Fisheries

Snapper (*Chrysophrys auratus*) Stock Assessment Report 2022

MJ Drew, TA Rogers, R McGarvey, J Feenstra, D Matthews, J Matthews, J Earl, J Smart, C Noell and AJ Fowler

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Report to PIRSA Fisheries and Aquaculture

Government of South Australia
Department of Primary Industries and Regions

SARDI
SOUTH AUSTRALIAN RESEARCH AND DEVELOPMENT INSTITUTE

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ABBREVIATIONS

Acronym	Meaning
CFL	Caudal Fork Length
CPUE	Catch Per Unit Effort
DEPM	Daily Egg Production Method
FAACC	Formalin, Acetic Acid, Calcium Chloride
FRDC	Fisheries Research and Development Corporation
GSV	Gulf St Vincent
GSVS	Gulf St Vincent Stock
IS	Investigator Strait
ISH	In Situ Hybridisation
ITQ	Individual Transferable Quota
LCF	Lakes and Coorong Fishery
MFA	Marine Fishing Area
MLL	Minimum Legal Length
MSF	Marine Scalefish Fishery
MSFMAC	Marine Scalefish Fishery Management Advisory Committee
NGSV	Northern Gulf St Vincent
NSG	Northern Spencer Gulf
NZRLF	Northern Zone Rock Lobster Fishery
PIRSA	Primary Industries and Regions, South Australia
PIRSA F&A	PIRSA Fisheries and Aquaculture
POF	Post Ovulatory Follicle
PPB	Port Phillip Bay
RNA	Ribonucleic Acid
SA	South Australia
SAASC	South Australian Aquatic Science Centre
SAFCOL	South Australian Fisherman's Co-Operative Limited
SARDI	South Australian Research and Development Institute
SE	South-East
SG	Spencer Gulf
SG/WCS	Spencer Gulf / West Coast Stock
SGSV	Southern Gulf St Vincent
SMAC	Snapper Management Advisory Committee
SSG	Southern Spencer Gulf
SZRLF	Southern Zone Rock Lobster Fishery
TAC	Total Allowable Catch
TACC	Total Allowable Commercial Catch
TARC	Total Allowable Recreational Catch
VNN	Voronoi Natural Neighbour
WA	Western Australia
WC	West Coast of Eyre Peninsula
WVS	Western Victoria Stock

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A significant component of this assessment was the collection of samples of adult fish and their processing to provide biological information. We are extremely appreciative of the continual support from the South Australian Fisherman's Co-Operative Limited (SAFCOL) Central Fish Market in Mile End, particularly Ian Mitchell and the numerous fish mongers who generously provided access to purchased Snapper. Numerous commercial MSF fishers contributed to the collection of these samples, namely Nick Faklis, Johnny Kouvaris, Kirk Kaehne, Daniel Hertz, Hadar Van Veen, David Wilks, Ray and Dave Zimmerman, Shane and Daniel Frensch, Michael Pennington, Zane Thomson, Gary Lloyd, Shane Bishop, Laurence Condo, Alex Blackley, and Noel Hayes. The samples of Snapper were processed at several commercial processing plants including Thevenard Fish Processors (Nick, Stelios, Maria, and Angela Faklis), Fergusons Australia (Kate and Andrew Ferguson), and Zimmerman Fish Processors (Ray, Dave, and Luke Zimmerman). Technical support was provided by Gretta Chaplin, Jeremy Bussell, Dave Delaine, Mark Sutcliffe, Sam Hamood-Smith, and Nat Navong. The fillets from these sampled fish were donated to Foodbank who subsequently distributed them to meet community needs.

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The report was subjected to an extensive review process. Firstly, it was reviewed internally by Dr Lachlan McLeay and Dr Gretchen Grammer of SARDI Aquatic and Livestock Sciences. It was then reviewed externally by PIRSA Fisheries and Aquaculture (Sam Stone and Dr Annabel Jones), a team at the Western Australian Department of Primary Industries and Regional Development, and by independent fisheries scientist Dr Tony Smith. We thank all reviewers for their constructive feedback which helped to improve the content and presentation of the report. Finally, it was reviewed and approved for release by Dr Mike Steer (Research Director, SARDI Aquatic and Livestock Sciences).

