## Fisheries

## SOUTH <br> AUSTRALIAN RESEARCH \& DEVELOPMENT

## Snapper (Chrysophrys auratus) Fishery



AJ Fowler, J Smart, R McGarvey, J Feenstra, F Bailleul, JJ Buss, M Drew, D Matthews, J Matthews and T Rogers

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Fishery Assessment Report to PIRSA Fisheries and Aquaculture

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## 1. EXECUTIVE SUMMARY

This stock assessment for the South Australian Snapper fishery is part of the Snapper Management, Science and Engagement Project Plan that was instigated following the changes in Snapper fishery management that were implemented in November 2019.

Stock status was assigned at the scale of biological stock for each of the Spencer Gulf / West Coast Stock (SG/WCS) and the Gulf St. Vincent Stock (GSVS). For the South East Region of the State (SE Region), data and model trends are summarised, but stock status is not provided because this regional population is a part of the Western Victorian Stock (WVS). Stock status was determined for each stock using the weight-of-evidence approach following the National Fishery Status Reporting Framework (Stewardson et al. 2018).

The stock assessments considered both fishery-dependent and fishery-independent data. The former included commercial fishery statistics, the 'general' fishery performance indicators, and population size and age structures from commercial market sampling. The fishery independent data were estimates of spawning biomass from applications of the daily egg production method (DEPM) for each of Northern Spencer Gulf (NSG) and Gulf St. Vincent (GSV).

For each of the SG/WCS, GSVS and SE Region, the SnapEst model was used to integrate key fishery and biological datasets to produce time series of 'biological' fishery performance indicators. These indicators of (i) fishable biomass, (ii) recruitment rates, (iii) harvest fractions, and (iv) egg production were a key information source on changes in stock size.

## Spencer Gulf / West Coast Stock

Estimates of total commercial catch, targeted handline and longline effort and targeted handline and longline catch per unit effort (CPUE) have been low since 2012, with most declining since 2007. In 2019, several general performance indicators were near their lowest levels and six trigger reference points were breached. These trends point to low levels of fishable biomass.

The age structures for each of Northern Spencer Gulf (NSG) and Southern Spencer Gulf (SSG) throughout the 2000s, indicated the lack of strong year classes since the late 1990s, demonstrating an extended period of poor recruitment.

The estimate of spawning biomass for NSG using the DEPM for 2019 was $177 \mathrm{t}( \pm 25$; SE), and demonstrates substantial ongoing decline from 2013.

The time series of fishable biomass from SnapEst declined from 5,350 t ( $\pm 112$; SE) in 2005 to its lowest recorded level of $468 \mathrm{t}( \pm 72$; SE) in 2020, a decline of $91 \%$. A $78 \%$ decline has occurred since 2013 when the modelled biomass was $2,106 \mathrm{t}$ ( $\pm 79$; SE).

Egg production in 2020 was the lowest on record and $2 \%$ of virgin egg production. Model outputs confirmed the low level of recruitment throughout the 2000s.

There is compelling evidence that the biomass and recruitment of the SG/WCS are at the lowest recorded levels and that the age structure of the population in NSG is truncated. Fishable biomass is depleted, recruitment is likely to be impaired and there is no evidence of stock recovery following implementation of management changes. Consequently, the SG/WCS remains classified as 'depleted' (Table 1-1).

## Gulf St. Vincent Stock

Estimates of total commercial catch, targeted handline and longline effort and targeted handline and longline CPUE increased between 2007 and 2010. They remained at nearrecord high levels until 2015 but have since declined considerably. These trends are consistent with a substantial increase in biomass followed by a rapid decline.

In 2019, the estimate of spawning biomass using the DEPM was 812 t ( $\pm 125$; SE). This represented a decline of $71 \%$ since 2014 when spawning biomass was estimated as $2,780 \mathrm{t}$ $( \pm 1,444$; SE) from a similar survey area.

Estimates of fishable biomass from SnapEst increased quickly from low values (864-1632 t) in the 1990's to a record level of $4,355 \mathrm{t}( \pm 112$; SE) in 2011, before declining by $90 \%$ to the lowest recorded level of $457 \mathrm{t}( \pm 81$; SE) in 2020. Egg production in 2020 was the lowest on record and $2 \%$ of virgin egg production. Model outputs confirmed the poor recruitment from 2010 to 2020.

There is compelling evidence that both the biomass and recruitment of the GSVS have declined since the 2018 assessment to reach the lowest levels on record. Fishable biomass is likely to be depleted and recruitment is likely to be impaired. Consequently, the GSVS is classified as 'depleted', reflecting a change from 'depleting' in 2018 (Table 1-1).

## South East Region (part of the Western Victorian Stock)

Commercial catches, effort and catch rates, increased rapidly between 2008 and 2012, but have subsequently returned to lower levels. Estimates of fishable biomass from SnapEst follow a similar trend to the commercial fishery statistics and reflect the recruitment of two strong year classes that were then subjected to high levels of fishing mortality. Estimated
fishable biomass increased slightly from $128 \mathrm{t}( \pm 44$; SE) in 2018 to $160 \mathrm{t}( \pm 70$; SE) in 2020, reflecting recruitment of the strong 2014 and 2015 year classes.

Stock status is determined for the Western Victorian Stock (WVS). This stock was classified as 'sustainable' in 2018 (Stewardson et al. 2018).

Table 1-1. Key statistics and results of this report for SG/WCS, GSVS and the South East Region in 2019.

| Stock | DEPM spawning <br> biomass (t) | Model estimated <br> stock biomass (t) | Harvest fraction <br> $(2019)$ | Stock status |
| :---: | :---: | :---: | :---: | :---: |
| Spencer Gulf / West <br> Coast Stock | $177( \pm 25 ;$ SE) | $468( \pm 73 ;$ SE) | $44 \%$ | Depleted |
| Gulf St. Vincent <br> Stock | $811( \pm 125 ;$ SE) | $457( \pm 81 ;$ SE) | $66 \%$ | Depleted |
| South East Region | - | $160( \pm 70 ;$ SE) | $20 \%$ | n.a. |

## 2. GENERAL INTRODUCTION

### 2.1. Introduction

The Snapper (Chrysophrys auratus) is a species of teleost fish in the family Sparidae. It is a large, long-lived, demersal, finfish species that is broadly distributed throughout the IndoPacific region where its extensive distribution includes the coastal waters of the southern two thirds of the Australian continental mainland as well as northern Tasmania (Kailola et al. 1993). Throughout this distribution, Snapper occupy a diversity of habitats from shallow bays and estuaries to the edge of the continental shelf across a depth range to at least 200 m . The stock structure for Snapper throughout Australian waters is complex, as there are considerable differences in the spatial scales over which populations are divisible into separate stocks (Fowler et al. 2016a; 2017). For South Australian coastal waters, a recent study indicated that there are three stocks (Figure 2-1) (Fowler 2016, Fowler et al. 2017). The cross-jurisdictional Western Victorian Stock (WVS) extends westward from Wilsons Promontory, Victoria into the south eastern waters of South Australia (SA). There are also two wholly South Australian stocks, i.e. the Spencer Gulf / West Coast Stock (SG/WCS) and the Gulf St. Vincent Stock (GSVS) (Fowler 2016, Fowler et al. 2017).

The recent study on the stock structure of Snapper was also informative about the demographic processes that underpin the replenishment of the three stocks. It indicated that each stock depends on recruitment into a primary nursery area: (i) Port Phillip Bay (PPB), Victoria for the WVS; (ii) Northern Spencer Gulf (NSG) for the SG/WCS; (iii) and Northern Gulf St. Vincent (NGSV) for the GSVS (Fowler 2016). For the South East Region (SE Region), Snapper abundance varies episodically, as fish of a few years of age migrate from PPB westwards to this region over hundreds of km (Fowler et al. 2017). This occurs in the few years following the recruitment of strong year classes to PPB. As such, it is likely to be a density dependent process that relates to inter-annual variation in recruitment of the 0+ fish (Fowler et al. 2017). The populations of Snapper that occupy the two northern gulfs are separate and self-recruiting. Each also experiences inter-annual variation in recruitment of $0+$ fish (Fowler and Jennings 2003, Fowler and McGlennon 2011), which is most likely a consequence of variable larval survivorship (Hamer et al. 2010). Each of the two northern gulfs is an important spawning and nursery area for the respective stocks. Each is a source population from which fish emigrate that then replenish regional populations in adjacent coastal waters (Fowler 2016). NSG is the source region for Southern Spencer Gulf (SSG) and most likely also for the West Coast of Eyre Peninsula (WC), whilst NGSV is the source for Southern Gulf St. Vincent (SGSV). As such, the dynamics in the regional populations of SA are primarily driven by
temporally variable recruitment into the nursery areas and subsequent emigration of fish from these source regions to adjacent regional populations (Fowler 2016).


Figure 2-1. Map of the coast of south eastern Australia, showing the stock structure for Snapper based on fish movement (Fowler 2016). The arrows indicate directions and extent of emigration of fish from three primary nursery areas in Northern Spencer Gulf, Northern Gulf St. Vincent and Port Phillip Bay, Victoria. Inset shows the broader geographic region. SG - Spencer Gulf, GSV Gulf St. Vincent, WC - west coast of Eyre Peninsula.

### 2.2. Fishery

Snapper is an iconic fishery resource in each mainland State of Australia (Kailola et al. 2003). Throughout the mid-2000s, SA was the dominant State-based contributor to the national total catches for both the commercial and recreational sectors (Fowler et al. 2016a). SA's Snapper fishery is geographically extensive and encompasses most of the State's coastal marine waters from the far west coast of Eyre Peninsula to the SE region, although the highest fishery catches have generally been taken either in Spencer Gulf (SG) or Gulf St. Vincent (GSV) (Fowler et al. 2016a, 2019; Steer et al. 2018a,b, 2020).

In SA, Snapper is a primary target species of the commercial and recreational fishery sectors (PIRSA 2013). License holders from four different commercial fisheries have access to the resource, i.e. the Marine Scalefish Fishery (MSF), the Northern Zone and Southern Zone Rock Lobster Fisheries (NZRLF, SZRLF) and the Lakes and Coorong Fishery (LCF) (PIRSA 2013). The main gear types used by commercial fishers in SA to target Snapper are handlines and longlines, since using hauling nets to take Snapper was prohibited in 1993. For local and interstate recreational fishers, Snapper has been an important species in SA's waters because of
their propensity to target the large trophy fish (Fowler et al. 2016a). Recreational fishers target Snapper using rods and lines, primarily from boats, although jetty and land-based catches do occur. Based on the most recent recreational fishing survey in 2013/14, the contributions to total catch by the commercial and recreational sectors were $62 \%$ and $38 \%$, respectively (Giri and Hall 2015, Fowler et al. 2016a).

The spatial structure of SA's Snapper fishery underwent considerable change between 2008 and 2012 (Fowler et al. 2016a, 2019). Historically, SG supported the highest catches and catch rates of Snapper, but these declined considerably, particularly through the latter 2000s. Contemporaneously, the catches and catch rates in NGSV and the SE increased to unprecedented levels (Steer et al. 2018a,b, 2020). For the three stocks these changes in fishery statistics reflect population changes associated with different, independent demographic processes relating to recruitment and adult migration (Fowler 2016, Fowler et al. 2017, 2019). From 2011 onwards, the changes in the spatial structure of the fishery and stock status caused considerable concern for the management of the fishery. This resulted in numerous management changes that were introduced to limit commercial catches and to maximise the opportunities for spawning and recruitment success. In late 2019, the management changes culminated in the implementation of stringent, spatially-explicit fishery closures. Furthermore, over the past decade, several research projects funded by the Fisheries Research and Development Corporation (FRDC) were undertaken to: (i) identify the demographic processes responsible for the observed spatial changes at the regional scale for the fishery (FRDC 2012/020, Fowler 2016), and; (ii) to develop a fishery independent index of fishable biomass (FRDC 2014/019, Steer et al. 2017).

### 2.3. Harvest Strategy

The harvest strategy for Snapper that is outlined in the current management plan relates to the changes and concerns about the challenges for managing the fishery that occurred up to 2013 (PIRSA 2013). This harvest strategy involved a watching brief until the two FRDC-funded projects described above (FRDC 2012/020, FRDC 2014/019) were completed. It did not include explicit decision rules with respect to responses to fishery status. A proposed review of the harvest strategy following finalisations of these reports was superseded by the deliberations about management of the Snapper fishery that took place throughout 2018 and 2019. This management review was in response to the recent poor status classifications (Steer et al. 2018b; Fowler et al. 2019).

A planned review of the management plan and harvest strategy will take into consideration the enhanced understanding of the biology and population dynamics of Snapper that resulted from the two FRDC projects. The harvest strategy will be revised to provide greater certainty
for sustainable management by developing explicit decision rules about management responses to fishery status based on the enhanced understanding of the biology and fishery. This is timely, due to the overall restructure of the MSF that is currently being implemented.

### 2.4. Management Regulations

In the text below there is a description of the broad approach and the historical changes to the management protocols for the commercial, recreational and charter boat sectors of the Snapper fishery. Nevertheless, since $1^{\text {st }}$ November 2019, these protocols have been superseded by the following significant spatial closures to Snapper fishing in SA's waters:

- a total Snapper fishing closure for the waters of the west coast of Eyre Peninsula, Spencer Gulf and Gulf St. Vincent until January 2023;
- $\quad$ an annual closure in the waters of the SE Region, to be applied from $1^{\text {st }}$ November until $31^{\text {st }}$ January each year until 2023. For the remainder of each year, this region will be open to fishing although a total allowable catch will apply, to be shared amongst the commercial, recreational and charter boat sectors.

The spatial closures that were imposed in November 2019 reflect the poor statuses that were assigned to the SG/WCS and the GSVS in the previous stock assessment report (Fowler et al. 2019). Their purpose is to return these Snapper fisheries to sustainable stock levels. Particularly for the SG/WCS, the 'depleted' status is the culmination of a deterioration in stock status since 2011 (Fowler et al. 2013, 2016, 2019). From then until late 2019, the management strategy was modified numerous times, attempting to redress the deteriorating stock status. Nevertheless, the strategies adopted did not result in recovery of the stock.

Prior to the fishery closures that were imposed in November 2019, regulations for the commercial sector of SA's Snapper fishery involved a suite of input and output controls (PIRSA 2013, 2014). Since 2012, there have been numerous changes to the regulations relating to these input and output controls. The four commercial fisheries with access to Snapper each have limited-entry, i.e. the numbers of fishers who can target Snapper have been limited for many years. There is a legal minimum length of 38 cm total length (TL), whilst there are also several gear restrictions. Snapper cannot be taken with fish traps, whilst the use of all nets, including hauling nets and large mesh gill nets for targeting Snapper has been prohibited since 1993. Commercial handline fishers are limited with respect to the numbers of lines and hooks per line that they can legitimately use. With respect to the use of set lines, from December 2012, the number of hooks that could be used was reduced from 400 to 200 in SG and GSV, but remained at 400 for other regions. Also, in 2012 a daily commercial catch limit of 500 kg was introduced for all South Australian waters. In December 2016, this was further reduced due to on-going concerns about the statuses of the different stocks (Fowler et al. 2016a). For
the SG/WCS, it was reduced to 200 kg with a limit of two days per trip. For GSV, the daily trip limit was reduced to 350 kg with a trip limit of two days. For the SE Region, the daily trip limit was also reduced to 350 kg , with a five-day trip limit. There is also a 50 kg by-catch trip limit for the Commonwealth-managed Southern and Eastern Scalefish and Shark Fishery.

For the recreational sector, the minimum legal length of 38 cm TL , as well as bag and boat limits apply. In December 2016, bag and boat limits were reduced in response to the recent changes in the spatial structure of the fishery and the classifications of stock status (Fowler 2016, Fowler et al. 2016a). Until that time, the bag and boat limits had differed geographically. However, from the review of the recreational fishery in 2016 (PIRSA 2016), the bag limit of 5 and boat limit of 15 fish for the size range of $38-60 \mathrm{~cm}$ TL, and bag limit of 2 fish and boat limit of 6 fish for fish > 60 cm TL, apply for all State waters. For the Charter Boat sector, from December 2018, the individual bag limit for Snapper was reduced to three small fish (38-60 cm TL ) and one large fish ( $>60 \mathrm{~cm} \mathrm{TL}$ ), with no boat limit.

Since 2000, the management regime for Snapper has involved at least one seasonal closure per year for both fishing sectors. From 2003 to 2011, this was a month-long fishery closure throughout November. From 2012, the seasonal closure for all fishing sectors was extended for several weeks until $15^{\text {th }}$ December. Furthermore, in 2013, five Snapper spawning spatial closures were implemented in the northern gulfs to extend the duration of protection of important spawning aggregations until the $31^{\text {st }}$ January, thereby conferring protection for Snapper in these areas for most of the reproductive season. The four spatial closures in NSG and one in NGSV were circular in shape with a 4-km radius from a fixed point. In December 2018, the spawning spatial closure in NGSV was removed and replaced with two new closures located in the southern gulf at Tapley Shoal and Sellicks Beach. These closures were extended to the $31^{\text {st }}$ March 2019. For SG, a new closure at Point Lowly was added to the existing four closures.

### 2.5. Recent History of Stock Statuses

In order to provide context for this stock assessment for the State's Snapper fisheries in 2020, this section provides an overview of the changes that have occurred in the fishery primarily throughout the 2000s. These include time series of the changes in stock statuses, reflecting spatial and temporal changes in fishery productivity, and augment the descriptions to the changes to the management of the fishery that were presented above.

During the 1980s, 1990s and early 2000s, SA's Snapper fishery was dominated by the SG/WCS. Nevertheless, regional analyses have indicated that around 1999, changes started to occur in the spatial structure of the Snapper fishery across the State (Fowler et al. 2016a). The relative contribution of the catches from NSG began to decline, and from 2005 to 2009
those from SSG dominated the State-wide catches. Subsequently, these also declined considerably. From 2007, the catches from the GSVS, particularly from NGSV, increased considerably from a very low level (Fowler et al. 2016a). By 2010, this stock had become the dominant contributor to the State's catch, and has subsequently remained so. Also, from 2008 to 2012, the catches from the SE Region were substantially higher than previously, but then declined back to a low level.

The significant changes in spatial structure of SA's Snapper fishery, particularly during the mid-2000s, resulted in considerable concerns for managing the fishery. These concerns were about sustainability associated with the declines in catches from the SG/WCS as well as the substantial increases in commercial fishing effort for the GSVS, particularly by the longline sector in NGSV. In response, since 2012, there have been numerous fishery management interventions, as described above. These included: (1) the introduction and subsequent reductions in daily commercial trip limits; (2) the tightening of restrictions on commercial fishing gear; (3) changes to recreational bag and boat limits; (4) extension of the duration of the Statewide annual, seasonal closure of the Snapper fishery; and (5) introduction of spatial spawning closures in both gulfs (Fowler et al. 2016a, 2019).

Despite the management interventions described above, the stock statuses have continued to deteriorate (Table 2-1). In 2013, from the regional statuses that were assigned (Fowler et al. 2013), it can be inferred that the status of the SGMCS was 'transitional depleting' and that for GSVS was 'sustainable’ (Table 2-1). In 2016, stock status was, for the first time, assigned at the scale of biological stock, recognising the recently-determined stock structure (Fowler 2016, Fowler et al. 2017). The SG/WCS was classified as 'transitional depleting' as commercial catch, effort and CPUE data declined to December 2015, reflecting poor recruitment throughout the 2000s. Again, the GSVS was classified as 'sustainable' as it continued to produce high catches and catch rates reflecting the recruitment of several strong year classes throughout the 2000s. The WVS, the cross-jurisdictional stock that spans the SE Region, was also classified as 'sustainable' based on relatively high recruitment throughout the 2000s (Hamer and Conlon 2016).

In the assessment undertaken in 2017, the classifications of 'transitional depleting' for the SG/WCS and 'sustainable' for the GSVS and WVS were maintained (Table 2-1, Steer et al. 2018a). However, in the following assessment in 2018, the status of the SG/WCS was downgraded to 'depleted' (Steer et al. 2018b). This reflected that the commercial fishery statistics to December 2017 remained at historically low levels, reflecting poor recruitment throughout the 2000s. This indicated that recruitment had become impaired (Fowler et al. 2016a). Furthermore, at this time, the low estimate of fishable biomass from the first DEPM survey in NSG in 2013 became available, a result that was consistent with the low fishery
statistics since 2012 (Steer et al. 2017). For the GSVS in 2018, declines in fishery performance indicators also suggested that the biomass had begun to decline. Nevertheless, because these recent estimates of performance indicators were considerably higher than the historical values from before 2008, the status of the GSVS was retained as 'sustainable'. Similarly, the WVS also continued to be classified as 'sustainable'.

