.

Svane, I. (2003). Prawn fishery bycatch and sicards: fates and consequences for a marine ecosystem. Final Report FRDC Project No. 1998/225. SARDI Aquatic Sciences, Adelaide.

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.

Triantafillos, L. (2008). Chpter 32. Cephalopods of Gulf St Vincent. 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. 437-447.

Tsikliras, A. C., Dinouli, A., Tsiros, V.-Z. and Tsalkou, E. (2015). The Mediterranean and Black Sea Fisheries at Risk from Overexploitation. PLoS ONE, 10(3): e0121188.

van Ruth, P. (2008). Chapter 23 Plankton in Gulf St Vincent and Investigator Strait. In: Shepherd, S. A., Bryars, S., Kirkegaard, I., Harbison, P. and Jennings, J. T. (Eds). Natural History of Gulf St Vincent. Royal Adelaide Society of South Australia, Adelaide, pp. 317-323.

van Ruth, P. D. (2012). Adelaide Desalination Project Plankton Characterisation Study - Phase 2. Report prepared for Adelaide Aqua. South Australian Research and Development Institute (Aquatic Sciences). Adelaide. SARDI Publication No. F2010/000378-2. SARDI Research Report Series No. 606. 40pp.

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.

Walters, C., Pauly, D. and Christensen, V. (1999). Ecospace: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. Ecosystems, 2(6): 539-554.

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

Ward, T. M., McLeay, L. J., Dimmlich, W. F., Rogers, P. J., McClatchie, S. A. M., 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(3): 191-207.

Ward, T. M., Whitten, A. R. and Ivey, A. R. (2015). South Australian Sardine (*Sardinops sagax*) Fishery: Stock Assessment Report 2015. Report to PIRSA Fisheries and Aquaculture. South Australian Research and Development Institute (Aquatic Sciences). Adelaide. SARDI Publication No. F2007/000765-5. SARDI Research Report Series No. 877. 103pp.

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). 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.

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

Zhou, S., Smith, A. D., Punt, A. E., Richardson, A. J., Gibbs, M., Fulton, E. A., Pascoe, S., Bulman, C., Bayliss, P. and Sainsbury, K. (2010). Ecosystem-based fisheries management requires a change to the selective fishing philosophy. Proceedings of the National Academy of Sciences, 107(21): 9485-9489.