EXECUTIVE SUMMARY

This stock assessment for the South Australian Snapper fishery is part of the Snapper Management, Science, and Engagement Project Plan that was initiated following the changes in Snapper fishery management that were implemented in November 2019. The changes in Snapper fishery management included the closing of extensive parts of the State to Snapper harvest by all sectors. This assessment considers data from 1 January 1984 to 31 January 2022, encompassing two years of the implemented management arrangements.

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 from South Australia fisheries are summarised; however, stock status is determined by the Victorian Fisheries Authority (VFA) since this regional population is a part of the Western Victorian Stock (WVS). Stock status for the SG/WCS and GSVS was determined using a weight-of-evidence approach following the National Fishery Status Reporting Framework (Piddocke et al. 2021).

The stock assessments used both fishery-dependent and fishery-independent data. The fishery-dependent data included commercial fishery statistics, the 'general' fishery performance indicators, and population length and age structures from targeted adult sampling and commercial market sampling. Fishery-independent data were estimates of spawning biomass derived from the application of the daily egg production method (DEPM) for Spencer Gulf (SG) and Gulf St Vincent (GSV).

The SnapEst model was applied to each biological stock and the SE Region to integrate fishery-dependent, fishery-independent, and biological datasets to produce a time series of 'biological' fishery performance indicators. Collectively, the indicators of (i) fishable biomass (all Snapper above legal size), (ii) recruitment, (iii) harvest fraction, and (iv) egg production underpinned this stock assessment.

Spencer Gulf / West Coast Stock

The age structures for each of the Northern Spencer Gulf (NSG) and Southern Spencer Gulf (SSG) components of the SG/WCS throughout the 2000s and 2010s indicated the absence of strong year classes since the late 1990s, demonstrating over two decades of poor recruitment.

The estimate of spawning biomass for SG calculated using the daily egg production method (DEPM) indicated ongoing decline since 2013. In 2021, the spatial distribution of eggs was heavily contracted compared to previous surveys. This resulted in a considerable decrease in spawning area and a proportional low estimate of spawning biomass.

The time series of fishable biomass from SnapEst showed a decline of 91% from 5,244 t (± SE; 103) in 2005 to its lowest recorded level of 469 t (± SE; 53) in 2020. The estimated fishable biomass of 543 t (± SE; 65) in 2022 represents a stabilisation at historically low levels. Model estimated egg production in 2022 was the equal lowest on record and was 2% of pristine egg production. Model outputs confirmed the low levels of recruitment throughout the 2000s and 2010s.

SnapEst modelling indicated that the fishable biomass of the SG/WCS in 2022 is depleted and that recruitment is impaired. The age structure of the population in NSG remains truncated and there is not yet evidence of measurable improvements in the stock following implementation of management changes in 2019. Consequently, the SG/WCS remains classified as '**depleted'** (Table E-1).

Gulf St Vincent Stock

The age structures for the Northern Gulf St Vincent (NGSV) component of the GSVS involved several year classes, including the 2007- and 2009-year classes that have persisted since the early 2010s. Recent age structures in both NGSV and Southern Gulf St Vincent (SGSV) show the emergence and persistence of the 2014-year class in both regional populations. However, the lack of strong year classes since 2009 show over a decade of poor recruitment.

In 2021, the estimate of spawning biomass calculated using the DEPM indicated ongoing biomass decline. The spatial distribution of Snapper eggs in 2021 was heavily contracted compared to previous surveys. This resulted in a considerable decrease in spawning area and a proportional decrease in estimated spawning biomass.

Estimates of fishable biomass from SnapEst increased rapidly from low values (~850 t) in the 1990s to a record high level of 4,300 t (± SE; 104) in 2011, before declining by 92% to the lowest recorded level of 343 t (± SE; 67) in 2020. The model estimate of fishable biomass was relatively stable from 2020 and was estimated at 368 t (± SE; 79) in 2022. Model-estimated egg production in 2022 remained at the lowest level on record at 2% of pristine egg production. Model outputs confirmed the low recruitment since 2009.