Table 2-1. Summary of the history of stock statuses that have been assigned throughout the latter 2000s to the three Snapper stocks that occur in South Australian waters.

| Year | Spencer Gulf / <br> West Coast Stock | Gulf St. Vincent <br> Stock | Western Victorian <br> Stock |
| :--- | :--- | :--- | :--- |
| 2012 | transitional depleting | sustainable | sustainable |
| 2013 | transitional depleting | sustainable | sustainable |
| 2016 | transitional depleting | sustainable | sustainable |
| 2017 | transitional depleting | sustainable | sustainable |
| 2018 | depleted | sustainable | sustainable |
| 2019 | depleted | depleting | sustainable |

In late 2018, in response to the apparent deterioration in the statuses of both the SG/WCS and GSVS, PIRSA Fisheries and Aquaculture (PIRSA F\&A) initiated a review of the management arrangements for SA's Snapper Fishery. This consultative process involved the fishery managers, scientists and representatives of the commercial, recreational and charter boat sectors of the MSF. To ensure that this process of management review was appropriately informed, PIRSA F\&A requested that SARDI provide an assessment of Snapper stock status that included estimates of spawning biomass based on a DEPM survey to be undertaken in December 2018 for each of NSG and GSV. The resulting stock assessment in 2019 concluded that the SGMCS was still 'depleted', based on lack of recovery in fishery statistics and a low estimate of fishable biomass for NSG in 2018 (Fowler et al. 2019). Furthermore, the status of the GSVS was downgraded to 'depleting', reflecting declines in fishery statistics and a low estimate of fishable biomass from the DEPM in December 2018.

The government response to the poor levels of stock status in 2019 included the stringent fishery closures that were implemented from $1^{\text {st }}$ November 2019, but also involved the establishment of the Snapper Management Science and Engagement Project Plan. This plan involves a suite of 19 projects that were designed to ameliorate the effects of the closure on the MSF and to maintain a flow of biological information that would inform about stock status. This project plan included this stock assessment to be delivered in 2020. This stock assessment report informs about stock status up to the end of 2019 for each of the SG/WCS,

GSVS and the SE Region, based on fishery dependent and fishery independent information that provide estimates of fishery performance indicators.

### 2.6. Objectives of this report

One project of the Snapper Management Science and Engagement Project Plan was to undertake a stock assessment following Fowler et al. (2019). This assessment is described in this report. It involved several significant components: (i) to undertake regional adult sampling to provide updated information on population structure; (ii) to undertake DEPM surveys in each of NSG and GSV to provide new estimates of spawning biomass; (iii) and to re-develop the fishery assessment model SnapEst, to provide a means for integrating all fishery and biological information in the determination of stock status. This report: (i) summarises the fishery dependent and fishery independent data that were used to determine stock status; (ii) assesses the status of the resource; (iii) identifies the uncertainty associated with the assessment; and (iv) identifies future research needs. Stock status was determined using the National Fishery Status Reporting Framework (Table 2-2; Stewardson et al. 2018), which is consistent with the South Australian fisheries harvest strategy policy (PIRSA 2015).

Table 2-2. Terminology for the status of key Australian fish stocks reports (reproduced from Stewardson et al. 2018).

|  | Stock status | Description | Potential implications for <br> management of the stock |
| :--- | :--- | :--- | :--- |
| Sustainable | Stock for which biomass (or biomass proxy) is at a level sufficient to <br> ensure that, on average, future levels of recruitment are adequate <br> (i.e. recruitment is not impaired) and for which fishing mortality (or <br> proxy) is adequately controlled to avoid the stock becoming <br> recruitment impaired | Appropriate management is <br> in place |  |
| Depleting | Biomass (or proxy) is not yet depleted and recruitment is not yet <br> impaired, but fishing mortality (or proxy) is too high (overfishing is <br> occurring) and moving the stock in the direction of becoming <br> recruitment impaired | Management is needed to <br> reduce fishing pressure and <br> ensure that the biomass <br> does not become depleted |  |
| Recovering | Biomass (or proxy) is depleted and recruitment is impaired, but <br> management measures are in place to promote stock recovery, and <br> recovery is occurring | Appropriate management is <br> in place, and there is <br> evidence that the biomass <br> is recovering |  |
| Depleted | Biomass (or proxy) has been reduced through catch and/or non- <br> fishing effects, such that recruitment is impaired. Current <br> management is not adequate to recover the stock, or adequate <br> management measures have been put in place but have not yet <br> resulted in measurable improvements | Management is needed to <br> recover this stock; if <br> adequate management <br> measures are already in <br> place, more time may be <br> required for them to take <br> effect |  |
| Undefined | Not enough information exists to determine stock status | Data required to assess <br> stock status are needed |  |
| Negligible | Catches are so low as to be considered negligible and inadequate <br> information exists to determine stock status | Assessment will not be <br> conducted unless catches <br> and information increase |  |

## 3. METHODS

### 3.1. Sources of Information

This stock assessment used a weight-of-evidence approach for the determination of stock status that considered both fishery dependent and fishery independent information (Fowler et al. 2019). The fishery-dependent data were: (i) commercial fishery statistics; (ii) recreational fishery data; and (iii) population size and age structures determined through commercial market sampling or targeted fishing by commercial fishers. These data were considered at several spatial scales, as appropriate, i.e., the State-wide scale, the scale of stocks, or at the regional population level. The fishery-independent data were estimates of spawning biomass for regional populations in SG and GSV using the daily egg production method (DEPM), based on surveys that were undertaken in December 2019 for SG and January 2020 for GSV. The need to use this approach to estimate spawning biomass of Snapper in the SA fishery has several sources. Firstly, the imposition of daily commercial trip limits in the fishery since 2012 compromised the relationship between fishery statistics, including catch per unit effort (CPUE), and stock biomass. Secondly, in this fishery, there are other issues with the relationship between CPUE and biomass that affect the usefulness of the former as an indicator of the latter. These include complexities associated with the increase in 'effective' effort over time associated with technology creep. Also, there is an issue of 'hyperstability' in the measure of CPUE (Fowler and McGlennon 2011, Fowler et al. 2019). The methods for applying the DEPM for Snapper were developed through an FRDC-funded project (Steer et al. 2017), which were then used as part of the previous stock assessment (Fowler et al. 2019).

### 3.2. Commercial Fishery Statistics

Since July 1983, commercial fishers in SA's MSF have been required to submit a monthly catch return that relates details of their catches and effort for the preceding month. These historical data now constitute the Marine Scalefish Fisheries Information System (MSFIS). For this report, the fishery data for Snapper from the MSF, NZRLF and SZRLF were extracted from the MSFIS and were combined with similar data from the Lakes and Coorong Fishery Information System. Commonwealth Snapper catches were obtained from AFMA. The data for the 36 -year period of 1984 to 2019 were considered in this assessment. Annual estimates by calendar year for total catch, effort and CPUE at the State-wide scale were calculated, differentiating the contributions of the two main gear types of handlines (HL) and longlines (LL). For the two stocks of SG/WCS and GSVS as well as the SE Region, the annual estimates of total catch, targeted catch, effort and CPUE by gear type (HL, LL) were calculated. The numbers of fishers taking and targeting Snapper by gear type are also presented. Furthermore, because of the imposition of daily trip limits for Snapper in 2012, two additional
fishery performance indicators were prescribed in the management plan (PIRSA 2013), i.e. the proportions of daily fishing trips for which both the HL and LL catches were $\geq 250 \mathrm{~kg}$ (PIRSA 2013). Note that the nominated catch amount has been reduced to $\geq 200 \mathrm{~kg}$, reflecting the reduction in the daily trip limit for the SG/WCS that was implemented in December 2016 (Fowler et al. 2019). The calculation of the annual estimates of the two performance indicators (Prop200kgTarHL, Prop200kgTarLL) used daily catch data from the commercial sector that are available in the MSFIS by calendar year from 2004 to 2019. For each year, only the targeted catch data from February to October were considered so as to remove the influence of the seasonal closure from the data (PIRSA 2013).

### 3.3. Regional Estimates of Size and Age Structures

Since 2000, annual estimates of size and age structures have been developed for Snapper in order to inform about the demographic processes that operate at the regional spatial scale. These have been based on market sampling of commercial catches, primarily at the SAFCOL fish market in Adelaide, but also augmented with estimates of size and age from occasional sampling trips to regional areas and biological data collected on research cruises. All such sampling has conformed to a two-stage sampling protocol (Fowler et al. 2016a). Fishery catches were accessed at the fish market and individual fish were measured for caudal fork length (CFL) to provide size information. When possible, further biological data were collected for a sub-sample of fish. Such fish were measured for CFL, weighed, sexed and stage of reproductive maturity was determined. They were dissected to remove the otoliths that were later used to determine fish age using an established ageing protocol (Fowler et al. 2016a). The fisher's details were recorded so that later on information about where the fish were caught and the capture method could be accessed from the submitted catch return.

Because of the closure to SA's Snapper fishery in November 2019, in early 2020 another form of sampling was used to access samples of fish to provide size and age information. Several commercial fishers were engaged to fish for and provide whole samples of Snapper from particular regions. Such sampling was done in NGSV, SGSV and SSG in January and February 2020. NSG had previously been adequately sampled by market sampling and by the DEPM research cruise in December 2019. Recent samples from the SE Region were obtained from the SAFCOL fish market after this regional fishery was reopened on the $1^{\text {st }}$ February 2020. Samples from the west coast of Eyre Peninsula were not collected in late 2019 or early 2020. Sample sizes for 2019 and 2020 (combined) were 592 for the SG/WC stock (NSG 280, SSG - 267, WC - 45), 614 for the GSV stock (NGSV - 259, SGSV - 355) and 533 for the SE region.

From the size and age data collected throughout each year, annual estimates of size and age structures were developed for regional populations, using the methods of McGlennon et al. (2000). Furthermore, annual estimates of weight structures were also calculated which show the distribution of the biomass harvested in each year, across the different size classes. These size, weight and age structures are presented at the regional spatial scale from 2008 onwards. For comparison with those for the period of 2000 to 2007 refer to the figures in Section 6.1 in the appendix of Fowler et al. (2016a). Where size, weight or age structures are not presented for a particular year and region, there were insufficient data available for their development. For considering the annual size and weight structures, four size categories are recognized: (i) 'small' fish in the $30-39.9 \mathrm{~cm}$ CFL range; (ii) 'medium’ fish that were $40-59.9 \mathrm{~cm}$ CFL; (iii) 'large' fish that were $60-79.9 \mathrm{~cm}$ CFL; (iv) and 'very large' fish that were $\geq 80 \mathrm{~cm}$ CFL.

### 3.4. Regional Estimates of Spawning Biomass

Estimates of spawning biomass of Snapper for two regions were developed using the DEPM (Lasker 1985). This method estimates the biomass of the spawning component of the population by empirically combining the estimates of the density of pelagic eggs and the estimates of a range of adult fish parameters that are all obtained from an intensive field sampling program, following the methods developed by Steer et al. (2017).

For this stock assessment, estimates of spawning biomass were determined for the northern and central parts of Spencer Gulf (northwards from a line across the gulf from Port Victoria to Tumby Bay), and also throughout GSV and Investigator Strait (IS). The estimates of egg density for each region were determined through a multi-step process that involved: (i) a plankton survey; (ii) the sorting of plankton samples to remove fish eggs that could potentially be from Snapper; and (iii) identifying and quantifying the Snapper eggs. The plankton survey in SG was done from $4^{\text {th }}$ to $15^{\text {th }}$ December 2019 during which 280 stations were sampled (Figure 3-1). This involved an extra 60 stations compared to those sampled in 2018 (Fowler et al. 2019). Plankton sampling was done throughout GSV and IS from 13-19 January 2020, during which a total of 264 stations were sampled. This survey included an extra 57 stations compared with the survey in 2018 (Fowler et al. 2019). At each station, an oblique plankton tow was done using paired bongo nets with $500 \mu \mathrm{~m}$ mesh. The net was deployed over the stern as the vessel proceeded at $<4 \mathrm{kn}$. The net was lowered to within 5 m of the bottom and then retrieved, sampling as it descended and as it was retrieved. The tide and wind conditions determined the distance travelled and volume of water sampled varied at different stations. These parameters were then estimated for each station based on readings from flowmeters (General Oceanics ${ }^{\top M} 2030$ ) that were located in the centre of the mouth of each net. On retrieval, the catches of plankton from the two nets were washed into a one litre jar and then
the sample was fixed in $99 \%$ ethanol. Later, the plankton samples from all stations were sorted for the separation of fish eggs that could potentially be from Snapper, based on their size and morphological characteristics (Steer et al. 2017). Then, for the extracted eggs, an identification process was undertaken to differentiate the Snapper eggs. This involved the application of the in situ hybridization (ISH) molecular technique that uses the mitochondrial 16S ribosomal RNA gene as a target for a specific oligonucleotide probe (Steer et al. 2017). In 2019, the hybridization of Snapper eggs was less pronounced than in previous assessments (Steer et al. 2017; Fowler et al 2019). Therefore additional procedures were undertaken to identify Snapper eggs based on their morphological characteristics and the size range of 775 - 900 $\mu \mathrm{m}$.

During the period of the plankton survey in each region, the adult fish were sampled through targeted fishing in order to provide estimates of the adult parameters. These samples were collected either through fishery-independent sampling from the RV Ngerin or by contracted commercial fishers. These samples provided data on fish size, weight, age, sex and stage of maturity. For those ovaries from GSV and IS for which there was some ambiguity about the stage of reproductive activity and whether spawning had occurred recently, histological analysis of the ovary structure was undertaken. The data from the macroscopic and microscopic analyses of ovaries were used to determine the spawning fraction. This information was further used to refine spawning fraction estimates for SG where histological analysis did not occur.

## Spawning area (A)

The spawning area $(A)$ in both surveys was determined using geostatistical kriging using the geographic information system 'ArcGIS'. This method interpolated the georeferenced point data (eggs.m ${ }^{-2}$ at each station) to predict the intermediate values through a Gaussian process governed by prior covariances. A minimum egg density of 0.1 eggs. $\mathrm{m}^{-2}$ was used to define the outer boundary of the spawning activity for each survey and determine $A$.


Figure 3-1. DEPM Survey areas for 2019/20. Locations of sampling stations throughout Spencer Gulf and Gulf St. Vincent. Black dots indicate the stations sampled in previous surveys. Crosses indicate the additional stations sampled during the 2019/20 surveys.

## Mean daily egg production ( $P_{0}$ )

The stage-based egg density estimator developed by McGarvey et al. (2018) was used to determine $P_{o}$. This method is an improved approach for demersal species such Snapper that spawn with much lower egg densities than small pelagic species. The advantage of this approach is that egg mortality $(Z)$ is specified a priori rather than estimated (McGarvey et al., 2018). However, as $Z$ is specified a priori sensitivity analyses were conducted over a range of
$Z\left(0.2-0.6 \mathrm{yr}^{-1}\right)$ to ensure that the $P_{0}$ estimate was not influenced by this value. A $Z$ of 0.4 yr ${ }^{1}$, was used to calculate $P_{0}$ in all subsequent results.

## Spawning fraction (S)

Spawning fraction was determined for GSV using gonad histology samples to determine the presence of post ovulatory follicles (POFs) that indicate whether a female fish has spawned recently. As no samples were available for histology in SG, inferences on the number of GSV spawning fish in a given macroscopic stage were used to estimate spawning fraction in SG. This introduces some imprecision into the $S$ estimate for $S G$ which is examined in the sensitivity analyses presented in Appendix 7.4. Macroscopic gonad stages for Snapper correspond to: stage $1=$ immature, stage 2 = developing, stage 3 = developed, stage $4=$ gravid and stage 5 = regressing (Saunders et al. 2012). All stage 1 and 2 females are not in spawning condition while females at stages 3 and 4 can be spawning on a given day. Stage 5 fish are those that are mature, but have finished spawning for the season. The proportion of spawning fish in stages 3 and 4 for GSV that had POFs was used to determine how many females in each respective gonad stage were spawning for the corresponding SG reproductive stages. Spawning fractions $(S)$ for each gulf were then calculated as a ratio estimate over all sampled females:

$$
S=\overline{N^{\mathrm{fem}, \mathrm{sp}}} / \overline{N^{\mathrm{fem}}} \cdot[\text { Equation 1] }
$$

Where, $\overline{N^{\text {fem }}}$ is the mean number of mature females across samples and $\overline{N^{\text {fem,sp }}}$ is the mean number of spawning female fish across samples. Standard errors were calculated using a mean ratio estimator.

## Sex ratio (R)

The weight of mature (Gonad stage >=2) males and females in each sample were used to estimate the sex ratio $(R)$ according to Equation 2:

$$
R=\overline{W^{\text {fem }}} / \overline{W^{\text {tot }}} \cdot \text { [Equation 2] }
$$

Where $\overline{W^{\text {fem }}}$ and $\overline{W^{\text {tot }}}$ are the respective mean weights of mature females and all mature fish across samples. Standard errors were determined using a mean ratio estimator.

Female weight ( $p_{w}^{\mathrm{N}}$ )
To account for variations in female body weight that are not normally distributed, adult samples were grouped into twenty six weight classes ranging from < $500-1300$ grams. The proportion
of fish in each weight bin was then included as an input into the size-based spawning biomass equation as described in McGarvey et al. (in review). A multinomial error distribution was applied to determine the uncertainty for the proportion of fish in each weight bin (McGarvey et al. in review). The midpoints of these weight bins ( $\breve{w}_{w}$ ) are included in the spawning biomass estimator.

## Batch Fecundity ( $F_{w}$ )

The relationship between female weight ( $W$ ) and batch fecundity was determined using an allometric function with residual error that increases with $W$ and used to estimate the batch fecundities of mature females in all samples. The allometric function for fecundity against weight was taken as a continuous variable: where $\alpha$ and $\beta$ are allometric coefficients:

$$
\hat{F}(W)=\alpha \cdot W^{\beta} \cdot[\text { Equation } 3]
$$

A maximum likelihood estimator that accounted for heteroscedasticity in the spread of the residuals was used in the model fit to estimate the parameters $\alpha$ and $\beta$ (McGarvey et al. in review). Weight-dependent batch fecundity estimates were calculated for the mid-point of each weight bin using this allometric relationship with normally distributed error

$$
F_{w}=\alpha \cdot \breve{w}_{w}{ }^{\beta} . \quad[\text { Equation 4] }
$$

These are also included into the size-dependent estimation of spawning biomass (McGarvey et al. in review).

## Spawning biomass (SB)

For Snapper, the estimates of the various parameters are combined empirically to estimate spawning biomass (SB) using the following equation:

$$
S B=\left(\frac{P_{0} \cdot A}{S \cdot R \cdot \sum_{w=1}^{\omega} F_{w} \cdot p_{w}^{\mathrm{N}}}\right) \cdot \sum_{w=1}^{\infty} p_{w}^{\mathrm{N}} \cdot \breve{w}_{w}
$$

## [Equation 5]

Where $P_{0}$ is mean daily egg production, $A$ is spawning area, $S$ is spawning fraction, $R$ is sex ratio in weight, $w$ is weight class number, $\omega$ is the number of weight classes, $\breve{w}_{w}$ is each weight-class midpoint, $F_{w}$ is the fecundity at $\breve{w}_{w}$ and $p_{w}^{\mathrm{N}}$ is the proportion of females in weight class w (Steer et al. 2017, McGarvey et al. 2018, McGarvey et al. (in review)).

The variance of each of these quantities is estimated using delta approximation, where the overall variance of the spawning biomass estimate is written as:

$$
\begin{aligned}
& V(S B)=\frac{1}{R^{4} \cdot S^{4}\left(\sum_{w=1}^{\infty} F_{w} \cdot p_{w}^{\mathrm{N}}\right)^{4}} \cdot \\
& \left\{\begin{array}{l}
{\left[P_{0}^{2} \cdot R^{2} \cdot S^{2} \cdot V(A)+A^{2} \cdot R^{2} \cdot S^{2} \cdot V\left(P_{0}\right)+A^{2} \cdot P_{0}^{2} \cdot S^{2} \cdot V(R)+A^{2} \cdot P_{0}^{2} \cdot R^{2} \cdot V(S)\right] \cdot} \\
\left\{\left(\sum_{w=1}^{\infty} F_{w} \cdot P_{w}^{\mathrm{N}}\right)^{2} \cdot\left(\sum_{w=1}^{\infty} p_{w}^{\mathrm{N}} \cdot \breve{w}_{w}\right)^{2}\right\}+ \\
A^{2} \cdot P_{0}^{2} \cdot R^{2} \cdot S^{2} \cdot\left\{\left(\sum_{w=1}^{\infty} V\left(F_{w}\right) \cdot\left(p_{w}^{\mathrm{N}}\right)^{2}\right) \cdot\left(\sum_{w=1}^{\infty} p_{w}^{\mathrm{N}} \cdot \breve{w}_{w}\right)^{2}\right\}+ \\
A^{2} \cdot P_{0}^{2} \cdot R^{2} \cdot S^{2} \cdot\left\{\sum_{i=1}^{\infty}\left[V\left(p_{i}^{\mathrm{N}}\right) \cdot\left(\breve{w}_{i} \cdot \sum_{w \neq i}^{\infty} F_{w} \cdot p_{w}^{\mathrm{N}}-F_{i} \cdot \sum_{w \neq i}^{\infty} p_{w}^{\mathrm{N}} \cdot \breve{w}_{w}\right)^{2}\right]\right\}
\end{array}\right.
\end{aligned}
$$

All DEPM calculations were produced using the 'DEPM' package in the ' R programming environment' (Smart et al. 2020; R core Team 2019).