SnapEst modelling indicated that fishable biomass and recruitment of the GSVS are near the lowest recorded levels. Despite evidence of the persistence of the 2014-year class, biomass is depleted, recruitment is still considered impaired, and there is not yet evidence of measurable improvements in the stock following implementation of management changes in November 2019. Consequently, the GSVS remains classified as '**depleted'** (Table E-1).

South-East Regional Population

Commercial catches, effort and catch rates increased rapidly in the SE Region between 2008 and 2010. In 2010, total catch peaked at 271.6 t and then declined back to 4.8 t in 2016. Thereafter, catches increased to 57.9 t in 2020 and were 31.6 t in 2021, which were constrained by TACs. Estimates of fishable biomass from SnapEst follow a similar trend to the commercial fishery statistics. Estimated fishable biomass increased by 105% from 170 t (± SE; 45) in 2015 to 349 t (± SE; 70) in 2022, which reflected recruitment of the strong 2013- and 2014-year classes. It is also anticipated that the 2018-year class, which is expected to be strong given the record high level of recruitment to Port Phillip Bay in 2018, will soon recruit to the fishery.

The South-East Region is the western extremity of the Western Victorian Stock (WVS). Stock status for the WVS was determined in 2020 (Conron et al. 2020) and in 2021 as part of the Status of Australian Fish Stocks (SAFS) (Piddocke et al. 2021). The WVS was classified as '**sustainable**' in 2021.

Table E-1. Key statistics and results of this report for the Spencer Gulf / West Coast Stock, Gulf St Vincent Stock, and the South-East Region in 2022. Values in parentheses are standard error. * The South-East Region is part of the Western Victoria Stock (WVS) which is assessed by the VFA and was classified as sustainable in 2021 (Piddocke et al. 2021).

Stock	Model estimated stock biomass (t)	Harvest fraction (2021)	Stock status
Spencer Gulf / West Coast Stock	543 (± 65)	0%	Depleted
Gulf St Vincent Stock	368 (± 79)	0%	Depleted
South-East Region	349 (± 70)	11%	Sustainable*

Keywords: Daily Egg Production Method, spawning biomass, SnapEst, recruitment variability, fishery closure.

1. GENERAL INTRODUCTION

1.1. Biology and Stock Structure

Taxonomy & distribution

Snapper (*Chrysophrys auratus*) is a large-bodied, demersal, marine fish species in the family Sparidae that is broadly distributed throughout temperate and sub-tropical waters of the Indo-Pacific region. This includes a continuous distribution around the southern coastline of mainland Australia and northern Tasmania (Kailola et al. 1993, Last et al. 2011). Throughout its distribution, Snapper occur across a diversity of habitats extending from shallow, coastal bays and estuaries to the edge of the continental shelf, and across a depth range of 1–200 m. It is a long-lived species, estimated to live up to 36 years in South Australia (SA) and up to 40 years in other parts of Australia (McGlennon et al. 2000, Norriss and Crisafulli 2010).

Life history

Snapper conforms to a bipartite life history which is characteristic of most demersal marine fish species, i.e., it has a pelagic egg and larval phase followed by a demersal juvenile and adult phase (Cowen and Sponaugle 2009). Snapper is a multiple batch spawning species with indeterminate fecundity and asynchronous oocyte development (McGlennon 2003, Saunders 2009). In SA, Snapper has a protracted spawning period that extends from November to February, peaking in December and January (McGlennon 2003, Saunders et al. 2012). Mature fish form dense aggregations prior and during the spawning period and can spawn repeatedly over consecutive days (Scott et al. 1993). In general, spawning commences in the evening and continues into the night (Saunders 2009, Wakefield 2010). The resultant eggs hatch after ~36 hours at 21°C and release larvae of ~2 mm body length (Pecl et al. 2014). The larvae reside in a mixed planktonic community for 20-30 days until they undergo settlement to the demersal environment and become juveniles at 10-12 mm standard length (Fowler and Jennings 2003, Hamer and Jenkins 2004). Nursery areas for Snapper are characterised by soft, silty, benthic substrate that supports appropriate planktonic prey. Juveniles initially remain in the vicinity of their nursery areas for several months after settlement and then transition to more complex habitats as they develop. Juvenile Snapper develop quickly and reach sexual maturity at 3–4 years of age and 25–35 cm fork length (Pecl et al. 2014).

Recruitment variability

The population dynamics and fishery productivity of Snapper in SA are fundamentally driven by high inter-annual variation in recruitment of the 0+ year class (McGlennon and Jones 1997, Fowler and Jennings 2003, Fowler and McGlennon 2011). Snapper is a highly fecund species that releases large numbers of very small and vulnerable eggs that experience extremely high mortality rates (McGlennon et al. 2000). As such, small variations in egg and larval

survivorship can result in significant variation in recruitment, and subsequently, the abundance of juvenile and adult fish (Doherty and Fowler 1994, Houde 2008, Murphy et al. 2013). This temporal variation in recruitment is manifested in the age structure of the population. Snapper populations in SA are characterised by distinct strong and weak age classes that are consistent amongst years, relating to inter-annual variation in recruitment of age 0+ juveniles and the resulting variation in year-class strength (McGlennon et al. 2000, Fowler and Jennings 2003). The occasional strong year class sustains the fishery through periods of time that are characterised by poor to average recruitment (McGlennon et al. 2000, Fowler et al. 2017a).

Stock structure

The stock structure for Snapper throughout its Australian distribution is complex, as there are considerable differences in the spatial scales over which populations are divisible into separate stocks (Fowler et al. 2016a, 2017a, Bertram et al. 2022, Gardner et al. 2022). For the coastal waters of SA, a recent study on demographic processes and stock structure identified three ecologically discrete stocks: the Spencer Gulf / West Coast Stock (SG/WCS), the Gulf St Vincent Stock (GSVS), and the Western Victoria Stock (WVS) (Fowler 2016, Fowler et al. 2017a). Each stock is likely to depend on recruitment into a primary nursery area.

The SG/WCS includes Spencer Gulf and all waters westward to the WA-SA border (Figure 1-1). The primary nursery area for the SG/WCS is Northern Spencer Gulf (NSG), which experiences highly variable inter-annual recruitment (Fowler and Jennings 2003, Fowler et al. 2010). The density dependent emigration of Snapper from NSG occurs as fish of a few years of age disperse and replenish the populations in Southern Spencer Gulf (SSG), and episodically in years of exceptionally high recruitment, the west coast of Eyre Peninsula (WC) (Fowler et al. 2017a).

The GSVS encompasses all waters of Gulf St Vincent and Investigator Strait (Figure 1-1). The exact location of the nursery area(s) for the GSVS has not yet been determined, but it is evident that recruitment is variable and there is density dependent dispersion throughout Northern Gulf St Vincent (NGSV) and Southern Gulf St Vincent (SGSV).

The WVS is a cross-jurisdictional stock that extends westward from Wilsons Promontory, Victoria, along south-eastern SA to a dynamic boundary near the Murray Mouth (Figure 1-1). The south-east region of SA (SE Region) is part of the WVS and the primary nursery area for this stock is Port Phillip Bay (PPB), Victoria. The abundance of Snapper in the SE Region varies episodically as fish of a few years of age emigrate from PPB in what is likely a density dependent process that relates to inter-annual variation in recruitment of the 0+ fish (Hamer and Jenkins 2004, Hamer and Conron 2016, Fowler et al. 2017a).