Daily egg production methods are known to have large imprecision which results from the combination of several parameters, that are themselves imprecise. While it is acknowledged that DEPM estimates are considered unbiased and are demonstrably capable of detecting changes in biomass, this imprecision requires sensitivity analyses to determine which parameters could influence estimates of biomass if determined inaccurately. A sensitivity analysis was performed using the 2019 surveys for the three most influential parameters in the DEPM analysis: egg density ( $P_{0}$ ), spawning area $(A)$ and spawning fraction $(S)$. This analysis is presented in Appendix 7.4.

### 3.5. Stock assessment model - SnapEst

The SA Snapper fishery stock assessment model SnapEst was developed with FRDC support (McGarvey and Feenstra 2004) as a dynamic, spatial, age- and length-structured model. SnapEst integrates multiple data sources, biological and fishery-derived, to estimate four model-based fishery performance indicators that are specified in the management plan (PIRSA 2013; Table 3-1). For the application of SnapEst to this stock assessment a number of significant changes were made to the version of the model that had been applied most recently (Fowler et al. 2016). One considerable change related to the spatial structure of the model, in order to conform to the recently-developed understanding of stock structure (Fowler et al. 2017). Previously, the model conformed to a dis-aggregated spatial structure which involved separate model regions for the northern and southern parts of the two gulfs. Here, for the updated model, the spatial structure was modified to involve the three spatial components of SG/WCS, GSVS and the SE Region, which were assumed to be independent, i.e. there was no movement between them.

Seven data sets were used as input to the SnapEst model: (1) commercial catch totals; (2) handline CPUE as an index of abundance; (3) recreational catch data from the telephone and diary surveys undertaken in 2000/01, 2007/08 and 2013/14; (4) charter boat catch totals from charter boat logbooks collected since September 2005/06; (5) commercial catch-at-age proportions; (6) commercial length-frequency samples; and (7) DEPM estimates of spawning biomass from summers of 2013 (SG), 2014 (GSV), 2018 (both gulfs), and 2019/20 (both gulfs).

For this revised version of SnapEst, handline CPUE was used as the preferred fishery-based index of Snapper abundance. This is because, over time, handline fishing practices have changed much less than those relating to longlines, whilst also longline catch rates show evidence of hyperstability (Fowler and McGlennon 2011, Fowler et al. 2019). Model harvestable biomass was fitted to HL CPUE from the start of the model time frame in October 1983 up to and including summer 2013 for SG and up to and including summer 2014 for GSV.

SnapEst runs on a half-yearly time step, fitting to data from each summer (October-March) and winter (April-September) since October 1983. For the SE Region, the model ran from October 1983 to September 2019, whilst for SG/WCS and the GSVS the model was extended by a half-year to include the most recent summer (October 2019 to March 2020), which allowed model fitting to the most recent DEPM biomass estimates.

The Snapper model employs the slice-partition method to estimate fish population numbers by both age and length (McGarvey et al. 2007). Model catch was incorporated from six fishery sectors (handline, longline, hauling net, all other commercial gears combined, charter boats, and other recreational). Target type was not differentiated and non-target catches of Snapper
were, for the most part, low. Furthermore, the model did not attempt to discern a stockrecruitment relationship and recruitment for each stock was freely estimated for each year. Instantaneous natural mortality was set to $M=0.05$ (consistent with previous versions of Snapper stock assessment models in South Australia). Full details of the model equations, fishery dynamics and likelihood function, are given in Appendix 7.1.

SnapEst integrates the seven input data sets to produce maximum likelihood estimates of four yearly indicators of fishery performance by stock: (i) fishable biomass (all Snapper above legal size); (ii) recruitment number; (iii) harvest fraction; and (iv) egg production. Annual fishable biomass is the average of the two half-yearly estimates. Yearly recruit numbers (i.e. numbers reaching 2 years of age at the start of the model half year that commences in October) are dated by cohort year class from the summer (1 January) when spawned. Harvest fraction is the yearly catch divided by the annual fishable biomass. The proportion of pristine egg production is the estimated annual number of eggs produced (assuming a $50: 50$ sex ratio, a fecundity versus length formula, and all Snapper age 2 years and older assumed to be mature) divided by a single estimate of 'pristine', i.e. pre-fishery egg production. The latter was obtained by running the model for 100 years longer with catch set to zero. The constant yearly recruitment assumed for this pristine egg production run was set to the average over 19822009, which covers years prior to the more recent gulf stock declines. More recent years are excluded since they potentially reflect recruitment reduced by high exploitation and so not be typical of pre-fishing levels.

Besides the changes to the spatial structure of the model, a further important change was that it is now catch-conditioned, rather than effort-conditioned. This meant that: (i) the reported catch totals by weight (or for charter and recreational catch, survey-estimated totals by number) are removed without error from the population in each time step; and (ii) the model fits to catch rate rather than to total catch. A further change to the model was that, for each of the two gulfs, the fishery-independent estimates of absolute biomass that were provided by the three DEPM surveys were fitted to for those summers during which the DEPM surveys were undertaken.

Catch rates for Snapper are known to be uncertain as an index of abundance. Catch conditioning permits the model to use only the measure of CPUE (here, from HL ) and only over the years that were deemed informative (up to the start of DEPM after which CPUE was not used in the two gulfs). In a catch-conditioned fishery model, effort data are not used when they are absent or highly uncertain such as for the recreational sector, or when the catch rate source is not thought to be reliable as a relative stock index (e.g., Snapper LL and charter boats). A Pope approximation was used in the catch-conditioning (Pope 1972).

In fitting to DEPM biomass, it was necessary to account for the fact that the model regions extended further than the area covered by the DEPM surveys. Only commercial catch data by MFA block exist to quantify the spatial distribution of Snapper within each model region. The model biomass fitted to DEPM was therefore scaled to approximate the DEPM biomass surveyed using the proportion of catch in the DEPM-surveyed MFA blocks divided by catch in each entire model region. The half-yearly summer catches spanning the first four DEPM surveys were used, namely 2013 (SG) or 2014 (GSV) and 2018 (both gulfs). For fitting to the results from the DEPM surveys in 2019 and 2020, because the two gulfs were closed to fishing for all summer months except October 2019, the 2018 spatial catches by MFA block (Oct18Mar19) were also used to construct the 2019/20 survey spatial catch proportions.

Because the spatial structure of SnapEst was modified for this stock assessment, two basic body size relationships, length-at-age, and weight-at-length, had to be re-estimated at the appropriate spatial scale for incorporation into the model. Weight-at-length was found to be similar amongst the three stocks, so a single relationship was derived that was applied to all three stocks. Alternatively, estimates of length-at-age varied amongst stocks and so different growth relationships were derived for the three stocks. For each, the von Bertalanffy formula was fitted to the estimates of length-at-age, and parameters of the formula were estimated by maximum likelihood. Details of the analytical process and the equations applied are provided in Appendix 7.2.

One important aspect of integrated fishery modelling is how much weighting to place on each data source. For the two gulf model stocks i.e. SG/WCS and GSVS, one important choice is the weighting to apply to the fishery-independent DEPM biomass estimates. HL CPUE, age composition and length moments are the additional data sources. For the baseline model runs in the gulfs, presented in Section 4.4., relatively stronger weighting was placed on DEPM and age composition data, and relatively lower weighting on length moments, while HL CPUE were not fitted at all after 2014. To quantify the impact of different choices for these relative data weightings, sensitivity analysis, i.e. multiple alternative runs of the model under different assumptions for these weightings were undertaken. The detailed results from these sensitivity analyses are presented in Appendix 7.3.

### 3.6. Assessment of Fishery Performance

A series of general and biological fishery performance indicators and associated reference points were used to assess stock status, which primarily relate to the fishery dependent data (PIRSA 2013). These were considered for each of the two main gear types of HL and LL at the scale of stock for the SG/WCS and GSVS and for the SE Region. The general performance indicators (PIRSA 2013) were considered in each case. These were: (i) total catch; (ii) targeted
handline effort; (iii) targeted handline CPUE; (iv) targeted longline effort; and (v) targeted longline CPUE. The estimate for each parameter in 2019 was compared against those calculated for the reference period of 1984 to 2018, and assessed using several trigger reference points (Table 3-1). The estimates of Prop200kgTarHL and Prop200kgTarLL for 2019 were compared against those from the reference period of 2004 to 2018, using the same trigger reference points that are used for the general performance indicators (Table 3-1).

Table 3-1. Performance indicators used to monitor the performance of South Australia's Snapper fisheries as prescribed in the MSF Management Plan (PIRSA 2013). Biological (B) and General (G) indicators and whether a primary $(P)$ or secondary $(S)$ indicator are identified.

| Performance | Type | P or S | Trigger Reference Point |
| :---: | :---: | :---: | :---: |
| Total catch | G | S | $3^{\text {rd }}$ lowest/3 $/{ }^{\text {rd }}$ highest |
|  |  |  | Greatest interannual change ( $\pm$ ) |
|  |  |  | Greatest 3-year trend ( $\pm$ ) |
|  |  |  | Decrease over 5 consecutive years? |
| Targeted handline effort | G | P | $3^{\text {rd }}$ lowest/3 ${ }^{\text {rd }}$ highest |
|  |  |  | Greatest interannual change ( $\pm$ ) |
|  |  |  | Greatest 3-year trend ( $\pm$ ) |
|  |  |  | Decrease over 5 consecutive years? |
| Targeted handline CPUE | G | P | $3^{\text {rd }}$ lowest//3 ${ }^{\text {rd }}$ highest |
|  |  |  | Greatest interannual change ( $\pm$ ) |
|  |  |  | Greatest 3-year trend ( $\pm$ ) |
|  |  |  | Decrease over 5 consecutive years? |
| Targeted longline effort | G | P | $3^{\text {rd }}$ lowest/3 ${ }^{\text {rd }}$ highest |
|  |  |  | Greatest interannual change ( $\pm$ ) |
|  |  |  | Greatest 3-year trend ( $\pm$ ) |
|  |  |  | Decrease over 5 consecutive years? |
| Targeted longlineCPUE | G | S | $3{ }^{\text {rd }}$ lowest/3 ${ }^{\text {rd }}$ highest |
|  |  |  | Greatest interannual change ( $\pm$ ) |
|  |  |  | Greatest 3-year trend ( $\pm$ ) |
|  |  |  | Decrease over 5 consecutive years? |
| Prop200kgTarHL |  | P | $3^{\text {rd }}$ lowest/ $/ 3^{\text {rd }}$ highest |
|  |  |  | Greatest interannual change ( $\pm$ ) |
|  |  |  | Greatest 3-year trend ( $\pm$ ) |
|  |  |  | Decrease over 5 consecutive years? |
| Prop200kgTarLL |  | S | $3^{\text {rd }}$ lowest/3 ${ }^{\text {rd }}$ highest |
|  |  |  | Greatest interannual change ( $\pm$ ) |
|  |  |  | Greatest 3-year trend ( $\pm$ ) |
|  |  |  | Decrease over 5 consecutive years? |
| Fishable biomass | B | P | $3-\mathrm{yr}$ ave is +/-10\% of previous 3-yr ave |
| Harvest fraction | B | P | above 32\% (int. standard) |
| Egg production | B | S | <20\% of pristine population |
| Recruitment | B | S | 3 -yr ave is +/-10\% of historical mean |
|  |  |  | 3 -yr ave is $+/-10 \%$ of previous 6-yr ave |
| Age composition | B | P | Prop >10yrs <20\% of fished population |

There are five biological performance indicators: fishable biomass; egg production; harvest fraction; recruitment; and age structures (Table 3-1). The first four are yearly time-series of output parameters from the SnapEst model described in the section above, whilst age
structures are catch proportions by age, derived from the market sampling. The SnapEst model performance indicators were compared against their trigger reference points for each of SG/WCS, GSVS and the SE Region and interpreted using the trigger reference points indicated in Table 3-1. For age structures, the trigger reference point is structured around the operational objective of maintaining the proportion of the fish older than 10 years of age at above $20 \%$ of the fished population.

## Allocation

The Fisheries Management Act 2007 states that the Management Plan must specify the allocation of the resource amongst the various sectors within the MSF. Allocated shares were derived from the catch data collected in 2007/08, when a State-wide recreational fishery survey was done (Jones 2009). For Snapper, there are three trigger limits for the assessment of all allocations amongst fisheries and sectors (Table 3-2). The first trigger limit (Trigger 1) relates to the allocated shares amongst the commercial fisheries, the recreational fishery and the charter boat sector (PIRSA 2013). Since there are no new recreational fishery data, this assessment was not done here (see Fowler et al. 2016a). The remaining two trigger limits (Triggers 2 and 3) relate specifically to the allocation of shares amongst the different commercial fisheries, and so can be assessed on an annual basis (Table 3-2). The trigger limits have been set at levels that are commensurate with the initial allocation and allow for variability in catches. Trigger 2 relates to exceeding the commercial sector allocation by the relevant percentage in three consecutive years or in four of the previous five years. Trigger 3 relates to exceeding the commercial sector allocation by the nominated percentage in any one year.

Table 3-2. Allocation of Snapper catch shares among the sectors as prescribed in the MSF Management Plan (PIRSA 2013).

|  | MSF | SZRLF | NZRLF | LCF |
| :---: | :---: | :---: | :---: | :---: |
| Commercial allocation | 97.5 | 1.78 | 0.68 | 0.04 |
| Trigger 2 (\%) | na | 2.68 | 1.3 | 0.75 |
| Trigger 3 (\%) | na | 3.58 | 2.0 | 1.0 |

## 4. RESULTS

### 4.1. Commercial Fishery Statistics

## State-wide

Estimates of total State-wide commercial catch of Snapper show cyclical variation, with the cycles typically encompassing a number of years (Figure 4-1). Since 2003, State-wide catch increased to a record level of $1,035 \mathrm{t}$ in 2010, before declining by $75.7 \%$ to 252 t in 2019, the $2^{\text {nd }}$ lowest recorded. Historically, HLs were the most significant gear type, whose catches largely accounted for the cyclical variation in total catch until 2008. The proportional contribution of LLs to total catch increased considerably between 2005 and 2010, becoming the dominant gear type. Both HL and LL catches have declined considerably since 2010.

Between the mid-1980s and 2008 there was a long-term, gradual declining trend in total commercial fishing effort that produced catches of Snapper (Figure 4-1). This was followed by a period of elevated fishing effort between 2009 and 2012 that related to the increase in LL effort. However, since 2010, LL effort has declined, complementing the on-going declining trend in HL effort since 2002. As such, the total fishing effort of 4,336 fisher-days in 2019 was the lowest recorded since 1984. State-wide HL CPUE showed cyclical variation, superimposed on a long-term increasing trend. However, since 2007 it has decreased considerably, concomitant with the emerging dominance of LL fishing. In contrast, LL CPUE increased considerably between 2004 and 2015, before declining in each year between 2016 and 2019.

The total number of fishers from across all four commercial fisheries who reported taking Snapper, declined consistently from 403 in 1984 to 244 in 2000. It then stabilised for a number of years before declining from 260 in 2010 to 163 fishers in 2019. The numbers who targeted Snapper varied similarly and fell from 201 in 2009 to 121 in 2019.

In 2019, the commercial catch was dominated by the MSF which contributed $97.7 \%$ of the total (Figure 4-2). The SZRLF accounted for most of the remaining catch. Catches by Commonwealth fishers were $<2 \%$ of the total catch in 2019.

## Stock

The relative contributions of the three stocks to total State-wide annual catches have changed considerably over time, particularly relating to the change in spatial structure of the fishery that occurred between 2008 and 2012 (Figure 4-2). The SG/WCS provided the highest proportions of annual catches up to 2009, after which they declined and fell to their lowest levels between 2012 and 2019. The catches from the GSVS were generally low until around 2004 after which they increased gradually for a few years before accelerating between 2007 and 2010. This
stock became and has subsequently remained the main contributor to the State-wide catch, up to 2019. The catches from the SE Region also increased rapidly between 2007 and 2010, before they declined back to a low level in 2017. They have increased marginally in 2018 and 2019.


Figure 4-1. Snapper. Long-term trends in: (A) total commercial catch by the main gear types (HLs and LLs), estimates of recreational catches and gross production value; (B) total commercial effort for HLs and LLs; (C) total catch per unit effort (CPUE) for HLs and LLs; and (D) the number of active commercial licence holders taking or targeting Snapper.


Figure 4-2. Snapper. (A) Distribution of commercial catch in 2019. Long term trends in: (B) the annual distribution of commercial catch among biological stocks, (C) months of the year t ); the proportion of catch distributed among the commercial sectors in 2019 (D); and among the fishery sectors in 2013/14, based on data from the latest recreational fishing survey (Giri and Hall, 2015).

## Spencer Gulf / West Coast Stock

Annual catches from the SG/WCS have varied cyclically with peaks in 1990, 2001 and 2007. The latter year produced the highest catch of 618.3 t (Figure 4-3). From 2007 to 2012, annual catches fell considerably, and have subsequently remained relatively stable at a low level. In 2019, the lowest recorded catch from this stock of 61.7 t was taken.

Targeted HL catches have varied over time. The highest of 516.4 t was taken in 2001, which has since fallen to the lowest of only 26.1 t in 2019 (Figure 4-3). Targeted HL effort increased between 1984 and 2002 to the highest level of 5,142 fisher-days. Since then, it has declined to the lowest level of 459 fisher-days in 2019. Targeted HL CPUE has varied cyclically, but showed a long-term increasing trend to 2011, which peaked in 2007 at $138.1 \mathrm{~kg} . f i s h e r-d a y ~^{-1}$, but in 2012 declined steeply to 63.8 kg.fisher-day ${ }^{-1}$, before dropping to 48.1 kg .fisher-day ${ }^{-1}$ by 2018. It subsequently increased to 56.8 kg.fisher-day $^{-1}$ in 2019. The numbers of licence holders who took and targeted Snapper with HLs declined slowly through the 1980s and 1990s but the rates of decline increased through the 2000s. Those taking Snapper with HLs fell from 219 in 1985 to 92 in 2019, and those targeting fell from 177 to 50 over the same period. Between 2004 and 2011, the number of reported daily HL catches (between February and October) declined considerably and from 2012 to 2019 have been relatively low, i.e. generally <400 catches.yr-1. The estimates of Prop200kgHLTar have been variable from year-to-year generally ranging from 0.1 to 0.25 , but show no long-term trend.

From 1984 to 2004, targeted LL catch for the SG/WCS was relatively flat before it increased and peaked at 154.2 t in 2006, before declining again (Figure 4-3). By 2019, it had fallen to 22.9 t , the $2^{\text {nd }}$ lowest amount. Since targeted LL effort peaked at 2,578 fisher-days in 1997, it has declined considerably. From 2014 to 2018, it was relatively flat and then declined to the lowest level of 523 fisher-days in 2019. Between 2005 and 2008, targeted LL CPUE peaked, with the highest at 98.7 kg.fisher-day $^{-1}$ in 2006. From 2008, it fell considerably and by 2014 had dropped to 33.7 kg.fisher-day ${ }^{-1}$. Subsequently it increased to 52.8 kg. fisher-day $^{-1}$ in 2018, but fell again to 43.9 kg. .fisher-day $^{-1}$ in 2019. Since 1988, the numbers of license holders taking Snapper fell from 118 to 40 and those targeting it fell from 100 to 32 (Figure 4-3). The numbers of reported daily LL catches fell between 2006 and 2011 and have subsequently remained at the relatively low level of $<500$ catches. $\mathrm{yr}^{-1}$. The annual estimates of Prop200kgLLTar declined to approximately 0.1 in 2011 and have since remained around this low level.


Figure 4-3. Key fishery statistics used to inform the status of the Spencer Gulf/ West Coast Stock of Snapper. Long-term trends in (A) total catch. (Left) trends in (B) targeted handline catch; (C) effort; (D) catch rate; and (E) the number of active licence holders taking and targeting the species; (F) number of targeted daily catches and Prop200kgTarHL. (Right) trends in (G) targeted longline catch; (H) effort; (I) catch rate; and (J) the number of active licence holders taking and targeting the species; (K) number of targeted daily catches and Prop200kgTarLL. Green and red lines represent the upper and lower reference points identified in Table 3-1.

## Gulf St. Vincent Stock

Between 1984 and 2006, the GSVS produced relatively low catches (Figure 4-4). However, from 2006 to 2010, total catch increased exponentially culminating in the record catch of 454.1 t. Total catch declined marginally between 2010 and 2015 after which the rate of decline increased. In 2019, total catch was 171.4 t, i.e. $37.7 \%$ of the record level, and the lowest since 2008.

Targeted HL catch has generally been low for this stock despite the high effort levels during the early 1980s (Figure 4-4). Targeted effort declined to a low level in 1995 and has since remained low but has varied cyclically. Estimates of annual targeted HL CPUE were low until 2006, before they increased to the highest levels between 2007 and 2013. It has subsequently decreased to a moderate level, with 34.6 kg.fisher-day ${ }^{-1}$ recorded in 2019. The numbers of handline license holders fell considerably through the 1980s and 1990s. The number that reported taking Snapper in 1984 was 96 , which fell to 41 in 2019. Similarly, the number who targeted Snapper fell from 89 to 28 . The numbers of reported daily handline catches have generally been $<300 . \mathrm{yr}^{-1}$ since 2004. The estimates of Prop200kgTarHL were $<0.2$ between 2007 and 2010, but since 2014 have been low at <0.1.

The LL fishery for the GSVS largely accounted for the recent rapid increase in total catches. Between 2008 and 2015, targeted LL catch increased from 46.7 t to 388.2 t (Figure 4-4). This increase was associated with a $334.1 \%$ increase in targeted longline fishing effort from 657 to 2,852 fisher-days. Nevertheless, targeted fishing effort declined between 2016 and 2019 from 2,558 to 1,487 fisher-days. Between 2000 and 2010, LL CPUE increased considerably, peaking at 145.7 kg.fisher-day ${ }^{-1}$. Since 2015 , it has declined consistently to 100.4 kg .fisherday ${ }^{-1}$ in 2019. The numbers of LL license holders who took and targeted Snapper peaked in 2012 at 66 and 64, respectively and have since declined considerably to 29 and 28 in 2019. The numbers of daily longline catches increased from 2007, peaked in 2012 at 1,448 catches and then declined considerably between 2016 and 2019 to 693 catches. The Prop200kgTarLL was low from 2004 to 2008 but then increased up to 0.57 in 2014. Since then there has been a general decline to 0.43 in 2019 .


Figure 4-4. Key fishery statistics used to inform the status of the Gulf St. Vincent Stock of Snapper. Long-term trends in (A) total catch. (Left) trends in (B) targeted handline catch; (C) effort; (D) catch rate; and (E) the number of active licence holders taking and targeting the species; (F) number of targeted daily catches and Prop200kgTarHL. (Right) trends in (G) targeted longline catch; (H) effort; (I) catch rate; and (J) the number of active licence holders taking and targeting the species; (K) number of targeted daily catches and Prop200kgTarLL. Green and red lines represent the upper and lower reference points identified in Table 3-1.

## South East Region

The SE region has generally produced low catches of Snapper (Figure 4-5). However, from 2006 to 2010 there was an exponential increase in catch that peaked in 2010 at 260.9 t . It then fell consistently and in 2016 was only 3.5 t . It has since increased to 19.1 t in 2018 and 18.4 t in 2019.

Targeted HL catch in the SE has always been low. There was an increase between 2006 and 2009, which peaked in 2007 at 12.4 t , but which has subsequently declined (Figure 4-5). Such catches reflect low but variable fishing effort, which peaked at 316 fisher-days in 2007. Up to 2003, targeted HL CPUE was generally $<20 \mathrm{~kg}$.fisher-day ${ }^{-1}$. It then increased to its highest levels from 2006 to 2009, peaking at 68.6 kg.fisher-day ${ }^{-1}$ in 2008. From then, HL CPUE declined to the lowest level in 2017 before increasing sharply in 2019. The numbers of HL fishers who took and targeted Snapper peaked in 2009, at 16 and 13, respectively. They have subsequently declined and were at seven and six fishers, respectively in 2019. Since 2004, the numbers of reported daily catches have been consistently low having declined from a peak of 93 catches in 2007 to only seven catches in 2019. Prop200kgTarHL was highest from 2006 to 2009 , but subsequently has generally been zero.

Up to 2007, targeted LL catches were generally less than one tonne. $\mathrm{yr}^{-1}$. After this, there was a rapid increase to the maximum level of 239.2 t in 2010 (Figure 4-5). It then declined to 9.0 t in 2017 before increasing to 18.6 t in 2018 and then to 16.6 t in 2019. There was a considerable increase in targeted LL effort that peaked in 2010 at 2,614 fisher-days. This subsequently declined to only 162 fisher-days in 2017 but has increased marginally to 203 fisher-days in 2019. Targeted LL CPUE also increased considerably between 2007 and 2010, peaking at 91.5 kg.fisher-day ${ }^{-1}$. Since then it has been variable, but shown no long-term trend. The numbers of LL fishers who took and targeted Snapper increased dramatically from 2005 and peaked in 2010 at 35 and 27, respectively. They declined to 11 and 10 in 2019. The reported numbers of daily catches increased from 2007, peaked in 2010 at 699 catches and subsequently declined to a minimum of 43 in 2016, before increasing marginally in 2017, 2018 and 2019. Prop200kgTarLL also peaked in 2010 at 0.52 and declined to 0.02 in 2016. It has risen again to 0.28 in 2018 and 0.44 in 2019.


Figure 4-5. Key fishery statistics for the population of Snapper in the South East Region. Longterm trends in (A) total catch. (Left) trends in (B) targeted handline catch; (C) effort; (D) catch rate; and (E) the number of active licence holders taking and targeting the species; (F) number of targeted daily catches and Prop200kgTarHL. (Right) trends in (G) targeted longline catch; (H) effort; (I) catch rate; and (J) the number of active licence holders taking and targeting the species; $(\mathrm{K})$ number of targeted daily catches and Prop200kgTarLL. Green and red lines represent the upper and lower reference points identified in Table 3-1.

### 4.2. Regional Estimates of Size and Age Structures

## Northern Spencer Gulf

For Snapper in NSG, the sample sizes of fish considered in the market sampling and the resulting size structures changed considerably through the period of 2008 to 2019 (Figure 46). Initially, the sample sizes were large due to the high catches from this region but they declined considerably particularly after 2012, reflecting the lower catches and fewer fish from this region that were passing through the SAFCOL fish market.

Through the mid-2000s from 2008 to 2015, the size distributions were generally broad, but were dominated by fish in the 'small' and 'large' size categories, whilst the weight distributions were dominated by the 'large' or 'very large' size categories. This situation changed considerably from 2016 onwards. In that year, the size and weight distributions were dominated by 'small' fish. This situation was similar in the following years of 2017 to 2019, although with higher representation from the 'medium' size category in the size and weight distributions. The age structures provide considerable insight into the demographic processes that were responsible for these changes. From 2008 to 2015, the age structures were relatively broad, i.e. involved a number of year classes but nevertheless were dominated by several year classes, particularly those that recruited in 1997 and 1999. By 2017, these two strong year classes were depleted and the age structures mainly involved fish that had recruited between 2012 and 2016, of which the 2014 year class appears to have been the strongest. These recent age structures display an obvious lack of older fish, suggesting that the population had become considerably truncated.


Figure 4-6. Estimated annual size and biomass distributions for Snapper caught in NSG from 2008 to 2019. Left hand graphs show the size structures. Right hand graphs show the percentage of biomass accounted for by each size class.


Figure 4-7. Estimated annual age structures for fish caught in NSG between 2008 and 2019. For each year, data are presented as the relative percentage of total catch accounted for by each year class, i.e. the years in which they were spawned.

## Southern Spencer Gulf

Population size structures for SSG are available for most years from 2008 to 2020 (Figure $4-8$ ). Sample sizes were highly variable amongst years, but generally declined over time, and were particularly low for several years after 2015. The annual size structures reflected broad size ranges of fish but generally involved modes of 'small' and 'large' fish, whose relative sizes varied between years. The weight distributions for this region were generally dominated by 'large' fish. The recent truncation in the size structures that was apparent for NSG was not as strongly apparent for this region.

There were sufficient fish sampled from this region to develop age structures for most years from 2008 to 2015. These age distributions were dominated by two particular year classes, i.e. those that had recruited in 1997 and 1999. By 2015, these two year cases had become depleted and the age distribution involved fish across a range of year classes particularly the strong 2007 year class. The age structure in 2020 was broad and involved several strong year classes, i.e. those that had recruited in 2006, 2007, 2009 and 2014.


Figure 4-8. Estimated annual size and biomass distributions for Snapper caught in SSG from 2008 to 2019. Left hand graphs show the size structures. Right hand graphs show the percentage of biomass accounted for by each size class.


Figure 4-9. Estimated annual age structures for fish caught in SSG between 2008 and 2019. For each year, data are presented as the relative percentage of total catch accounted for by each year class, i.e. the years in which they were spawned.

## Northern Gulf St. Vincent

This region has dominated the catches of Snapper in SA since 2012. Relatively high numbers of fish from this region were measured annually up to 2017, but sample sizes have subsequently declined considerably. All four size categories were generally well represented in the size structures in each year, although with some variation amongst years in their relative contributions (Figure 4-10). In 2008, the 'small' fish were most numerous, whilst in 2009 and 2010 the 'large' and 'very large' fish were most numerous. No modal structure was evident in the size structures of 2011 and 2012, indicating that all size categories contributed to the catches. The annual size structures from 2016 to 2020 were dominated by 'large' fish. Besides there being relatively fewer 'small' fish in these size structures, there were also fewer 'very large' fish compared to previous years, suggesting some recent contraction in the size structures. Such contraction is more evident in the annual weight distributions. From 2008 to 2015, these were unimodal and involved 'large' and 'very large' fish. From 2016 onwards, the contribution of the 'very large' fish declined, and the weight distributions became dominated by the 'large' fish.

For NGSV, there were sufficient otoliths collected in most years from 2008 to 2020 to develop population age structures (Figure 4-11). These were generally characterised by a broad number of year classes. Furthermore, numerous strong year classes contributed to the catches in some years, which were consistent across a number of consecutive years. These were the 1991, 1997, 1999, 2001, 2004, 2006, 2007 and 2009 year classes. The age structures in 2018, 2019 and 2020 also indicate that the 2014 year class was at least a moderate one. The age structure developed in 2020 also suggests the possible emergence of the 2017 year class as another strong one.


Figure 4-10. Estimated size and biomass distributions for Snapper caught in NGSV from 2008 to 2019. Left hand graphs show the size structures. Right hand graphs show the percentage of biomass accounted for by each size class.


Figure 4-11. Estimated annual age structures for fish caught in NGSV between 2008 and 2019. For each year, data are presented as the relative percentage of total catch accounted for by each year class, i.e. the years in which they were spawned.

## Southern Gulf St. Vincent

The numbers of fish measured from SGSV were relatively high until 2015 after which they have declined considerably. The size structures from 2008 to 2015 were generally multi-modal and dominated by 'small' and 'medium' sized fish (Figure 4-12). The weight distributions were dominated by the 'medium' sized fish. From 2016 to 2019, modal structure was less evident in the size and weight distributions, but they suggest relatively higher numbers of 'large' fish. In 2020, the size distribution was broad, but was nevertheless dominated by 'medium' fish, whilst the weight distribution reflected the relatively high contributions of both 'medium' and 'large' fish.

The age structures from 2008 onwards involved very few fish that recruited throughout the 1990s (Figure 4-13). Those from 2008 to 2016 were dominated by the year classes of 2001, 2004, 2006, 2007 and 2009 with the relative significance of each year class increasing and then decreasing over time as it became depleted. The age structures for 2017 and 2018 were dominated by fish that recruited between 2005 and 2010, and also reflected the emergence of the 2014 year class as another possible strong one. This was reinforced by the age structures for 2019 and 2020 that are dominated by the 2007, 2009 and 2014 year classes. The recent age structures are similar to those of NGSV.


Figure 4-12. Estimated annual size and biomass distributions for Snapper caught in SGSV from 2008 to 2019. Left hand graphs show the size structures. Right hand graphs show the percentage of biomass accounted for by each size class.


Figure 4-13. Estimated annual age structures for fish caught in SGSV between 2008 and 2019. For each year, data are presented as the relative percentage of total catch accounted for by each year class, i.e. the years in which they were spawned.

## South East

For the SE Region, size distributions are available for most years from 2008 to 2020 (Figure 4 -14). The sample sizes were variable but generally higher up to and including 2014. The size structures to 2012 were dominated by 'small' and 'medium' fish, and rarely involved fish $>60 \mathrm{~cm}$ CFL. For several years until 2017, there was a proportional increase in the representation of 'large' fish. As such, the weight distributions from 2008 to 2017 were dominated by 'medium' fish, but involved some 'large' fish. The size distribution in 2020 had a large mode of 'small' fish as well as some 'medium' ones. The weight distribution was bimodal, involving both size categories.

The age structures up to 2014 were dominated by two strong year classes, i.e. the 2001 and 2004 year classes, whose relative contributions changed between 2009 and 2014. The age structure in 2015 was dominated by the 2007 and 2009 year classes. Those for 2019 and 2020 were both dominated by two pairs of year classes, i.e. those that recruited in 2009 and 2010, and those that recruited in 2013 and 2014.


Figure 4-14. Estimated annual size and biomass distributions for Snapper caught in SE from 2008 to 2019. Left hand graphs show the size structures. Right hand graphs show the percentage of biomass accounted for by each size class.


Figure 4-15. Estimated annual age structures for fish caught in SE between 2008 and 2019. For each year, data are presented as the relative percentage of total catch accounted for by each year class, i.e. the years in which they were spawned.

## Growth: Length-at-age for the three stocks

Previous analysis of growth for Snapper in SA (e.g. Fowler et al. 2013; McGarvey and Feenstra 2004) demonstrated high regional variation in mean Snapper length-at-age, with SSG showing smaller mean lengths at age than NSG. Snapper also grow more slowly in the SE. In this assessment, Snapper growth estimates were updated using the revised model regional breakdown aligned with the reproductively discrete sub-populations, SG/WC, GSV, and SE.

Slower growth of Snapper from the SE Region compared with those from GSV was confirmed in this re-fitting of mean length-at-age (Figure 4-16). Results for mean growth were more uncertain in the now aggregated SG/WCS. Snapper of both large and small size were evident in the scatterplot of individual samples in this SG/WCS (Figure 4-16), with visually distinct groupings of larger and smaller lengths-at-age above 15 years of age. A bimodal distribution of lengths-at-age is unusual and likely reflects the presence of two distinct growth groups in the SG/WCS. Little sampling has been undertaken on the WC so nearly all the length-at-age samples are from SG. It seems likely that this bimodal separation reflects the previously estimated slower growth of SSG Snapper compared with NSG, these two populations now aggregated in the model.

Overall variation in Snapper growth among individual fish in all regions is high (Figure 4-16). In the two gulfs, a 10 -year-old Snapper can vary in total length from around $350-900 \mathrm{~mm}$. This span of body lengths is the approximate growth in mean size between about 3 and 20 years of age. Such wide variation in body sizes at age means that cohorts take many years to grow fully into legally harvestable size, the faster growers reaching 380 mm around age 3, and slower growers recruiting around age 12 in GSV, and at still older ages in the slow-growing region of SSG. One important advantage of the slice-partition modelling method employed in assessing the three MSF stocks is that this wide variation in lengths-at-age is accounted for. Slices model this on-going partial recruitment of Snapper to legal size more accurately in fitting to the age proportions, mean lengths, catch totals, and catch rates, all of which are strongly dependent on the proportions of each cohort that are susceptible to capture.

Further details, including growth model equations and maximum likelihood fitting method used to estimate the red-line mean growth curves of Figure 4-16 and their confidence intervals, are given in Appendix 7.2.


Figure 4-16. Fitted von Bertalanffy growth as Snapper length-at-age for the SA three regions. See Appendix 7.2 for mathematical details. Solid red lines plot the fitted mean length at age (Eq. 7.2). Shaded bands show $95 \%$ confidence intervals (1.96 times estimated $\sigma\left(a_{i}\right)$ ). Scatterpoints of SA Snapper age-length samples are translucent to graphically display the density of points in the data scatterplot. Black data points shown are commercial, primarily SAFCOL market, samples. Blue points represent age-length samples taken in the two gulfs by SARDI researchers that were not subject to the cut-off at the legal minimum length of 380 mm TL.

### 4.3. Estimates of Spawning Biomass

## Female Weight (W)

Weight frequency histograms for each gulf were constructed using the total weights of mature females collected through fishery dependent and independent sampling from 2019 and 2020. Small Snapper weighing <2 kg dominated the SG, accounting for $65 \%$ of the sample (Figure 4-17). Similar size fish were also evident in GSV, however, there were significantly more large fish ( $>4 \mathrm{~kg}$ ) in the population, accounting for $50 \%$ of the population (Figure 4-17). The weight distribution in GSV was bimodal with peaks at 1.5 kg and 6 kg whereas the weight distribution for SG was left skewed with a peak at 1 kg (Figure 4-17). This non-normality in the data was accounted for when estimating $p_{w}^{\mathrm{N}}$ by using the approach of McGarvey et al. (in review) which uses a multinomial distribution rather than assuming a single mean weight.


Figure 4-17. Weight frequencies of Snapper combined from the fishery-dependent sampling program and fishery-independent adult sampling for Spencer Gulf in 2013, 2018 and 2019 and Gulf St. Vincent in 2014, 2018 and 2019.

## Sex Ratio (R)

Snapper collected from fishery dependent and independent methods in 2019 and 2020 demonstrated a slight balance towards females in each gulf with an $R$ of $54 \%$ in GSV and $55 \%$ in SG (Table 4.1).

Table 4-1. Population sex ratio (R) by weight for Spencer Gulf in 2013, 2018 and 2019 and in Gulf St. Vincent in 2014, 2018 and 2019.

| YEAR | GULF | FEMALE SEX RATIO (SE) |
| :---: | :---: | :---: |
| $\mathbf{2 0 1 4}$ | GULF ST. VINCENT | $0.59(0.035)$ |
| $\mathbf{2 0 1 8}$ | GULF ST. VINCENT | $0.40(0.033)$ |
| $\mathbf{2 0 1 9}$ | GULF ST. VINCENT | $0.54(0.027)$ |
| $\mathbf{y y y}$ |  |  |
| $\mathbf{2 0 1 3}$ | SPENCER GULF | $0.57(0.059)$ |
| $\mathbf{2 0 1 8}$ | SPENCER GULF | $0.40(0.062)$ |
| $\mathbf{2 0 1 9}$ | SPENCER GULF | $0.55(0.037)$ |

## Batch Fecundity (F)

The relationship between batch fecundity ( $F$ ) and total female weight (W) was best described by allometric linear regression given that variance increases with weight (Figure 4-18). No statistical differences were detected between the relative slopes (analysis of covariance, year*weight interaction: $F_{2,109}=0.07, p=0.94$ ) nor intercepts (year: $F_{2,109}=0.23, p=0.53$ ) of the linear relationships between years. Consequently all data were combined into a single analysis and fitted using maximum likelihood (Figure 4-18).


Figure 4-18. Batch fecundity versus body weight for South Australian Snapper. The error bars indicate the standard error of the residuals (light blue shading) and the standard error of the $F_{w}$ estimates (blue shading).

## Spawning Fraction (S)

Spawning fraction was estimated as 0.85 for GSV at the time of the egg survey (January 2020) based on the results of histology conducted on 99 mature fish (Table 4-2). Samples for histology could not be accessed for SG, therefore inferences were made from the results of GSV based on the proportion of fish in each macroscopic stage that were in spawning condition. This allowed fish that were staged macroscopically during the SG egg survey (December 2019) to be used to estimate a spawning fraction of 0.37 (Table 4-2). This lower spawning fraction for SG represents the large number of mature females that were not in spawning condition (i.e. Stage 2) at the time of the survey. In previous surveys, a spawning fraction of 0.72 was used in each gulf based on the results of Saunders (2009).

Table 4-2. Population spawning fraction (S) for Spencer Gulf in 2013, 2018 and 2019 and in Gulf St. Vincent in 2014, 2018 and 2019.

| YEAR | Gulf | SPAWNING FRACTION <br> (SE) |
| :---: | :---: | :---: |
| $\mathbf{2 0 1 4}$ | GULF ST. VINCENT | $0.72(0.054)$ |
| 2018 | GULF ST. VINCENT | $0.72(0.054)$ |
| 2019 | GULF ST. VINCENT | $0.85(0.095)$ |
| $\mathbf{2 0 1 3}$ | SPENCER GULF | $0.72(0.054)$ |
| $\mathbf{2 0 1 8}$ | SPENCER GULF | $0.72(0.054)$ |
| $\mathbf{2 0 1 9}$ | SPENCER GULF | $0.37(0.047)$ |

## Distribution and Abundance of Eggs

In 2019, the mean egg densities were considerably lower in SG in comparison to previous surveys (Figure 4-19). Despite an expanded survey area that covered $9235 \mathrm{~km}^{2}$, eggs were not present at numerous sites across the gulf. Where eggs were present, their density was often less than 1 egg. $\mathrm{m}^{-2}$ with the highest recorded density of 3.9 eggs. $\mathrm{m}^{-2}$. In 2013, Snapper eggs were patchily distributed in SG with densities generally $<5$ eggs. $\mathrm{m}^{-2}$ with a number of hotspots with densities of $5-15$ eggs. $\mathrm{m}^{-2}$ (Figure 4-19). In 2018, Snapper eggs were distributed more evenly throughout the gulf but had generally lower densities at locations that were hotspots in 2013 (i.e. the Illusion, the Santa Anna, the Estelle Star and Jurassic Park). The low egg densities sampled in 2019 are consistent with the low $S$ estimated in SG (Table 4-2). It should be noted that the effect of low egg densities on $P_{0}$ and $A$ in 2019 was offset by the low $S$ in the final estimate of spawning biomass.