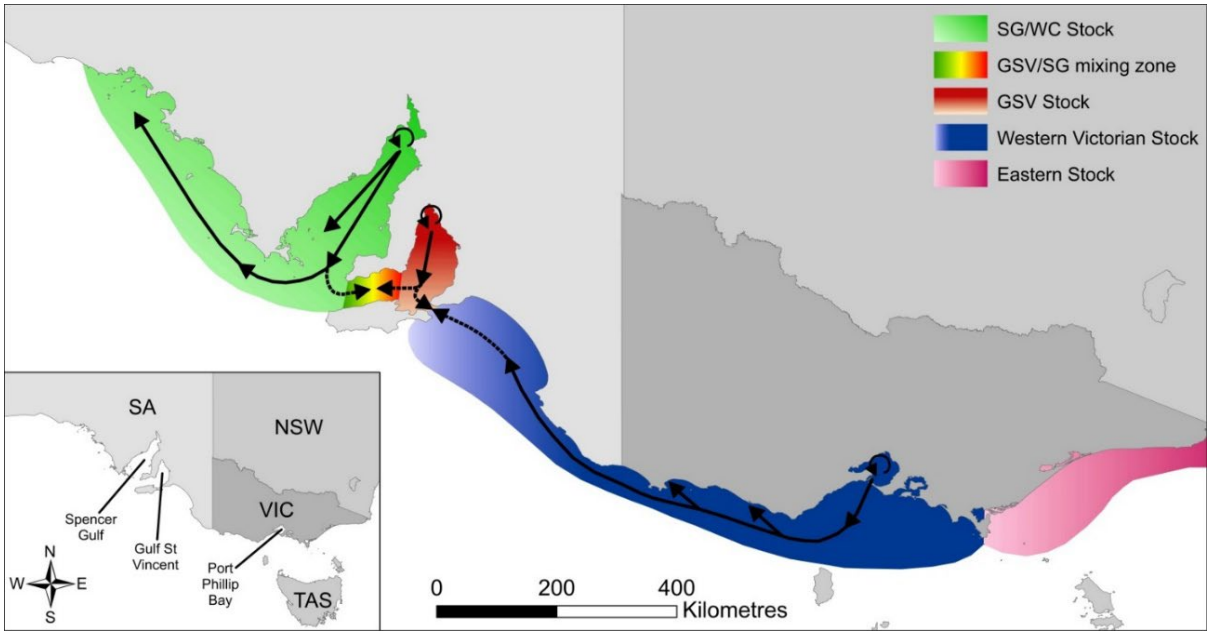

Figure 1-1. Map of the coast of south-eastern Australia showing the stock structure for Snapper based on implied fish movement (Fowler 2016, Fowler et al. 2017a). The arrows indicate directions and extent of emigration of fish from primary nursery areas in northern Spencer Gulf, northern Gulf St Vincent, and Port Phillip Bay, Victoria. Inset shows the broader geographic region. SG – Spencer Gulf, GSV – Gulf St Vincent, WC – west coast of Eyre Peninsula.

1.2. Fishery

Snapper is an iconic fishery species and important fishery resource in each mainland State of Australia. 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 from either Spencer Gulf (SG) or Gulf St Vincent (GSV) (Fowler et al. 2016a, 2019, 2020, Steer et al. 2018a, 2018b).

In SA, Snapper is a primary target species of the commercial, charter boat, and recreational fishery sectors (PIRSA 2013). Licence 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. The use of hauling nets for taking Snapper was prohibited in 1993. For recreational fishers, Snapper has been an important species because of fishers' propensity to target the large trophy fish using rods and lines primarily from boats, although jetty and land-based catches do occur (Fowler et al. 2016a). From the most recent recreational fishing survey in 2013/14, the relative proportions of the total catch taken 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, but these declined considerably through the late 2000s. Concurrently, the catches and catch rates in NGSV and the SE Region increased to unprecedented levels (Fowler et al. 2019, 2020). For the three stocks, these changes in fishery statistics reflected changes associated with different, independent, demographic processes relating to recruitment and adult migration (Fowler 2016, Fowler et al. 2017a, 2019, 2020). From 2011, the changes in the spatial structure of the fishery and stock status caused considerable concern for fishery management. This resulted in a succession of management changes that were implemented to limit commercial catches and to maximise the opportunities for spawning and recruitment (Fowler 2016). By late 2019, it was apparent that the management changes had not arrested the decline in the statuses of the SG/WCS and GSVS, which led to the implementation of stringent, spatial fishery closures. Furthermore, since 2011, several research projects funded by the Fisheries Research and Development Corporation (FRDC) were undertaken to: (i) identify the demographic processes responsible for the spatial changes at the regional scale for the fishery (FRDC 2012-020, Fowler 2016); (ii) develop a fishery independent index of fishable biomass (FRDC 2014-019, Steer et al. 2017); and (iii) develop an appropriate sampling strategy to monitor inter-annual variation in recruitment (FRDC 2019-046).