In GSV in 2019, higher mean egg densities were recorded than in 2018 (Figure 4-19). Many stations had egg densities of $>5$ eggs. $\mathrm{m}^{-2}$ while stations had densities of $5-15$ eggs.m ${ }^{-2}$ (Figure 4-19). Two hotspots occurred in northern GSV that had densities of $15-25$ eggs.m² (Figure 4-19). For GSV, in 2014, the highest egg densities occurred in northern GSV with a
large hotspot of $>25$ eggs. $\mathrm{m}^{-2}$ at Tapley Shoal. However, egg densities were lower in southern GSV (often <1 egg.m²) despite eggs being present at most stations (Figure 4-19). In 2018, egg densities were highest in the northern stations with several hotspots located across the gulf between Port Adelaide and Black Point. As sampling did not occur in southern GSV due to weather constraints during the 2018 survey, egg densities in Investigator Strait were not available.


Figure 4-19. Estimates of the densities of Snapper eggs in Spencer Gulf in 2013, 2018 and 2019 and in Gulf St. Vincent in 2014, 2018 and 2019. The sample stations in each region and year are indicated as dots.

## Spawning Area (A)

In 2019, the spawning areas for GSV and SG were larger than in previous surveys due to the expanded survey areas covered in these surveys (Table 4-3). In SG in 2019, the spawning area was the largest recorded at $4772.79 \mathrm{~km}^{2}$ (Table 4-3). However, this only represents an increase of $20 \%$ from 2018 , despite the surveyed area increasing by $52 \%$. Similar to the measured egg densities in SG, this contraction of spawning area relative to surveyed area is consistent with the low $S$ of 0.37 (Table 4-3).

The 2019 survey in GSV covered more than twice the area of the previous survey due to the impacts of weather during the 2018 survey. As a result, $A$ in 2019 is largest recorded at $10,111.49 \mathrm{~km}^{2}$.

Table 4-3. Regional estimates of Snapper spawning area $(A)$ and the area covered in each survey

| YEAR | Gulf | SURVEY AREA (km²) | SPAWNING AREA ( $\mathrm{km}^{2}$ ) |
| :---: | :---: | :---: | :---: |
| 2014 | GULF ST. VINCENT | 8021.77 | 6434.47 |
| 2018 | GULF ST. VINCENT | 5059.41 | 3125.78 |
| 2019 | GULF ST. VINCENT | 10244.54 | 10111.49 |
|  |  |  |  |
| 2013 | SPENCER GULF | 4609.8 | 2799.99 |
| 2018 | SPENCER GULF | 4791.6 | 3979.29 |
| 2019 | SPENCER GULF | 9234.8 | 4772.79 |

## Daily Egg Production ( $\mathrm{P}_{0}$ )

The estimates of $P_{0}$ in SG have declined consistently over time, regardless of the area surveyed (Table 4-3; Table 4-4). In 2013, $P_{0}$ was estimated at 2.37 eggs.m² which decreased to 0.99 eggs. $\mathrm{m}^{2}$ in 2018 and 0.47 eggs. $\mathrm{m}^{2}$ in 2019. However, the low spawning fraction in 2019 (Table 4-3) accounts for this lower estimate of $P_{0}$.

For GSV, a large decline in $P_{0}$ from 11.56 eggs. $\mathrm{m}^{-2}$ to 1.83 eggs. $\mathrm{m}^{-2}$ occurred between 2014 and 2018 in GSV (Table 4-4). Previously, there was some uncertainty in the $2018 P_{0}$ estimate as the egg sampling was only partially completed in that year. However, the estimated $P_{0}$ in 2019 (2.24 eggs. $\mathrm{m}^{-2}$ ) was very similar to that in 2018 , which validates that $P_{0}$ has declined significantly since 2014.

The assumed value of $0.4 \mathrm{yr}^{-1}$ for egg mortality $(Z)$ was not influential for $P_{0}$ estimates in either gulf (Figure 4-20). Sensitivity analyses demonstrate that altering $Z$ within reasonable bounds ( $0.2-0.6 \mathrm{yr}^{-1}$ ) increases or decreases $P_{0}$ by up to $10 \%$ in either gulf (Figure 4-20).

Table 4-4. Estimates of mean daily egg production ( $P_{0}$ ) for Snapper for 2013, 2018 and 2019 for Spencer Gulf; 2014, 2018 and 2019 for Gulf St. Vincent. All estimates of $P_{0}$ were determined using an egg mortality rate $(Z)$ of 0.4 day $^{-1}$.

| YEAR | Gulf | $P_{0}$ | SE |
| :---: | :---: | :---: | :---: |
| 2014 | GULF ST. VINCENT | 11.56 | 4.31 |
| 2018 | GULF ST. VINCENT | 1.83 | 0.26 |
| 2019 | GULF ST. VINCENT | 2.24 | 0.21 |
|  |  |  |  |
| 2013 | SPENCER GULF | 2.37 | 0.54 |
| 2018 | SPENCER GULF | 0.99 | 0.10 |
| 2019 | SPENCER GULF | 0.47 | 0.06 |



Figure 4-20. Estimates of egg density $\left(P_{0}\right)$ for GSV and SG in 2019. Black lines represent the mean $P_{0}$ values and the extent of the coloured bars shows the mean $\pm$ standard error. Five sensitivity scenarios are presented (coloured bars) for different assumed egg mortalities ( $Z$ ). A $Z$ of 0.4 is used as standard in all further analyses. Note that scales on the $y$-axis differ across panels.

## Spawning Biomass (SB)

The estimates of spawning biomass of Snapper in both gulfs have declined significantly since the initial surveys were undertaken in 2013 for SG and 2014 for GSV (Table 4-5). It is important to note that each estimate of spawning biomass from DEPM must be considered as the population size within the area covered by each survey. Therefore, it is difficult to directly compare amongst surveys in each gulf, given changes in survey coverage (Table 4-5; Figure $4-21)$. However, conclusions regarding changes in spawning biomass can still be made by considering the key differences between surveys when examining each estimate. For example, the GSV spawning biomass was initially estimated as $2,780 \mathrm{t}$ in 2014 but has since fallen to 811 t in 2019 (Table 4-5). In 2018, the survey was not completed due to bad weather. Therefore, the surveyed area in 2014 was $59 \%$ larger than 2018, whilst 2019 is twice the size of 2018 (Table 4-5; Figure 4-21). Therefore, a larger estimate of spawning biomass of Snapper in 2019 is unlikely to represent an increasing population since 2018 but rather a more accurate estimate of the biomass in the whole of GSV (Table 4-5; Figure 4-21). An additional sensitivity analysis was performed for GSV where the 2019 survey data was subset to match the 2018 survey design. This analysis revealed that if the same sample design and methods had been used in 2018 and 2019, the estimate of spawning biomass would only have differed by approximately 20 t . This is a non-significant difference between the two surveys due the estimated error around spawning biomass in 2018.

Table 4-5. Comparison of the estimates of spawning biomass between DEPM surveys undertaken in 2013, 2018 and 2019 for Spencer Gulf; 2014, 2018 and 2019 for Gulf St. Vincent.

| YEAR | Gulf | Spawning Biomass $\boldsymbol{t}$ (SE) |
| :---: | :---: | :---: |
| $\mathbf{2 0 1 4}$ | GULF ST. VINCENT | $2,780(1,444)$ |
| 2018 | GULF ST. VINCENT | $343(130)$ |
| 2019 | GULF ST. VINCENT | $811(125)$ |
| 2013 | SPENCER GULF | $280(152)$ |
| 2018 | SPENCER GULF | $192(63)$ |
| 2019 | SPENCER GULF | $177(34)$ |

The spawning biomass in SG has declined from 280 t in 2013 to 177 t in 2019 (Table 4-5; Figure 4-21). However, the SG survey area was significantly larger in in 2019 in comparison to 2013 (Table 4-3; Figure 4-21). Therefore, a direct comparison between these two surveys cannot be made. Considering the expanded survey area undertaken in 2019, it is evident that the spawning biomass of Snapper has continued to decline (Table 4-5; Figure 4-21).


Figure 4-21. Estimates of Snapper spawning biomass (SB) and surveyed areas in 2013, 2018 and 2019 for Spencer Gulf and 2014, 2018 and 2019 in Gulf St. Vincent. Shaded areas represent the egg survey coverage in each of the six surveys and indicate the areas that correspond to each estimate of spawning biomass that are given on each respective panel.

### 4.4. Stock assessment model - SnapEst

## Spencer Gulf / West Coast Stock

The output performance indicators from the SnapEst model for the SG/WCS are presented in Figure 4-22. The notable features of the model estimates of fishable biomass are: (i) the decline through the 1980s and early 1990s to a minimum in 1993; (ii) the subsequent recovery, particularly through the 1990s, increasing to a peak biomass in 2005; and (iii) the subsequent long-term reduction in biomass to 2020, the lowest ever estimated level. The latter decline is strongly influenced by the DEPM survey estimates of biomass from 2013, 2018 and 2019. The reduction in biomass from 2005 onwards reflects low recruitment as well as the continuation of removal of fish through fishing. The poor recruitment is evident as weak year classes throughout the 2000s, following the three strong year classes that had recruited throughout the 1990s (Figure 4-22). Such poor recruitment is consistent with the age structures for NSG, which particularly demonstrate the significance of the 1997 and 1999 year classes as well as the lack of large, old fish in the population of NSG after 2015 (Figure 4-7).

From 1984 to 2014, estimates of harvest fraction generally varied between 0.1 and 0.2. These rose steeply from 2014 onwards and by 2019 exceeded 0.4 (Figure 4-22). This recent rise, which occurred despite the considerable declines in recent fishing effort (Figure 4-3), follows as a consequence of the low and declining biomass. The trends in model-estimated egg production over time largely reflected the trends for fishable biomass, indicating a significant decline from 2005 to the lowest level ever in 2020.

Sensitivity analyses were run for this stock in order to determine the relative influences of the different input data sources on output biological performance indicators (Appendix 7-3). The SG/WCS model estimates did not achieve reasonable levels, giving much higher values of biomass, when all three DEPM estimates were excluded from the model fit (Figure 7-1; Figure 7-17). Thus, the model could not provide realistic SG/WCS outputs without the addition of DEPM spawning biomass estimates to anchor the levels of absolute abundance. Apart from the case where all three years of DEPM were excluded, sensitivity of the model to varying the importance of other input data sources was very low, implying that the model results are robust, including a case where the last year of DEPM was omitted. This analysis also shows that age structures are relatively important in model inference.

The fits to input data of the corresponding model-predicted quantities for catch-at-age proportions (handline and longline), mean lengths at age (handline and longline), handline CPUE by half-year, and DEPM biomass, separately by model region, are graphed and presented in Appendix 7.5.


Proportion of pristine
egg production


Figure 4-22. Time series of the four annual biological performance indicators from the SnapEst fishery assessment model for the Spencer Gulf/West Coast Stock. Error bars show 95\% confidence intervals. For comparing these indicators with Management Plan trigger reference points given in Table 3.2, green lines show averages over the last three years compared with blue lines giving averages over the preceding three years for biomass and preceding six years for recruitment.

## Gulf St. Vincent Stock

The estimates of fishable biomass from SnapEst for the GSVS declined marginally from 1984 to 1994, but then rose considerably to a record peak in 2011, particularly from 2004 onwards (Figure 4-23). From 2011, model-estimated biomass has declined, particularly from 2015 onwards dropping to the lowest ever level in 2020. These increasing and then declining trends are consistent with the variation in fishery catches (Figure 4-4). The fits to DEPM estimates of biomass from 2014, 2018 and 2019 have informed this recent decline, whilst the model estimates have declined lower than the levels implied by recent catch rates and DEPM estimates.

The increase in fishable biomass from 1994 to 2011 reflects the numerous strong recruitment year classes that were added to the population from 1991 to 2009, particularly the strong year classes of 2001, 2004, 2007 and 2009 (Figure 4-23). These were evident in the age structures for NGSV (Figure 4-11). However, the ensuing decline in biomass from 2011 onwards, reflects the relatively low recruitment year classes between 2010 and 2017. Model estimates of recruitment averaged 29,500 Snapper per year between 2011 and 2017, which was 11\% of the average of 275,000 per year that had recruited between 2001 and 2010.

From 1984 to 2010, the estimates of harvest fraction were <0.2 (Figure 4-23). They rose steeply from 2008 to the maximum in 2019. This reflected that fishery catches and effort declined more slowly than estimated biomass, resulting in the proportion of biomass that was removed increasing over time. The trend in egg production strongly reflects the variation in fishable biomass over time.

The sensitivity analyses that assessed the results of SnapEst for the GSVS indicated that the model outputs were robust to the different weightings that were assigned to the input parameters (Appendix 7-3). Strong agreement between the model run that excluded DEPM and the (therefore independent) DEPM estimates, both in absolute biomass and in the trend over 2014 to 2019, provide strong mutual validation of this large biomass decline. These results provide confidence in the model and DEPM outputs.




Figure 4-23. Time series of the four annual biological performance indicators from the SnapEst model for Gulf St. Vincent Stock. Error bars show $95 \%$ confidence intervals. For comparing these indicators with Management Plan trigger reference points given in Table 3.2, green lines show averages over the last three years compared with blue lines giving averages over the preceding three years for biomass and preceding six years for recruitment.

## South East Region

DEPM spawning biomass surveys have not been undertaken in the SE. Model biomass in this model region is therefore estimated using standard stock assessment inference, from age proportions, catch totals removed, and HL CPUE as an index of biomass. From 1984 to 2004, the model-estimates of biomass were relatively low and flat, but subsequently increased considerably to a record level in 2009 before declining again over several years (Figure 4-24). The estimated biomass dropped to a minimum in 2016 before increasing marginally in 2017, 2018, and 2019. The large peak in biomass from 2005 to 2013 was related to the recruitment of two strong year classes, i.e. those for 2001 and 2004, which are evident in the population age structures (Figure 4-15). The trend of increasing biomass in recent years appears related to recruitment from the relatively strong 2014 year class.

The harvest fraction for Snapper in the SE Region was relatively consistent at around 0.2 before it increased considerably through the period of 2009 to 2015, reaching a maximum of approximately 0.6 . This related to a substantial increase in fishing effort for Snapper throughout the period of high biomass, primarily for LLs (Figure 4-5). The variation in egg production over time was similar to the temporal variation in fishable biomass.

The Snapper caught in the SE Region are known to have originated from Port Phillip Bay (PPB), Victoria, and then to have migrated to SA waters (Fowler et al. 2017). Yearly variation in recruitment success to PPB has been monitored since 1993 through an annual survey for 0+ Snapper, providing a time-series for a recruitment index. Here, this yearly index of recruitment to PPB was compared with the model estimates of recruitment numbers to age 2 (Figure 4-25). There was a significant correlation ( $R=0.57, p=0.003$ ) between the two time series. This provides evidence that the variation in recruitment and biomass in the SE Region is driven by the annual variation in recruitment to PPB.

For most estimation years, the sensitivity to data weightings shows relatively small divergence among model estimates (Figure 7-3), as we found for the two gulfs. However, in the SE, the model outcomes diverge widely in the last few years (Figure 7-3). This higher model uncertainty as reflected in sensitivity analysis is confirmed by the much wider confidence intervals on model estimates (error bars in Figure 7-3) in the last years of the model time frame. These two differently estimated but self-consistent indications of higher model uncertainty in recent years reflects both the minimal amount age sampling from the SE after 2014 and the lack of DEPM estimates to anchor the model absolute abundance.




Figure 4-24. Time series of the four annual biological performance indicators from the SnapEst model for the South East Region. Error bars show 95\% confidence intervals. For comparing these indicators with Management Plan trigger reference points given in Table 3.2, green lines show averages over the last three years compared with blue lines giving averages over the preceding three years for biomass and preceding six years for recruitment.


Figure 4-25. Comparing South East SnapEst recruitment estimates with the yearly juvenile Snapper (age 0+) survey index from Port Phillip Bay, Victoria. 'Cohort year' indicates the 1 January summer of spawning. The blue line is the recruitment estimated by SnapEst while the red line is the recruitment index from PPB.

### 4.5. Fishery Performance Indicators

## Allocation

For the assessment of the catches of the different commercial fisheries in 2019 against their allocations, their percentage contributions to annual total were compared using Triggers 2 and Trigger 3 reference points (Table 4-6). In this year, no trigger reference points were exceeded.

Table 4-6. Snapper Commercial Fishery Allocation.

|  | MSF | SZRLF | NZRLF | LCF |
| :--- | :---: | :---: | :---: | :---: |
| Commercial allocation | 97.5 | 1.78 | 0.68 | 0.04 |
| Trigger 2 (\%) | $n a$ | 2.68 | 1.3 | 0.75 |
| Trigger 3 (\%) | $n a$ | 3.58 | 2.0 | 1.0 |
| \% total 2015 | 99.37 | 0.46 | 0.18 | 0 |
| \% total 2016 | 99.90 | 0.05 | 0.06 | 0 |
| \% total 2017 | 98.75 | 1.10 | 0.16 | 0 |
| \% total 2018 | 96.35 | 3.59 | 0.06 | 0 |
| \% total 2019 | 97.67 | 2.11 | 0.12 | 0.11 |

## Assessment against Fishery Performance Indicators

Across the SG/WCS, GSVS and the SE Region, there were 11 breaches of trigger reference points for the general fishery performance indicators and 16 breaches for the biological performance indicators (Table 4-7, Table 4-8).

Twelve of the 27 breaches were for the SG/WCS, which included five primary performance indicators. The breaches included the lowest recorded total catch, targeted HL and LL effort and Prop200kgTarLL. The average estimated fishable biomass from SnapEst in the last three years was $51 \%$ below that of the previous three years and approximately $10 \%$ of the peak in 2005 (Table 4-7). Harvest fraction in 2019 was $44 \%$, exceeding the $32 \%$ trigger reference point. Egg production was $2 \%$ of that expected for an unfished stock, and below the $20 \%$ trigger reference point. The average recruitment over the last three years was $34 \%$ lower than that of the previous six years, and $81 \%$ lower than the historical mean.

There were seven breaches for the GSVS, which included three primary performance indicators. In 2019, there was the fifth consecutive annual decrease in catch, whilst the lowest targeted handline effort was recorded. Estimates of fishable biomass over the last three years were, on average, $66 \%$ lower than those of the previous three years, and nearly $20 \%$ of the peak recorded in 2011. Harvest fraction was $66 \%$, more than double the $32 \%$ trigger reference point. Egg production was 2\% of that expected for an unfished stock, below the $20 \%$ trigger
reference point. Average recruitment over the last three years was $87 \%$ lower than that for the previous six years and $88 \%$ lower than that for historical years.

Table 4-7. Comparison of South Australia's Snapper fishery performance indicators against the trigger points prescribed in the MSF Management Plan (PIRSA 2013). Biological (B) and General (G) indicators and whether a primary or secondary indicator are identified. Red shading indicates a negative trigger has been activated. Green shading indicates a positive trigger has been activated.

| Performance Indicator | Type | Primary or Secondary | Trigger Reference Point | SG/WC | GSV | SE |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Total catch | G | Secondary | $3{ }^{\text {rd }}$ lowest/3 ${ }^{\text {rd }}$ highest | Lowest | n | N |
|  |  |  | Greatest interannual change ( $\pm$ ) | n | n | N |
|  |  |  | Greatest 5-year trend ( $\pm$ ) | n | n | N |
|  |  |  | Decrease over 5 consecutive years? | n | y | N |
| Targeted handline effort | G | Primary | $3{ }^{\text {rd }}$ lowest/3 ${ }^{\text {rd }}$ highest | Lowest | Lowest | N |
|  |  |  | Greatest interannual change ( $\pm$ ) | n | n | N |
|  |  |  | Greatest 5-year trend ( $\pm$ ) | n | n | N |
|  |  |  | Decrease over 5 consecutive years? | n | n | N |
| Targeted longline effort | G | Primary | $3{ }^{\text {rd }}$ lowest/3 ${ }^{\text {rd }}$ highest | Lowest | n | N |
|  |  |  | Greatest interannual change ( $\pm$ ) | n | n | N |
|  |  |  | Greatest 5-year trend ( $\pm$ ) | n | n | N |
|  |  |  | Decrease over 5 consecutive years? | n | n | N |
| Targeted handline CPUE | G | Primary | $3{ }^{\text {rd }}$ lowest/3 ${ }^{\text {rd }}$ highest | n | n | N |
|  |  |  | Greatest interannual change ( $\pm$ ) | n | n | Highest |
|  |  |  | Greatest 5-year trend ( $\pm$ ) | n | n | N |
|  |  |  | Decrease over 5 consecutive years? | n | n | N |
| Targeted longline CPUE | G | Secondary | $3{ }^{\text {rd }}$ lowest/3 ${ }^{\text {rd }}$ highest | n | n | $2^{\text {nd }}$ |
|  |  |  | Greatest interannual change ( $\pm$ ) | n | n | N |
|  |  |  | Greatest 5-year trend ( $\pm$ ) | n | n | N |
|  |  |  | Decrease over 5 consecutive years? | n | n | N |
| Prop200kgTarHL |  | Primary | $3^{\text {rd }}$ lowest/3 ${ }^{\text {rd }}$ highest | $3{ }^{\text {rd }}$ | n | Lowest |
|  |  |  | Greatest interannual change ( $\pm$ ) | $2^{\text {nd }}$ | n | N |
|  |  |  | Greatest 5-year trend ( $\pm$ ) | n | n | N |
|  |  |  | Decrease over 5 consecutive years? | n | n | N |
| Prop200kgTarLL |  | Secondary | $3{ }^{\text {rd }}$ lowest/3 ${ }^{\text {rd }}$ highest | Lowest | n | N |
|  |  |  | Greatest interannual change ( $\pm$ ) | n | n | N |
|  |  |  | Greatest 5-year trend ( $\pm$ ) | n | n | N |
|  |  |  | Decrease over 5 consecutive years? | n | n | N |
| Fishable biomass |  | Primary | 3-yr ave is +/-10\% of previous 3-yr ave | -51\% | -66\% | +19\% |
| Harvest fraction |  | Primary | above 32\% (int. standard) | 44\% | 66\% | 15\% |
| Egg production |  | Secondary | <20\% of pristine population | 2\% | 2\% | 4\% |
| Recruitment |  | Secondary | $3-\mathrm{yr}$ ave is +/-10\% of historical mean | -81\% | -88\% | +12\% |
|  |  |  | $3-\mathrm{yr}$ ave is +/-10\% of previous 6-yr ave | -33\% | -87\% | +157\% |
| Age composition |  | Primary | Prop >10yrs <20\% of fished population |  |  |  |

For the SE Region, there were three breaches for the general performance indicators and five for the biological indicators. Two were for primary fishery performance indicators. Handline CPUE had the greatest annual increase and longline CPUE was the $2^{\text {nd }}$ highest recorded, but Prop200kgTarHL was the lowest recorded. Fishable biomass was, on average, 19\% higher in the last three years compared with the previous three years (Figure 4-24). Harvest fraction was $15 \%$, which is below the $32 \%$ trigger reference point. Egg production was $4 \%$ of that
expected for an unfished stock. Average recruitment of the last three years was $157 \%$ higher than for the previous six years and $12 \%$ above the historical mean.