1.3. Harvest Strategy

The harvest strategy for Snapper that is outlined in the current Management Plan relates to the challenges for managing the fishery that occurred up to 2013 (PIRSA 2013). That harvest strategy involved a watching brief until two of the 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 completion of these projects was superseded by the management deliberations that took place throughout 2018 and 2019 in response to the depleting/depleted classifications of stock status to that time.

In the future, development of a replacement Management Plan including a new harvest strategy for Snapper will take into consideration the improved understanding of the biology and population dynamics of Snapper that resulted from the FRDC projects. The current harvest strategy will be revised to provide greater certainty for sustainable fishery management by developing explicit decision rules about management responses. This work will consider the Snapper fishery in the context of the significant fishery closures since

November 2019, as well as the restructure of the MSF that was implemented in 2021, which recognised Snapper as a Tier 1 species that will be managed using a total allowable commercial catch (TACC) which has been shared amongst commercial licence holders in the form individual transferable quota (ITQ) (Smart et al. 2022).

1.4. Recent History of SA's Snapper Fisheries

1.4.1. Changes to stock status

To provide context for the 2022 assessment, this section summarises the changes that occurred in the fishery throughout the 2000s, including the spatial and temporal trends in fishery productivity that resulted in changes to stock status and management regimes.

Over the past 20 years there have been significant changes in the spatial structure of SA's Snapper fishery. Historically, the SG/WCS made the main contribution to the State's catches (Fowler et al. 2020). Up to the early 2000s, these were dominated by catches from NSG, which at the time was the most productive regional Snapper fishery in Australia (Fowler et al. 2010). From 2005 to 2008, SSG produced the highest regional catches in SA. From 2007 onwards, the catches from both NSG and SSG declined considerably. Concurrently, the catches and catch rates in NGSV increased exponentially to unprecedented levels, before they also declined from 2015 onwards. A challenge in assessing catch rates is the 'hyperstability' in LL CPUE identified for both stocks. High catch rates can persist whilst fishers target aggregated fish. Therefore, reductions in CPUE may underestimate declines in biomass, and a rapid reduction in CPUE may only occur once stocks have become depleted (Fowler and McGlennon 2011, Fowler et al. 2019). In 2019, for the first time, SGSV produced the highest regional commercial catch of Snapper in SA. For the SE Region between 2004 and 2010, fishery catches increased exponentially but then declined back to a very low level by 2016 (Fowler et al. 2019, 2020).

The significant changes in spatial structure of SA's Snapper fishery caused considerable concerns about sustainability. These concerns primarily came from the declines in catches and catch rates for NSG and SSG and the substantial increases in longline fishing effort for NGSV. From 2012 onwards, this prompted a succession of management interventions which involved: (1) the introduction and subsequent reductions in daily commercial trip limits; (2) increased restrictions for commercial fishing gear; (3) changes to recreational bag and boat limits; (4) extension of the duration of the State-wide annual, seasonal closure of the Snapper fishery; and (5) introduction of spatial spawning closures in both gulfs (Fowler et al. 2016a, 2019). Despite these management interventions, the stock statuses continued to deteriorate to 2019, when the fishery closures were implemented (Table 1-1). The management changes from 2012 to 2020 are described comprehensively in Fowler et al. (2020).

The status of the SG/WCS in 2012 and 2013 was 'transitional depleting', which reflected significant declines in commercial catches and catch rates for NSG and SSG (Table 1-1) (Fowler et al. 2013). This status was retained in 2016 and 2017, as commercial catches, effort and CPUE continued to decline (Fowler et al. 2016a, Steer et al. 2017). In the assessment in 2018, the status of this stock was further downgraded to 'depleted', reflecting that commercial fishery statistics to 2017 remained at historically low levels (Steer et al. 2018b). The declines in fishery productivity and stock status primarily reflected poor recruitment throughout the 2000s, which indicated that recruitment had become impaired.

The GSVS was classified as 'sustainable' from 2012 to 2017 based on record high catches and catch rates (Fowler et al. 2013, 2016a, Steer et al. 2017). These reflected the recruitment of several strong year classes throughout the 2000s. The status of 'sustainable' was retained in 2018, although there were declines in fishery performance indicators that suggested that biomass had begun to fall (Steer et al. 2018b).