For 2019 and 2020, age structures were generated for five of the six regional populations. Reference points for SSG, NGSV and SGSV were not breached because the percentages of fish older than 10 years ranged from 33.1 to $64.3 \%$ (Table 4-8). However, for NSG and the SE Region, the percentages of fish >10 years old were 7.5 and $14.7 \%$, respectively.

Table 4-8. Sample sizes of Snapper measured and aged by market sampling in 2018. Also, shown is the result from assessment of the trigger reference point for age structure. Red shading indicates a trigger has been activated.

| Region | No <br> measured | No aged | Prop <br> $>10 y r s$ |
| :--- | :---: | :---: | :---: |
| NSG (2019) | 280 | 280 | 7.5 |
| SSG (2020) | 274 | 255 | 64.3 |
| NGSV (2020) | 208 | 200 | 62.5 |
| SGSV | 284 | 290 | 33.1 |
| SE Region | 497 | 85 | 14.7 |
| WC | 0 | 0 | n.a |

## 5. DISCUSSION

### 5.1. Context of this assessment

State-wide commercial catch of Snapper shows variation up to 2010, followed by a substantial decline to 2019. The highest recorded catch was $1,035 \mathrm{t}$ in 2010, which declined by $>75 \%$ to 252 t in 2019, the $2^{\text {nd }}$ lowest recorded. Historically, HLs were the most significant gear type, but the proportional contribution of LLs to total catch increased between 2005 and 2010, resulting in LLs becoming the dominant gear type. Both HL and LL catches have declined since 2010, reflecting the reduction in total fishing effort from 11,895 fisher-days in 2010 to 4,336 fisher-days in 2019, which is the lowest recorded since 1984. The total number of commercial fishers who reported taking Snapper in 2019 was 121, the lowest recorded.

The SG/WCS provided the highest annual catches up to 2009, after which they declined and fell to their lowest levels between 2012 and 2019. In contrast, catches from the GSVS were generally low until around 2004, after which they increased gradually for a few years before an accelerated increase between 2007 and 2010. This stock has become the main contributor to the State-wide catch. The catches from the SE Region also increased rapidly between 2007 and 2010, before they declined to much lower levels from 2017.

The trends in fishery statistics point to ongoing reductions in the biomass of Snapper for both the SG/WCS and the GSVS, over numerous years. In each case they reflect periods of poor recruitment.

### 5.2. Stock Status

## Spencer Gulf / West Coast Stock

From the mid-2000s, the commercial fishery statistics for the SG/WCS showed substantial declines. These declines were apparent for total catch, targeted HL effort and CPUE, targeted LL effort and CPUE, Prop200kgTarLL, targeted catches by gear type and the numbers of fishers who took and targeted Snapper. In 2019, most of the general performance indicators were at, or near, historically low levels. Six trigger reference points for general performance indicators were negatively activated. These patterns suggest a rapid decline and persistent low biomass levels.

Applications of the DEPM (after Steer et al. (2017)) in NSG in December of 2013, 2018 and 2019 demonstrate a decline in spawning biomass over this period. The estimate of spawning biomass in 2019 was $177 \mathrm{t}( \pm 34$; SE); the biomass in 2013 was $280 \mathrm{t}( \pm 152$; SE) when the surveyed area was $41 \%$ smaller. The results from these three applications of the DEPM confirm the inference from the commercial fishery statistics that the spawning biomass of

Snapper in NSG remained low to the end of 2019 and had been at this level for a number of years.

Age structures for the years of 2017, 2018 and 2019 show the population in NSG was dominated by small, young fish up to five years of age, with few older fish. Such age structures contrast with historical ones that included many fish >20 years of age and some >30 years old (Fowler et al. 2016a). Recent age structures no longer include representatives of the 1997 and 1999 year classes, which dominated the age structures up to 2015. These data show that the age structures for NSG are severely truncated and that recent recruitment has been low.

The SnapEst model estimates of fishable biomass declined by $91 \%$ from 5,350 ( $\pm 112$; SE) in 2005 to $468 \mathrm{t}( \pm 72$; SE) in 2020, the lowest estimated value. Model outputs indicate that this decline in fishable biomass relates to poor recruitment throughout the 2000s and increasing harvest fractions, caused by the continued fishing of a depleting stock. The model outputs also show that egg production in 2019 was $2 \%$ of that expected for an unfished stock and that average recruitment over the last three years was $34 \%$ lower than the previous six years, and $81 \%$ lower than the historical mean. Consistent with low recent biomass, poor recent recruitment and high harvest fractions, six biological performance indicator reference points were negatively triggered.

Several independent data-sets demonstrate that the fishable biomass and recruitment for the SG/WCS are at historically low levels. These include: (i) low estimates of commercial catch, effort and CPUE; (ii) the absence of large, old fish in the population; (iii) lack of evidence for the recruitment of any new strong year classes; and (iv) ongoing declines in spawning biomass, from the low level in 2013. The decline in biomass of the SG/WCS has occurred over a number of years and has been apparent at the regional and biological stock levels since 2013 (Fowler et al. 2013). The primary causes of the decline are poor recruitment since 1999, evident as the lack of strong year classes in the annual age structures throughout the 2000s (Fowler et al. 2016a, Fowler et al. 2019), coupled with ongoing fishing of a depleting stock.

The SG/WCS was classified as 'depleted' in 2018. There is compelling evidence that the biomass and recruitment of the SG/WCS are at the lowest recorded levels and that the population in NSG is truncated. The fishable biomass is depleted, recruitment is likely to be impaired and there is no evidence of stock recovery following implementation of management changes. Consequently, the SG/WCS remains classified as 'depleted'.

## Gulf St. Vincent Stock

The commercial fishery statistics for the GSVS, particularly for the LL sector, increased to unprecedented levels between 2007 and 2010, and then remained near these levels until 2015. Since 2015, there have been substantial declines in total catch, targeted LL catch, effort,

CPUE, the number of LL fishers targeting and taking Snapper, the number of their reported daily catches, and Prop200kgTarLL. Two reference points for general performance indicator reference points were negatively triggered. These trends in the fishery statistics are consistent with an increase in biomass that was maintained until around 2015, followed by a rapid decline.

Fishery-independent estimates of spawning biomass from applications of the DEPM in 2014, 2018 and 2020 confirm the decline in biomass, from $2,780( \pm 1,444$; SE) in 2014 to $811 \mathrm{t}( \pm$ 125; SE), despite an expansion of survey area in 2019.

Outputs from SnapEst show fishable biomass increased from a low level in the 1990s to a record level in 2011, before declining by 90\% between 2011 and 2020. The estimate of fishable biomass in 2020 was $456 \mathrm{t}( \pm 81$; SE), the lowest recorded value. The increase in biomass through the 2000s reflected recruitment of numerous strong year classes (1991, 1997, 1999, 2001, 2004, 2007 and 2009) to the population. The subsequent reduction in biomass related to relatively poor recruitment from 2009 to 2017, when catches remained high and harvest fractions increased. Model-estimated egg production in 2020 was $2 \%$ of that expected for an unfished stock; average recruitment over the last three years was $87 \%$ lower than for the previous six years and $88 \%$ lower than the historical level. Consistent with low recent biomass, poor recent recruitment and high harvest fractions, seven biological performance indicator trigger reference points were negatively activated.

In 2019, the status of the GSVS was changed from 'sustainable' to 'depleting' (Table 2-1, Fowler et al. 2019). This reflected the decline in spawning biomass estimated from DEPM surveys that had occurred since 2014, poor recruitment since 2009, and persistent high targeted fishery catch and effort. The evidence in 2020 demonstrates ongoing deterioration of this stock: (i) commercial fishery statistics show further decline in 2019; (ii) the 2019 DEPM estimate confirmed the low level of spawning biomass; (iii) poor recruitment between 2010 and 2017, despite the moderate 2014 year class; and (iv) model-estimated fishable biomass and egg production have declined since 2011, and were at their lowest estimated levels in 2020. There is compelling evidence that the biomass and recruitment of the GSVS are at their lowest recorded levels. The fishable biomass is depleted and recruitment is likely to be impaired. Consequently, the GSVS is classified as 'depleted', reflecting a change from 'depleting' in 2018.

## South East Region

The Snapper population in the SE Region in SA is the western extremity of the crossjurisdictional Western Victorian Stock (Fowler 2016, Fowler et al. 2017). This population is sustained through emigration of fish from the main nursery area, which is located in PPB,

Victoria, i.e. approximately 600 km to the east. This SE region remains open to fishing ( $1^{\text {st }}$ February - $31^{\text {st }}$ October) with a TAC for 2020 of 75 t .

Substantial increases in annual fishery catches, effort and catch rates occurred primarily between 2008 and 2012, but these have subsequently declined. Outputs from the SnapEst model indicate that this reflected a substantial increase in fishable biomass following recruitment of two strong year classes in PPB in 2001 and 2004, and the emigration of Snapper from PPB to the SE Region (Fowler et al. 2017), followed by a decline in fishable biomass due to reduced recruitment into PPB since 2004. Model-estimated fishable biomass increased slightly from 128 t ( $\pm 45$; SE) in 2018 to 160 t ( $\pm 70$; SE)) in 2020, reflecting recruitment of the strong 2014 year class.

Several additional issues need to be considered when setting the target harvest fraction, and TAC, for the SE Region. Firstly, this region is a sink population of the WVS with adult abundance dependent on recruitment success in PPB. Snapper spawned in PPB move to the SE Region of South Australia, but fewer (or many fewer) return. Secondly, recent strong recruitment to PPB in 2013, 2014 and 2018 suggests forthcoming replenishment of the SE Region.

In 2016 (Hamer and Conron 2016) and 2018 (Stewardson et al. 2018), the WVS was classified as 'sustainable'. The annual 0+ recruitment survey showed that over the 12 years to 2016, there had been six years for which recruitment was at, or above, the long-term average. Furthermore, the 2018 year class in PPB was the largest yet recorded. This evidence shows that the adult biomass is at a level sufficient to ensure that, on average, future levels of recruitment are adequate, i.e. recruitment is not impaired, and fishing mortality is adequately controlled to avoid the stock from becoming impaired.

### 5.3. Assessment Uncertainties

There was a high level of consistency in the inferences on stock status from the differing datasets available for this assessment. Nevertheless, there was some uncertainty associated with this assessment.

There were two primary sources of uncertainty in the 2019 DEPM estimates. Firstly, the in situ hybridisation (ISH) of Snapper eggs was less effective in 2019, in comparison to applications for previous surveys (Steer et al 2017; Fowler et al. 2019). The main purpose of ISH is to reduce the need for manual Snapper egg identification and increase identification confidence because eggs from several other fish species can closely resemble those of Snapper (Oxley et al. 2017). As the hybridisation of Snapper eggs in 2019 was weaker than previous surveys, eggs had to be identified manually. The impact of egg mis-identification on the estimates of
spawning biomass is likely to be small. This is because (i) Snapper eggs have a specific size range ( $0.8-0.9 \mathrm{~mm}$; Steer et al 2017) and there was some ISH hybridisation to support identification; and (ii) sensitivity analyses for both gulfs demonstrate that $P_{0}$ would need to differ by orders of magnitude before error in this parameter would substantially alter the conclusions of the 2019 DEPM surveys.

The second source of DEPM uncertainty was the spawning fraction $(S)$ in SG. Spawning fraction is the parameter that scales the number of fish spawning during a survey to the total number of fish within the survey area and is one of the most influential DEPM parameters. As $S$ was low in SG (0.37), small changes to the value of this parameter can strongly influence the spawning biomass estimate. A $5 \%$ variation when $S$ is high adds little uncertainty to the estimate of spawning biomass (see results for GSV), whereas a $5 \%$ variation to a low spawning fraction can alter an estimate of spawning biomass considerably. However, the lowest possible $S$ that could be estimated in SG was 0.22 . If this value of $S$ was used, the estimate of spawning biomass increases from 177 t to 294 t . Conversely, if $S$ in SG was underestimated and similar to that used in 2018, the spawning biomass would be 92 t . Thus, the impact of potential error in $S$ does not change the conclusion that spawning biomass remains low in SG and has continued to decline since 2013.

There is also uncertainty in commercial CPUE. For example, the introduction of daily trip limits in 2012 potentially obfuscated trends in population abundance in CPUE due to changes in fisher behaviour (Fowler et al 2016a). These restrictions had the potential to impact on the financial benefit for fishers to target Snapper, thereby resulting in a decline in targeted fishing effort. This difficulty has largely been overcome through investment in DEPM and its integration into SnapEst.

As there is uncertainty in each SnapEst data input (i.e. hyperstability and thus reliability of CPUE as an index of abundance, and precision of DEPM estimates), model outputs also have uncertainty. Where DEPM estimates of biomass were unavailable, HL CPUE was used to determine historical population trends. This was problematic for the SG/WCS when DEPM was removed, as the truncated age structures in recent years, yielded insufficient information on annual harvest fractions for the model. Thus, DEPM spawning biomass estimates were required for SnapEst to generate outputs coherent with the fishery-dependent data for the SG/WCS. These same issues were not evident for the GSVS or the SE region, where the model outputs were consistent with other data in the absence of spawning biomass estimates. The reliance of DEPM inputs for the SG/WCS is demonstrated by the fits to HL CPUE (Figure 7-16), which were poor after 2007. However, for the GSVS model, the fit to HL CPUE remained comparatively good as there was strong agreement between all of the model inputs. This was
demonstrated in the sensitivity analysis performed in Appendix 7-3 and the results displayed in Figure 7-2.

As is the case with all stock assessments for MSF species, there is a high degree of uncertainty in the magnitude and size-structure of the recreational catch.

### 5.4. Future Work

There are five key priorities for improving the understanding and assessments of Snapper stocks in SA. These include: (i) continuation of DEPM surveys; (ii) accessing biological data such as age and length samples; (iii) better understanding the drivers of Snapper recruitment; (iv) improving the information on recreational catches; and (v) understanding the post-release mortality of Snapper.

Given the cessation of fishing in the gulfs and on the west coast, and the consequent lack of fishery-dependent data for the foreseeable future, monitoring of stock abundance needs to rely heavily on obtaining estimates of spawning biomass from DEPM surveys and contracting commercial fishers to undertake targeted fishing for Snapper to provide biological data, particularly population age structures. Both of these datasets are key inputs to the SnapEst model, which strongly influence the reliability of the model outputs. Determining the appropriate CPUE series to include in SnapEst also remains important for years when DEPM are not available. There is strong agreement between DEPM biomass and the index of abundance inferred from CPUE for the GSVS. However, the poor fits to CPUE for the SG/WCS indicate that a better understanding of CPUE, and how it can be effectively used in assessments, remains important for this stock. Related to this, there is also a need to investigate the reasons for the unexpected low estimates of uncertainty in the model outputs. These could be driven by the method used to incorporate CPUE in the model. This may require further model modification than was undertaken for this assessment.

As indicated above, the DEPM remains a key input for both the GSVS and the SG/WCS while these fisheries remain closed. Therefore, continuing to improve and build upon the advances in Snapper DEPM science remains important. In future assessments, the DEPM sensitivity analysis presented in Appendix 7-4 will be expanded to determine the model sensitivity to uncertainty in individual DEPM parameters.

There is also a need to better monitor recruitment for 0+ Snapper to provide an early indication of year class strength for the SG/WCS and GSVS. This approach, which is used in Victoria for the WVS, is an important information source for managing that fishery (Hamer and Conron 2016). The research project addressing this need is FRDC 2019/046, focussed on recruitment variability of Snapper in SA, has two primary objectives: (i) to develop a sampling method and
protocol for documenting the inter-annual variation in recruitment of 0+ Snapper; and (ii) enhancing our understanding of the variability in recruitment of Snapper in SA.

The fourth and fifth research priorities relate to the level and size structure of the recreational catch and the magnitude of Snapper post-release mortality, respectively. Infrequent and imprecise estimation of recreational catch is a primary limitation on assessment of all Marine Scalefish species. Recreational surveys are run infrequently and commonly result in large standard errors. The regional distribution of catch, due to the smaller sample sizes, is more uncertain. Between survey years, recreational catches are obtained by extrapolation using SA population statistics, which provides no yearly information about the intervening change over time. The next recreational survey is planned for 2020-21. While estimates will become available for the SE Region, if Snapper harvesting in the remainder of SA remains closed, the paucity of information on recreational fishing will persist. Therefore, there will be further need to collect more information on recreational catches in the future, as well as determining the sensitivity of the stock assessment model to the uncertainty of these estimates.

The need to quantify Snapper post-release mortality is being addressed through FRDC 2019/044. This project will inform about the release rates of Snapper from fishing activities by the different fishery sectors, and their post-release survival, and enable identification of 'best practice' fishing and development of 'codes of conduct'. These 'codes of conduct' would promote responsible fish-handling practises, humane treatment and harm minimisation to maximise the survivorship of Snapper after their capture and release.

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## 7. APPENDICES

### 7.1. $\quad$ Snapper Stock Assessment Model (SnapEst)

In this section we summarise the following components of the stock assessment model: (1) growth, (2) recruitment, (3) the population array including length slices, (4) mortality, and (5) the likelihood function relating model to data. The slice-partition method, with detailed pseudocode, is described in Appendix C of the 2015 Garfish stock assessment report (Steer et al. 2016).

## Growth

The starting point and basis of the slice method for partitioning fish cohorts by length is the length-at-age growth submodel. A statistical growth submodel is needed which fully specifies the probability density function (pdf) of fish lengths for each model age. This represents the (normal) distribution of fish by length in each cohort age that would be observed in the absence of length-asymmetric mortality, because length-selective capture mortality will subsequently be imposed on these model cohorts, after they are partitioned into slices. To model mean fish length $\bar{l}$, the mean of the normal length-at-age pdf, for any half-yearly cohort age, $a$, we employed a 4-parameter exponent-generalized von Bertalanffy mean length-at-age curve:
$\bar{l}(a)=L_{\infty}\left\{1-\exp \left[-K\left(\frac{a-t_{0}}{2}\right)\right]\right\}^{r}$ (McGarvey and Fowler 2002). Using two additional parameters, the dependence of the length-at-age standard deviation $\sigma(a)$ is modelled as an allometric function of mean length: $\sigma(a)=\sigma_{0} \cdot(\bar{l}(a))^{\sigma_{1}}$.

The growth parameters can be estimated by fitting to length-at-age samples (1) previous to, or (2) by integrating growth estimation into, the stock assessment likelihood. We undertook both in that order. First we fitted the growth submodel directly to catch lengths-at-age to obtain approximate growth parameter estimates. A likelihood probability of observation truncated at LML was assumed to make explicit the absence of sublegal Snapper in these catch samples (McGarvey and Fowler 2002). A second growth estimation was integrated into the stock assessment likelihood, re-estimating the two parameters that most directly determine the mean rate of growth and spread of lengths at each age, von Bertalanffy $K$ and the normal length-at-age standard deviation coefficient $\sigma_{0}$.

Starting from this growth submodel, an algorithm (described in Appendix C of Steer et al. 2016) was devised to effectively 'slice off' the length subintervals of fish which have grown past legal minimum length (LML) in each model time step. Once this population number is assigned to each newly created slice bin by transferring these fish from the sublegal
component, there is no subsequent further exchange of fish between length bins. Fish within slices incur only mortality. The simplification of neglecting growth diffusion among length bins affords the slice approach large reductions in computation time compared with, for example, a length-transition approach, which requires $\left(n_{L}\right)^{2}$ growth-transition multiplications in each model time step and for each cohort, where $n_{L}$ is the number of length bins. In a slice partition model, growth is quantified as the increasing length range with age of each slice subinterval, and no computation is needed to shift fish among bins.

## Recruitment

Recruitment is defined as the creation of the (normal) length-at-age cohort at age $a_{b}=5$ halfyears (at age 2 years) when the fastest growing fish first reach legal size. The number of fish in each cohort at age, $a_{b}$, is the model estimate of yearly recruitment. Each yearly recruit number is a freely estimated model parameter. The numbers of Snapper above legal minimum length at age $a_{b}$ (in the upper tail of the length at age pdf) are computed (Appendix C of Steer et al. 2016) and defined as the first newly created slice. In subsequent model time steps, new slices are created as the calculated proportion of sublegal fish in each cohort that have grown into legal size since the previous time step, thereby modelling the gradual recruitment of each cohort to fishable sizes over the number of model time steps required, as determined by the growth submodel (Appendix C of Steer et al. 2016).

## Model population array

The model Snapper population array $N(t, r, c, s)$ is 4-dimensional, fish numbers broken down by (1) half-yearly model time step ( $1=$ summer 1983 to $73=$ summer 2020) " $t$ ", (2) spatial region ( $1=$ SGWC, $2=$ GSV, $3=$ SE) " $r$ ",(3) cohort (i.e. year-class, given by year of spawning) " $C$ ", and (4) slice " $S$ ". For region SE the last model time step is 72.

Variable subscripts for winter or summer half-year $\left(t_{\text {season }}\right)$, and cohort age in half-years ( $a=a(t, c)$ ), were calculated as functions of model time step, $t$, and cohort year, C. Ages ran from $a_{b}$ to 48+ half-years, the oldest age being a 'plus' group. Snapper catch and effort, for data and model, were divided into six effort types, $i_{E}$ : (i) handline (all target types), (ii) longline (all target types), (iii) hauling nets and minor gears (all target types), (iv) all other commercial gears combined, (v) charter boat, and (vi) recreational. The three commercial gears, $g$, are handline, longline, and hauling net, with handline having age selectivity modelled by a decreasing logistic function and longline having length selectivity modelled by an increasing logistic function.

## Mortality

Fishing mortality is differentiated for legal and sublegal fish. Legal-size fish, partitioned into length slices, are subject to both fishing and natural mortality. In addition to the knife-edge cutoff below legal minimum length, gear-specific length and age selectivity is modelled for legal size Snapper. Sublegal population numbers (fish below the legal minimum length) incur only natural mortality. Instantaneous natural mortality rate was taken as constant, $M=0.05 \mathrm{yr}^{-1}$ (Gilbert et al. 2006; Gilbert, pers. comm.).

Length selectivity, $\boldsymbol{S}_{\text {len }}$, by region is applied for the longline gear only ( $g\left(i_{E}=2\right)$ ) and follows a logistic function of fish length, the latter specified by the midpoint of each slice (s) $(\bar{l}(s))$, as

$$
\begin{equation*}
s_{l e n}\left(r, g\left(i_{E}\right), s\right)=1 /\left(1+\exp \left[-r_{\text {sel }}\left(r, g\left(i_{E}\right)\right) \cdot\left(\bar{l}(s)-l_{50}\left(r, g\left(i_{E}\right)\right)\right)\right]\right) \tag{7.1.1}
\end{equation*}
$$

where $r_{\text {sel }}\left(r, g\left(i_{E}\right)\right)$ is the logistic slope parameter, $l_{50}\left(r, g\left(i_{E}\right)\right)$ is the logistic $50 \%$ level parameter.

A decreasing logistic selectivity function of fish half-yearly age (a), $\boldsymbol{S}_{\text {age }}$, was applied for handline only $\left(g\left(i_{E}=1\right)\right.$ ), by region, as

$$
\begin{equation*}
s_{\text {age }}\left(r, g\left(i_{E}\right), a\right)=1-1 /\left(1+\exp \left[-r_{\text {agesel }}\left(r, g\left(i_{E}\right)\right) \cdot\left(a-l_{50 \text { agesel }}\left(r, g\left(i_{E}\right)\right)\right)\right]\right) \tag{7.1.2}
\end{equation*}
$$

where the two parameters are analogous to those for the length selectivity function above.
The SnapEst model is catch conditioned with fishing mortality applied by directly subtracting catch in number from the legal sized population slices, and catch data is assumed to be without error.
$\tilde{C} w\left(t, r, i_{E}\right)=$ commercial catch data that is reported in weight (kg) for time step, $t$, region, $r$, and effort type $i_{E}<n_{E}-1$.
$\tilde{C} n\left(t, r, i_{E}\right)=$ charter boat $\left(i_{E}=n_{E}-1\right)$ catch data in number are available since summer 2007, and similarly for recreational ( $i_{E}=n_{E}$ ) catch data in number from three telephone and diary surveys run in 2001/02, 2007/08 and 2013/14. For recreational catch data during nonsurvey periods, catch data input into SnapEst are obtained by linearly interpolation between the surveys. Catches from the 2013/14 survey are carried forward for SGWC and SE and
scaled according to commercial catch for GSV. For the period before the 2001/02 survey catches from the 2001/02 survey were scaled back to 1983 using the trend in human population per region.

The model predicted equivalents to the above for catch in number and in weight are respectively denoted $\hat{C} n\left(t, r, i_{E}\right)$ and $\hat{C} w\left(t, r, i_{E}\right)$ for each $i_{E} \leq n_{E}$, which are computed as sums over cohorts (c) and legal sized slices (s) of $\hat{C} n\left(t, r, c, s, i_{E}\right)$ and $\hat{C} w\left(t, r, c, s, i_{E}\right)$, defined below.

$$
\begin{equation*}
\hat{C} n\left(t, r, c, s, i_{E}\right)=\exp [-0.5 \cdot M] \cdot N(t, r, c, s) \cdot s_{l e n}\left(r, g\left(i_{E}\right), s\right) \cdot s_{a g e}\left(r, g\left(i_{E}\right), a(t, c)\right) \cdot H \exp \left(t, r, i_{E}\right) \tag{7.1.3}
\end{equation*}
$$

and

$$
\begin{equation*}
\hat{C} w\left(t, r, c, s, i_{E}\right)=\hat{C} n\left(t, r, c, s, i_{E}\right) \cdot w(a(t, c), s) \tag{7.1.4}
\end{equation*}
$$

where the weights by age and slice, $w(a(t, c), s)$, are derived in Appendix C of Steer et al. (2016), and harvest rate $\operatorname{Hexp}\left(t, r, i_{E}\right)$ is the predicted harvest fraction of the exploitable population or biomass (i.e. portion of fishable biomass accessible by the gear).
$H \exp \left(t, r, i_{E}\right)$ is calculated using catch in weight data and biomass for the commercial effort types, and in terms of catch in number data and population for charter and recreational anglers.

For $i_{E}<n_{E}-1$,
$H \exp \left(t, r, i_{E}\right)=\frac{\tilde{C} w\left(t, r, i_{E}\right)}{\exp [-0.5 \cdot M] \cdot \sum_{c=\text { cohort1 }}^{\text {cohor } 2 \text { 23plus } n \text { nlegs }}} \sum_{s=1} s_{\text {len }}\left(r, g\left(i_{E}\right), s\right) \cdot s_{\text {age }}\left(r, g\left(i_{E}\right), a(t, c)\right) \cdot w(a(t, c), s) \cdot N(t, r, c, s)$
and for $i_{E}=n_{E}-1$ or $n_{E}$
$H \exp \left(t, r, i_{E}\right)=\frac{\tilde{C} n\left(t, r, i_{E}\right)}{\exp [-0.5 \cdot M] \cdot \sum_{c=\text { cohort1 }}^{\text {chorr } 2 \sum^{\text {pllus nlegs }}} \sum_{s=1} s_{l e n}\left(r, g\left(i_{E}\right), s\right) \cdot s_{\text {age }}\left(r, g\left(i_{E}\right), a(t, c)\right) \cdot N(t, r, c, s)}$

For each region, half-year, and effort type, after the population has undergone growth, it is depleted mid-way for each time step per cohort slice by applying $50 \%$ of natural mortality, removing the catch, followed by the remaining $50 \%$ of natural mortality, thus

$$
\begin{equation*}
N(t+1, r, c, s)=\left(\exp [-0.5 \cdot M] \cdot N(t, r, c, s)-\hat{C} n\left(t, r, c, s, i_{E}\right)\right) \cdot \exp [-0.5 \cdot M] \tag{7.1.7}
\end{equation*}
$$

## Estimation: Parameters and model likelihood

The model likelihood (Fournier and Archibald 1982) was fitted, for each region, to half-yearly (1) handline catch rates (catch totals by weight divided by effort totals), (2) absolute biomass from DEPM surveys (SGWC and GSV), (3) market sample catch proportions by age for handline and longline, and (4) market sample catch moment properties of fish length, including mean Snapper length, for each age for handline and longline.

## Parameters

Estimated parameters for the model fall into five categories: (1) yearly recruit numbers, (2) logistic length and age selectivity, (3) growth, (4) catchability, and (5) standard deviations for the likelihood fits to half-yearly catch rates, and of fits to length moments.

## Likelihood for catch rates

Predicted catch rates were fitted to data for the handline (HL, $i_{E}=1$ ) effort type by region ( $r$ ) and model time step ( $t$ ) using a lognormal likelihood. The periods over which catch rates were fit varied by region, with SE HL catch rate fitted over all model time steps, SGWC catch rate fitted for $t \leq 61$ (up to and including summer 2013/14), and GSV for $t \leq 63$ (summer 2014/15). The catch rate data, $\tilde{C} p u e\left(t, r, i_{E}\right)$, was calculated as the ratio of total catch by weight (kg) divided by total effort (fisher-days), while predicted catch rate was calculated as

$$
\begin{equation*}
\hat{C} p u e\left(t, r, i_{E}\right)=q\left(t_{\text {season }}, r, i_{E}\right) \cdot B \exp _{\operatorname{mid}}\left(t, r, i_{E}\right) \tag{7.1.8}
\end{equation*}
$$

where $B \exp _{\text {mid }}\left(t, r, i_{E}\right)$ is the predicted exploitable biomass mid-way into time step $t$ (after exactly half of the catch is taken), and $q\left(t_{\text {season }}, r\right)$ is the absolute catchability, by region, season (summer/winter) and effort type (HL).

$$
\begin{align*}
B \exp _{m i d}\left(t, r, i_{E}\right)= & \sum_{c=c o h o r t 1}^{c o h o r r 23 \text { plus }} \sum_{s=1}^{\text {nlegs }} s_{\text {len }}\left(r, g\left(i_{E}\right), s\right) \cdot s_{\text {age }}\left(r, g\left(i_{E}\right), a(t, c)\right) \cdot w(a(t, c), s) \\
& \left(\exp [-0.5 \cdot M] \cdot N(t, r, c, s)-0.5 \cdot \hat{C} n\left(t, r, c, s, i_{E}\right)\right) \tag{7.1.9}
\end{align*}
$$

The likelihood for fitting to catch rates was written:

$$
\begin{equation*}
L_{C_{\text {pue }}}=\prod_{t=1}^{n_{t}} \prod_{r=1}^{n_{r}} \frac{1}{\sqrt{2 \pi} \cdot \sigma_{\text {Cpue }}\left(t_{\text {season }}, r, i_{E}\right) \cdot \tilde{C} \text { pue }\left(t, r, i_{E}\right)} \exp \left[-\frac{1}{2}\left(\frac{\ln \left[\hat{C} \text { pue }\left(t, r, i_{E}\right)\right]-\ln \left[\tilde{C} \text { pue }\left(t, r, i_{E}\right)\right]}{\sigma_{C_{\text {pue }}}\left(t_{\text {season }}, r, i_{E}\right)}\right)^{2}\right] \tag{7.1.10}
\end{equation*}
$$

where
$\sigma_{\text {Cpue }\left(t_{\text {season }}, r, i_{E}\right)=\text { estimated standard deviation parameter, one per season, region, and effort }}$ type (HL).

## Likelihood for DEPM biomass

Predicted total fishable biomass was fitted to DEPM survey data on biomass, by survey (surv) and region ( $r$ ) using a normal likelihood with fixed standard deviations. DEPM surveys were available only for the gulf regions, for model time steps corresponding to survey periods as follows: SGWC time steps $(t)=\{61,71,73\}$ and GSV $(t)=\{63,71,73\}$. Legal size $(\geq 380$ mm ) DEPM spawning biomass was assumed to equate to model legal size biomass, with the main correction being to use MFA-block catches to account for the partial coverage of gulf model regions by the DEPM survey as described in Section 3.5.

The likelihood for fitting to the DEPM biomass was written:

$$
\begin{equation*}
L_{D E P M}=\prod_{t=1}^{n_{t}} \prod_{r=1}^{n_{r}} \frac{1}{\sqrt{2 \pi} \cdot \sigma_{D E P M}(\operatorname{surv}(t), r)} \exp \left[-\frac{1}{2}\left(\frac{P(\operatorname{surv}(t), r) \cdot B_{\operatorname{mid}}(t, r)-\operatorname{Bdepm}(\operatorname{surv}(t), r)}{\sigma_{D E P M}(\operatorname{surv}(t), r)}\right)^{2}\right] \tag{7.1.11}
\end{equation*}
$$

where
$B_{\text {mid }}(t, r)=$ predicted total fishable biomass mid-way into time step $t$ (after exactly half of the catch is taken $)=\sum_{c=\text { cohort } 1}^{\text {cohort } 23 \text { plus }} \sum_{s=1}^{s l e g s} w(a(t, c), s) \cdot\left(\exp [-0.5 \cdot M] \cdot N(t, r, c, s)-0.5 \cdot \hat{C} n\left(t, r, c, s, i_{E}\right)\right)$, $\operatorname{Bdepm}(\operatorname{surv}(t), r)=$ DEPM biomass estimate from the survey at time step $t$, $P(\operatorname{surv}(t), r)=$ proportion, per survey, by which model $B_{\text {mid }}(t, r)$ is scaled down to account for a survey not covering the whole fishing region, and $\sigma_{\text {DEPM }}(\operatorname{surv}(t), r)=$ fixed standard deviation parameter, one per survey and region.

## Likelihood for catch samples by age

A multinomial likelihood was used to fit to catch-sample proportions by age. The source data, from the samples per principal gears handline and longline in the half-yearly time steps and four regions where catch was monitored, consists of the observed counts of sampled fish falling into each half-yearly age, $\tilde{n}\left(a ; i_{A}\right)$. But the data fitted consists of the observed counts multiplied by a factor that depends on the relative discrepancy ratio of each age sampled length value compared to that length in the full market samples of lengths (including fish not aged), the latter samples taken as being more length-representative of the population than the aged samples (see the FRDC report, McGarvey and Feenstra (2004)). Finally, each such corrected count at age-length was multiplied by a scaling factor so that the total raw sample size is preserved at the level of region, time step, and gear. The multinomial likelihood factor is written

$$
\begin{equation*}
L_{\text {Ages }}=\prod_{i_{A}=1}^{n_{A}} \prod_{a=a_{b}}^{48+} \hat{p}\left(a ; i_{A}\right)^{\tilde{n}_{\text {cor }}\left(a ; i_{A}\right)} \tag{7.1.12}
\end{equation*}
$$

where
$\boldsymbol{i}_{A}=$ index over the set of $n_{A}$ catch samples of fish ages over half-year, region, and gear;
$\hat{p}\left(a ; i_{A}\right)=$ an array of model-predicted fish proportions captured by age for each sample indexed by $i_{A}$;
$\tilde{n}_{\text {cor }}\left(a ; i_{A}\right)=$ scaled and corrected observed fish numbers for each age in the catch-at-age sample $i_{A}$.

Note that in the SE, the most recent age composition data were gathered over summer October-2019 to March 2020, but the SE model ends the time step prior since catch-log data are incompletely reported from this period. This last summer's age composition data were therefore incorporated into SnapEst's input data set for the last available SE model time step ( $t=72$ ), half-year April-2019 to September-2019 (after decrementing half-yearly age values by 1 ).

## Likelihood for catch samples by length

A normal likelihood was applied to fit the model to data moment 'properties', mean length, standard deviation of length, skewness, and kurtosis. Fournier and Doonan (1987) first proposed fitting to length moments and also fitted a normal likelihood, but to the central moments rather than moment properties. The likelihood for the length moments fit was written:

$$
\begin{equation*}
L_{\text {Lenghs }}=\prod_{i_{A}=1}^{n_{A}\left(i_{m p}\right)} \prod_{i_{m p}=1}^{4} \prod_{a=a_{b}}^{48+}\left\{\frac{\exp \left[-\frac{1}{2}\left(\frac{\left\{\tilde{b}\left(i_{m p}, a ; i_{A}\right)-\hat{b}\left(i_{m p}, a ; i_{A}\right)\right\}}{\sigma_{m p}}\right)^{2}\right]}{\sqrt{2 \pi} \cdot \sigma_{m p}}\right\}^{\tilde{n}\left(a ; i_{A}\right)} \tag{7.1.13}
\end{equation*}
$$

where
$\sigma_{m p}=$ is the estimated moment-likelihood standard deviation parameter, separately per season, region, and gear.
$\tilde{b}\left(i_{m p}, a ; i_{A}\right)=$ observed moment, indexed by $i_{m p}$, per sample and half-yearly age.
$\hat{b}\left(i_{m p}, a ; i_{A}\right)=$ model-predicted counterpart to $\tilde{b}\left(i_{m p}, a ; i_{A}\right)$.
The observed moments were not calculated using the raw counts of fish per age and length category, but instead were based on length counts from the aged fish that were corrected for representative length sampling as noted further above (see the FRDC report, McGarvey and Feenstra (2004)). We weighted each factor in the log-likelihood by the uncorrected sample size $\left(\tilde{n}\left(a ; i_{A}\right)\right)$, that is by the actual number of aged fish. Higher moment properties require more data to be informative. We therefore set criteria for exclusion of smaller catch sample data sets, $i_{A}$, from the $L_{\text {Lenght }}$ likelihood, depending on the moment property fitted. Thus the number of qualifying data sets, $n_{A}\left(i_{m p}\right)$, decreased with increasing moment property $i_{m p}$. We required at least 8 aged fish for kurtosis, 4 for skewness, 2 for standard deviation, and 1 for fitting to mean length. Similarly we required 4 model slices for kurtosis, 3 for skewness, 2 for standard deviation, and 1 for fitting mean length.

## Objective function

Parameters were estimated by minimising the negative of the sum of the logarithm of the likelihood terms described above, using the ADMB estimation software, namely

$$
\begin{equation*}
-\left(\lambda_{\text {Cpue }} \cdot \ln \left[L_{\text {Cpue }}\right]+\lambda_{\text {DEPM }} \cdot \ln \left[L_{\text {DEPM }}\right]+\lambda_{\text {Ages }} \cdot \ln \left[L_{\text {Ages }}\right]+\lambda_{\text {Lengths }} \cdot \ln \left[L_{\text {Lengths }}\right]\right) \tag{7.1.14}
\end{equation*}
$$

where each lambda represents the chosen weighting for each data source. The following values were used for the baseline model run: $\lambda_{\text {DEPM }}=1.0, \lambda_{\text {Ages }}=0.5, \lambda_{\text {Lenghs }}=0.025$, and
$\lambda_{\text {Cpue }}=1.0$ for SE for all $t \geq 1$, for SG $=1.0$ for $t \leq 61$, for GSV $=1.0$ for $t \leq 63$, and $=0$ otherwise.

## Model performance indicators

## Yearly recruitment numbers

Each yearly recruitment number, by region, is an estimated model parameter, as described in Section 3.5. Due to an absence of data on ages for certain year classes, SnapEst estimated extremely low values for the some recruitment years. These were set to a minimum value of 1000 for SE 1984-1990, 2011, 2015-2016, and for GSV 2015.

## Annual fishable biomass

The fishable biomass indicator reported in Section 4.4 and used in trigger reference point comparisons is the biomass summing over all legal-size Snapper, all those greater than 380 mm , by region and time step. The annual fishable biomass indicator is the average of the two half-yearly model estimates, which, for each region $r$ and year $y$ is calculated as follows

$$
\begin{equation*}
B(y, r)=\frac{1}{2} \sum_{t=t_{\text {sasos }}(1, y)}^{t_{\text {seaen }}(2, y)} B(t, r) \tag{7.1.15}
\end{equation*}
$$

where the fishable biomass at the start of each half-yearly time step $t$ in year $y$ is given by

$$
\begin{equation*}
B(t, r)=\sum_{c=\text { cohort1 }}^{\text {cohorr23 plus nlegs }} \sum_{s=1} N(t, r, c, s) \cdot w(a(t, c), s) . \tag{7.1.16}
\end{equation*}
$$

For each time step $t$, ranging from 1 (October 1983 to March 1984) to 73 (October 2019 to March 2020), the biomass sum is over cohorts ( $C$ ranging from cohort1 of 2 year olds to cohort23plus of 23 year olds, i.e. the plus group) and over length slice within each cohort (the slice index $s$ ranging from 1 to nlegs which is the number of length slices of legal size). The summation limits in Equation 7.1.15 span the two seasonal half-yearly times steps in each full year $\left(t_{\text {season }}(1, y)=\right.$ October-March, $t_{\text {season }}(2, y)=$ April-September $)$.