The WVS, the cross-jurisdictional stock that includes the SE Region, was classified as 'sustainable' in each year from 2012 to 2019 (Table 1-1). This reflected the recruitment of several strong year classes to PPB between 2004 and 2014 (Hamer and Conron 2016, Fowler et al. 2020).

Table 1-1. Summary of stock statuses for South Australia's Snapper Stocks from 2012 to 2021.

Year	Spencer Gulf / West Coast Stock	Gulf St Vincent Stock	Western Victorian 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
2020	depleted	depleted	Sustainable
2021	depleted	depleted	Sustainable

In late 2018, in response to the deterioration in stock status for 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 fishery managers, scientists, and representatives of the commercial, recreational and charter boat sectors of the MSF. To ensure that this 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 the DEPM for each of NSG and GSV. The resulting stock assessment in 2019 concluded that the SG/WCS remained 'depleted', based on a lack of

recovery in fishery statistics and a low estimate of spawning biomass (Table 1-1). Furthermore, the status of the GSVS was downgraded to 'depleting', following further declines in fishery statistics and a low estimate of spawning biomass (Fowler et al. 2019).

In response to the deteriorated levels of stock status in 2019, stringent fishery closures were implemented from 1 November 2019 for the SG/WCS and GSVS. Furthermore, the management response involved establishing the Snapper Management, Science, and Engagement Project Plan. This plan included a suite of 19 projects that were designed to ameliorate the effects of the fishery closures on the MSF and to maintain a flow of biological information that would inform stock status. This plan included the stock assessment that was delivered in 2020, which assigned stock status to the SG/WCS and GSVS based on fishery-dependent and fishery-independent information that was assessed against fishery performance indicators (Fowler et al. 2020). The assessment contained several significant components: (i) regional adult sampling that provided updated information on population structure; (ii) DEPM surveys in each of NSG and GSV that provided estimates of spawning biomass; (iii) and the re-development of the fishery assessment model SnapEst that integrated all fishery and biological information to inform stock status. The assessment determined that the SG/WCS remained 'depleted' (Table 1-1). It also determined that the status of the GSVS had further deteriorated necessitating its reclassification to 'depleted' (Fowler et al. 2020)

1.4.2. Changes to fishery management

In response to the declining trends in stock status for the SG/WCS and GSVS from 2012 to 2019 (Table 1-1) (Fowler et al. 2019), previous management regulations were superseded by a total Snapper fishing closure for all sectors (commercial, charter and recreational) that was imposed for the waters of the WC, SG, and GSV from 1 November 2019 to 31 January 2023. The total fishery closure prohibited the targeting and take of Snapper, including tag and release activities. Consequently, the SG/WCS and GSVS have been closed to fishing since the previous assessment. The management regulations prior to the fishery closure are detailed in Fowler et al. (2020).

The SE Region remained open to Snapper fishing and, in addition to other regulations, catches were managed under a Total Allowable Catch (TAC). The TAC was divided among the commercial, charter, recreational, and Aboriginal / Traditional sectors according to the allocations defined in the Management Plan (PIRSA 2013). Since February 2020, there have been three discrete fishing seasons in the SE Region that were managed under different TACs: 1 February to 31 October 2020 (2020 fishing season), 1 February to 30 June 2021 (2021 fishing season), and 1 July 2021 to 30 June 2022 (2021/22 fishing season) (Table 1-2). The differences in the timing and duration of fishing seasons related to the removal of a

temporal spawning closure over summer and the transition to reporting by financial year following the implementation of the MSF reform on 1 July 2021 (Smart et al. 2022). The TAC for 2020 was set at 75,000 kg as recommended by the Snapper Management Advisory Committee (SMAC) based on a recent time series of commercial catches and was divided among sectors following the State-wide allocations in the Management Plan (PIRSA 2013). The TACs for the 2021 and 2021/22 fishing seasons were set based on a harvest fraction of the estimated fishable biomass for the SE Region from the SnapEst model. For the 2021 fishing season, the TAC was set at 26,667 kg (pro-rata for five of the possible nine-month season) and divided among sectors following the State-wide allocations. For the full 2021/22 fishing season, the TAC was set at 48,000 kg and divided among sectors according to the regional distribution of sector allocations (Table 1-2) (Smart et al. 2022).