## Annual harvest fraction

A yearly harvest fraction is defined as the sum of the model-predicted half-yearly catches by weight divided by the annual average total fishable biomass (defined above), as follows

$$
\begin{equation*}
H(y, r)=\frac{\sum_{i_{E}=1 t=t t_{\text {ceason }}(1, y)}^{n_{E}} \sum_{\text {seacom }}(2, y)}{t_{2}} \hat{C} w\left(t, r, i_{E}\right) \tag{7.1.17}
\end{equation*}
$$

where $i_{E}$ is the index for effort type ranging from 1 (handline) to $n_{E}$ (recreational).

## Annual egg production

The annual egg production indicator is calculated as a proportion of pristine, i.e. as a proportion of an unfished stock. The estimated total annual egg production (1) assumes a 50:50 sex ratio, (2) includes both legal size and undersize females, and (3) employs a fecundity-at-weight relation of $f m(t, c, s)=61398 \cdot[w(a(t, c), s)]^{0.9942}$ and assumes $100 \%$ maturity of all slices. The measure of pristine egg production is obtained by running the model without estimation (as a projection) with all catches set to nearly zero. A single equilibrium projected value of egg production is calculated as the value at the end of a 137 year model period, where annual recruitment is set to be fixed at the average over historical estimates from 1982-2009. The estimated annual egg production for the summer half-year of each year $y$ is given by
$\operatorname{Eggs}(y, r)=\sum_{c=\text { cohort1 }}^{\text {cohort23plus nlegs }} \sum_{s=0} 0.5 \cdot\left(\exp [-0.5 \cdot M] \cdot N\left(t_{\text {season }}(1, y), r, c, s\right)-0.5 \cdot \hat{C} n\left(t, r, c, s, i_{E}\right)\right) \cdot f m(t, c, s)$

### 7.2. Estimating Length-at-age and Weight-length

Growth, as increasing mean and standard deviation of observed body lengths for every halfyearly age was estimated from catch length-at-age samples. A normal distribution of lengths-at-age for each cohort age is assumed. The large majority of these aged Snapper were SAFCOL market samples, though some, in SGWC and GSV, were obtained by researchers. Market samples are subject to the knife-edge cut-off at the legal minimum length (LML) of 38 cm . This length cut-off in the sampling from commercial lengths-at-age was accounted for by fitting to a likelihood pdf that is truncated at 38 cm , imposing a model probability of observing a market-sampled fish below LML of zero. The few market sampled Snapper less than 38 cm were removed from this analysis. This truncation length-at-age estimation method (McGarvey and Fowler 2002) avoids growth over-estimation bias that would otherwise arise from catch
samples including in the fitted data only the faster growing individuals that grow past minimum legal size at an earlier age.

The normal likelihood pdf for each sampled fish was fitted to its observed length at age by

$$
\begin{equation*}
L_{i}=\frac{1}{\sqrt{2 \pi} \sigma\left(a_{i}\right)} \exp \left[-\frac{1}{2}\left\{\frac{l_{i}-\bar{l}\left(a_{i}\right)}{\sigma\left(a_{i}\right)}\right\}^{2}\right] \tag{7.2.1}
\end{equation*}
$$

where $l_{i}=$ length of fish $i$, and $a_{i}=$ age of fish $i$ given in half years obtained from count of its otolith annuli and an assumed birthdate of 1 January of each (year class) summer spawning. This untruncated (regular normal) likelihood pdf was applied to SARDI research length-at-age samples which were not subject to the LML cut-off.

The mean length-at-age

$$
\begin{equation*}
\bar{l}\left(a_{i}\right)=L_{\infty}\left\{1-\exp \left[-K\left(\frac{a_{i}-t_{0}}{2}\right)\right]\right\} \tag{7.2.2}
\end{equation*}
$$

was modeled by a von Bertalanffy growth formula. Seasonality in growth was not estimated due to the small number (2) of yearly time steps.

The likelihood standard deviation $\left(\sigma\left(a_{i}\right)\right)$ quantifying the residual spread of normal lengths-at-age for each half-yearly age $a_{i}$ was modelled as an allometric function of mean length:

$$
\begin{equation*}
\sigma\left(a_{i}\right)=s_{0} \cdot\left(\bar{l}\left(a_{i}\right)\right)^{s_{1}} . \tag{7.2.3}
\end{equation*}
$$

This power function for standard deviation in terms of model-predicted mean length has the desired property that as observed in the data, the spread of lengths-at-age increases with fish body size, but once growth stops, the standard deviation in lengths-at-age also ceases to change. In this assessment, $s_{1}$ was fixed to 1 , there being insufficient support for freely
estimating this exponent, implying the spread of lengths at age increases linearly with estimated mean length $\left(\bar{l}\left(a_{i}\right)\right)$.

The left-truncated normal likelihood, which applies to samples from the commercial or recreational catch,

$$
L_{i}=\left\{\begin{array}{l}
\frac{1}{\sigma\left(a_{i}\right)} \exp \left[-\frac{1}{2}\left\{\frac{l_{i}-\bar{l}\left(a_{i}\right)}{\sigma\left(a_{i}\right)}\right\}^{2}\right] /\left\{\int_{L M L}^{+\infty} \frac{1}{\sigma\left(a_{i}\right)} \exp \left[-\frac{1}{2}\left\{\frac{l-\bar{l}\left(a_{i}\right)}{\sigma\left(a_{i}\right)}\right\}^{2}\right] d l\right\}, \text { if } l_{i} \geq L M L  \tag{7.2.4}\\
0, \text { if } l_{i}<L M L
\end{array}\right.
$$

postulates a probability of zero for landed samples less than LML and a renormalised probability, integrating to 1 , for the range of legal lengths.

Parameters were estimated by minimising the negative sum of log-likelihood probabilities using the ADMB estimation software:

$$
\begin{equation*}
O=-\sum_{i=1}^{n} \ln \left(L_{i}\right) . \tag{7.2.5}
\end{equation*}
$$

The estimated length-at-age curves with associated $95 \%$ confidence intervals obtained from Eq. 7.2.3 (Figure 4-16) were taken as inputs into SnapEst. Subsequently, two key growth parameters ( $K$ and $s_{0}$ ) were further re-estimated in SnapEst, integrated with the overall stock assessment estimation. Because of the slice-partition age and length population breakdown in SnapEst, this re-estimation allows for further correction of growth bias, notably accounting for the asymmetric nature of fishing mortality which removes faster growing fish from the population at younger ages (the Rosa Lee phenomenon) when they reach legal harvestable size sooner than slower growing fish (McGarvey and Feenstra 2007). Because the fishing mortality rate on SA Snapper is generally lower, notably compared to Garfish, this growth bias correction is small.

Parameters for weight-at-length relationship were also re-estimated for this assessment. In previous years, separate weight-at-length relationships were estimated by region. Examining the separate weight-at-length curves obtained in re-analysis this year found negligibly small differences, and residuals about the fitted curve are small, implying that a single weight-atlength relationship is accurately applicable to all SA regions. Mean weight versus total length was modeled by an allometric relationship:

$$
\begin{equation*}
\bar{w}\left(l_{i}\right)=\alpha l_{i}{ }^{\beta} . \tag{7.2.6}
\end{equation*}
$$

A normal likelihood was again used. The standard deviation $\sigma_{w}\left(l_{i}\right)$ of the likelihood (i.e. of the fitted spread of observed weights about the mean $\left.\bar{w}\left(l_{i}\right)\right)$ was assumed to vary in a power relationship with model predicted weight at each given fitted total fish length applying an analogous error structure to that assumed for length-at-age in Eq. 7.2.3:

$$
\begin{equation*}
\sigma_{w}\left(l_{i}\right)=\sigma_{w 0}\left(\bar{w}\left(l_{i}\right)\right)^{\sigma_{w 1}} . \tag{7.2.7}
\end{equation*}
$$

The resulting weight-length curve (not shown) was obtained by minimising the negative loglikelihood function. The weight-length exponent $\beta$ was set equal to 2.8 , with preliminary least squares estimates by region all close to 2.8 . The maximum likelihood estimate of $\alpha$ was obtained using the TMB parameter estimation package in $R$. The final weight-at-length formula used for SnapEst in all SA regions is $\bar{w}(l)=4.00 \times 10^{-8} l^{2.8}$.

### 7.3. SnapEst Sensitivity Analysis

One important input to fishery model construction is the choice of how much weighting to place on different data sources. For the application of SnapEst in the current stock assessment, the choices related to the relative weightings to be assigned to: (i) the estimates of biomass from the DEPM surveys; (ii) the annual estimates of age composition from market sampling; (iii) and the estimates of CPUE. As such, to determine the sensitivity of the model outputs for each of the three stocks, four different weighting scenarios were tested for each: the baseline run in which the different data sources retained their default weightings; and three weightingchoice variations. The estimates of the time series of the different model output parameters, i.e. recruitment, biomass, harvest fraction and egg production (the latter not rescaled to a percentage of pristine for these sensitivities) are presented in Figures 7-1 to 7-3. How the data weightings were altered for the three alternative choices are detailed in the figure legends.

## Spencer Gulf/West Coast Stock

The sensitivity analyses that tested alternative data weightings for the SG/WCS showed robust model outcomes, where the trends and absolute levels of biomass did not vary meaningfully from the baseline run (Figure 7-1). Such small disparity among different assumed weightings of different data sources suggested that the choice of data weighting did not meaningfully affect the stock assessment conclusions and provided confidence in the model outcomes. Among the sensitivity runs, weightings set equal on all half-yearly age composition samples showed the biggest difference from baseline, where baseline weightings naturally varied in proportion to age-length sample size. These results highlighted the importance of the agecomposition data to the outputs of the SnapEst model.

In a further sensitivity test, all of the DEPM estimates for SG were removed from the model rather than just that for 2019. In this case, substantial differences in model outcomes were obtained (pink lines of Figure 7-1). The resulting model estimates of biomass were an order of magnitude higher than the estimates of biomass from the DEPM surveys and the biomass estimates of all other sensitivity scenarios (Figure 7-1). In addition, this sensitivity analysis showed strongly increasing biomass over time, which was disparate to all other assessment information that demonstrated stock depletion. Thus, for the SG/WCS, the DEPM surveys were essential for producing reliable levels of biomass in model estimates, i.e. to anchor absolute population levels and to infer recent large stock declines. This reliance on the DEPM estimate of biomass occurs as the usual overfishing paradigm is not identifiable by model inference since the commercial catches and effort are so low, yet the stock does not recover.

In SG, the model estimate of fishable biomass in 2013 was higher than the DEPM survey estimate, demonstrating larger decline from 2013 to 2018 than established from DEPM alone
(Figure 7-17, top right graph). However, over the last two summers (i.e. 2018/19 and 2019/20), model and DEPM survey estimates of SG biomass were similar (Figure 7-17).

## Gulf St. Vincent Stock

As for the SG/WCS, the sensitivity analyses for the model outcomes for the GSVS were insensitive to the different weightings on input data sources (Figure 7-2). The model outputs for the four performance indicators were similar under the four weighting scenarios, suggesting that the model results were quite robust. This includes the minimal impact on model estimates when all three estimates of biomass from the DEPM surveys were excluded as input data (Figure 7-2). That the model version from which DEPM estimates of biomass were excluded should independently estimate the same absolute biomass levels and the large decline since 2014 seen in DEPM survey estimates is strong confirmation that both are correct. This corroboration by two independent methods confirms the large decline in biomass of the GSVS between 2014 and 2018.

The trend in biomass differed between the model and the DEPM surveys in 2018 and 2019. These DEPM estimates demonstrated a stable or slight increase in biomass whereas the model determined that biomass continued to decline across these time steps (Figure 7-17). However, given that the survey in 2018 was incomplete, it remains difficult to determine from DEPM alone what occurred over this period. Therefore, inferences on recent declines were made using the model as it represents the integration of all available data.

## South East Region

From the sensitivity analyses for the SE Region there was again relatively small divergence in the model outputs from the different scenarios until around 2014 or 2015, after which they diverged considerably (Figure 7-3). For this region, for which no estimates of absolute biomass were available from DEPM surveys, the estimates of biomass and other output parameters were inferred by standard fishery modelling inference (based primarily on age proportions and total catches, supplemented by CPUE as an index of abundance). These estimates were much less precise for the latter years of the time series (error bars in Figure $4-25)$. Much of this uncertainty is due to the extended period that it takes for each cohort to fully recruit to the fishery, i.e. for all fish in the cohort to grow across the legal minimum length of 38 cm TL. Snapper in the SE Region reach 10 years of age before the cohort length-at-age distribution is fully represented in the input data of age-length samples, catch rates, and catch totals. As such, the cohort is about 10 years of age before model estimates of year class strength become precise.


Figure 7-1. Summary of results from the analyses of sensitivity of the time series of output parameters from SnapEst for the Spencer Gulf/West Coast Stock to the different scenarios of weighting among input data sources. The different scenarios considered are described in the legend. The case of 'Equal weighting on half-yearly age samples' reweights the halfyearly age composition samples so that each half-year has an equal contribution rather than half-years with more samples have a correspondingly higher weighting.


Figure 7-2. Summary of results from the analyses of sensitivity of the time series of output parameters from SnapEst for the Gulf St. Vincent Stock to the different scenarios of weighting among input data sources. The different scenarios considered are described in the legend. The case of 'Equal weighting on half-yearly age samples' reweights the halfyearly age composition samples so that each half-year has an equal contribution rather than half-years with more samples have a correspondingly higher weighting.


Figure 7-3. Summary of results from the analyses of sensitivity of the time series of output parameters from SnapEst for the SE Region to the different scenarios of weighting among input data sources. The different scenarios considered are described in the legend. The case of 'Equal weighting on half-yearly age samples' reweights the half-yearly age composition samples so that each half-year has an equal contribution rather than half-years with more samples have a correspondingly higher weighting

### 7.4. DEPM sensitivity analysis

## Methods

Daily egg production methods are known to have large imprecision which results from the combination of several parameters, that are themselves imprecise. While it is acknowledged that DEPM estimates are considered unbiased and are demonstrably capable of detecting changes in biomass, this imprecision requires sensitivity analyses to determine which parameters could influence estimates of biomass if determined inaccurately. A sensitivity analysis was performed using the 2019 surveys for the three most influential parameters in the DEPM analysis: egg density $\left(P_{0}\right)$, spawning area $(A)$ and spawning fraction $(S)$.

The sensitivity analysis for spawning area was performed by maintaining all other parameters ( $P_{0}, S, R, F_{w,} \breve{w}_{w}$ and $p_{w}^{N}$ ) at their mean values and altering the value of $A$ in the equation (Equation 6). This same process was followed for $P_{0}$ and $S$. The values included in the sensitivity analysis were determined differently for each parameter. Mean daily egg production $\left(P_{0}\right)$ was analysed by increasing and decreasing the estimated values by $50 \%$ and using the estimates from 2018. Spawning area ( $A$ ) was analysed using the estimate of each gulf in 2018, an upper limit where $A$ was the entire survey area, a $50 \%$ decrease in $A$ and for GSV, a reestimation of $A$ for the surveyed stations in 2018. Values of $S$ were included as estimate used in 2018, a spawning fraction of $100 \%$ and the minimum and maximum values of $S$ that could be estimated for each gulf given their different methods. For GSV, the $2.5 \%$ and $97.5 \%$ quantiles were calculated from the mean and standard error of the estimate. For SG, alterations to the inferences from the GSV histology were made to attain the smallest and largest value for this parameter.

## Results

The spawning biomass for GSV in 2019 was not considerably influenced by uncertainty in either $S$ or $A$. Spawning fraction has an inverse exponential relationship with spawning biomass as it is a proportional measure. Therefore when $S$ is high, estimates of spawning biomass are more certain and when it is low small inaccuracies can be highly influential for spawning biomass. As spawning fraction was estimated at 0.85 for GSV, any inaccuracy around this estimate would have little influence on spawning biomass (Figure 7-4). The minimum $S$ that could be estimated from data collected in 2019 was 0.65 which would produce a spawning biomass estimate of $1,060 \mathrm{t}$. The maximum $S$ that could be estimated from data collected in 2019 was 0.98 which would produce a spawning biomass estimate of 701 t . Very little uncertainty is present in the $A$ for GSV as eggs were present across the gulf and the kriging estimated an $A$ that was similar to the overall survey area. Therefore, the only parameter that could introduce uncertainty into the GSV spawning biomass estimate is
$P_{0}$ (Figure 7-4). This parameter has a linear relationship with spawning biomass and unlike the corresponding examples of $S$ and $A$, does not have theoretical bounds on its range. However, $P_{o}$ would need to be considerably underestimated in order for the biomass estimate to extend beyond the upper error range of 936 t (biomass estimate ( 811 t ) $\pm$ standard error (125 t)) (Figure 7-5).


Figure 7-4. Sensitivity analysis of the three most influential DEPM parameters on Snapper biomass for GSV. The red line marks the value used in the 2019 DEPM assessment. The point at which it intersects with the horizontal black line is the biomass estimate produced by it. Remaining coloured lines represent other parameter values included in the analysis.

Based on the data collected in 2019, $S$ was the most influential parameter for the spawning biomass estimate for SG (Figure 7-5). This occurred because the estimate of $S$ is 0.37 and therefore uncertainties around this value have greater influence on spawning biomass than when $S$ is high. The minimum value of $S$ that could be estimated in 2019 is 0.22 which would
produce of biomass estimate of 294 t . This is $66 \%$ larger than the spawning biomass estimated using $S=0.37$ (Figure 7-5). As with GSV, the $P_{0}$ and $A$ parameters for SG have a linear relationship with spawning biomass. Again, uncertainty in $P_{0}$ would need to be considerably underestimated in order to produce a spawning biomass outside of the upper error range of 211 t (biomass estimate ( 177 t ) + standard error (34 t)). Unlike GSV, there is no theoretical upper bound that could be considered for $A$ in SG, as the entirety of the gulf was not covered in the 2019 survey. The maximum value of $A$ that could be estimated would therefore be equivalent to entire survey area. This would approximately double the spawning biomass to 353 t but would be highly unrealistic (Figure 7-5).


Figure 7-5. Sensitivity analysis of the three most influential DEPM parameters on Snapper biomass for SG. The red line marks the value used in the 2019 DEPM assessment. The point at which it intersects with the horizontal black line is the biomass estimate produced by it. Remaining coloured lines represent other parameter values included in the analysis.

### 7.5. SnapEst model fits to data

## Fits to age proportions

In the graphs below showing the fits of age proportions to sample data (primarily from SAFCOL market), the thick line is model-predicted and the thin line is data.


Figure 7-6. Fits to age proportions for the Spencer Gulf/West Coast stock, for handline.


Figure 7-7. Fits to age proportions for the Spencer Gulf/West Coast stock, for longline.


Figure 7-8. Fits to age proportions for the Gulf St. Vincent stock, for handline.


Figure 7-9. Fits to age proportions for the Gulf St. Vincent stock, for longline.


Figure 7-10. Fits to age proportions for the SE Region, for handline and longline.

## Fits to mean length-at-age

In the graphs below showing the fits of model-predicted mean lengths of Snapper versus age, the solid markers are model and open markers are data.


Figure 7-11. Fits to mean length-at-age for the Spencer Gulf/West Coast stock, for handline.


Figure 7-12. Fits to mean length-at-age for the Spencer Gulf/West Coast stock, for longline.


Figure 7-13. Fits to mean length-at-age for the Gult St. Vincent stock, for handline.


Figure 7-14. Fits to mean length-at-age for the Gulf St. Vincent stock, for longline.


Figure 7-15. Fits to mean length-at-age for the SE Region, for handline and longline.

Fit to handline catch rate


Figure 7-16. Fits to the handline catch rate for the three regions. For the Spencer Gulf/West Coast stock (top) and Gulf St. Vincent stock (middle), the later years of CPUE have been omitted since they are not used in the model. Missing data CPUE for SE Region (bottom) are for years where the confidentiality requirement (5 licences or more) has not been met.

## Fit to DEPM



Figure 7-17. Model fits to the DEPM estimated biomass for Spencer Gulf (SG) and Gulf St. Vincent (GSV). The model biomass estimates (solid lines with grey shading) have been rescaled to include only the areas of SG and GSV covered by DEPM surveys (dotted lines with error bars). The fits of model to DEPM are shown for three model-fitting sensitivity scenarios: (1) baseline model (all years of DEPM fitted, top two graphs), (2) model with 2019 DEPM omitted (middle two graphs), and (3) model with all DEPM biomass estimates omitted from model fitting (bottom graphs). The same DEPM biomasses and their errors are shown for all three graphs, for GSV and for SG.