Table 1-2. Allocations of the total allowable catch (TAC) for Snapper in the SE Region for the fishing periods from 1 February 2020 to 30 June 2022. Values are weight in kg.

Time period	TAC	Commercial	Charter	Recreational	Aboriginal / Traditional
1 Feb 2020 – 31 Oct 2020	75,000	60,750	7,500	6,000	750
1 Feb 2021 – 30 June 2021	26,667	21,600	2,667	2,133	267
1 July 2021 – 30 June 2022	48,000	36,000	6,401	5,119	480

Aside from the TAC and minimum legal length (MLL) of 38 cm total length, management regulations for Snapper in the SE Region since the last assessment differed between sectors. Management regulations for the commercial sector involved a suite of input and output controls (PIRSA 2013, 2014). The four commercial fisheries (i.e., MSF, SZRLF, NZRLF, and LCF) with access to Snapper have each been limited-entry fisheries for many years, thereby limiting the numbers of fishers who could target Snapper. Commercial line fishers were limited with respect to the numbers of lines and hooks per line that they could use. A maximum of 400 hooks were permitted per vessel at any one time. In 2020 and 2021 there was a daily trip limit of 350 kg and a possession limit of 1050 kg (for a trip of more than 1 day in 2020; for a trip of 3 days in 2021). Consistent with statutory reporting requirements, commercial fishers were required to report all Snapper caught prior to landing and on mandatory catch and disposal records. Several aspects of fishery management changed following the implementation of MSF reforms on 1 July 2021. This included the regionalisation of the fishery, regional distribution of State-wide sector allocations, and the introduction of individual transferable quota (ITQ) management (Smart et al. 2022).

From 1 February 2020, a Total Allowable Recreational Catch (TARC) was applied to the recreational and Charter Boat sectors in addition to bag and boat limits. Catch and release fishing was prohibited and all undersized or incidentally caught Snapper were required to be

released using a release weight. In 2020, both sectors were managed through a harvest tag system with compulsory catch reporting. For all retained Snapper, a harvest tag was immediately attached through the fish's mouth once it was removed from the water and the tag could not be removed until it was processed or consumed. From 1 February 2021, the recreational sector transitioned to mandatory reporting through the SA Fishing Mobile Application or via FishWatch. The Charter boat sector continued with the harvest tag system until March 2022, when it also changed to mandatory reporting through the SA Fishing App or FishWatch.

1.5. Objectives of this Report

This report: (i) summarises the data that were used to determine stock status; (ii) assesses the status of the resource; (iii) identifies areas of uncertainty; and (iv) identifies future research needs. Stock status was determined using the National Fishery Status Reporting Framework (Table 1-3) (Piddocke et al. 2021), which is consistent with the South Australian fisheries harvest strategy policy (PIRSA 2015).

Table 1-3. Terminology for the status of key Australian fish stocks reports (Piddocke et al. 2021).

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

1.6. Report structure

This assessment used a weight-of-evidence approach to determine stock status that considered both fishery-dependent and fishery-independent information (Fowler et al. 2019, 2020). The fishery-dependent data were: (i) commercial fishery statistics (Chapter 2); (ii) recreational fishery data; and (iii) population size and age structures determined through commercial market sampling or targeted fishing by commercial fishers (Chapter 3). 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) (Chapter 4). Each of these data sources were incorporated into the SnapEst Stock Assessment Model (Chapter 5) which estimated four biological fishery performance indicators that are specified in the Management Plan (PIRSA 2013). For each stock (i.e., SG/WCS, GSVS, and the SE Region) the model produced annual estimates of fishable biomass, recruitment, harvest fraction, and egg production that were assessed against prescribed trigger reference points (Chapter 6). For the SE Region (i.e., the only population open to fishing since the previous assessment), fishery statistics were compared against the reference period of 1984 to 2021 and assessed using a series of general fishery performance indicators. The performance of the SG/WCS, GSVS, and SE Region were determined based on the assessment of the fishery performance indicators, which underpin the weight-of-evidence approach to assign stock status (Chapter 7) (Figure 1-2).

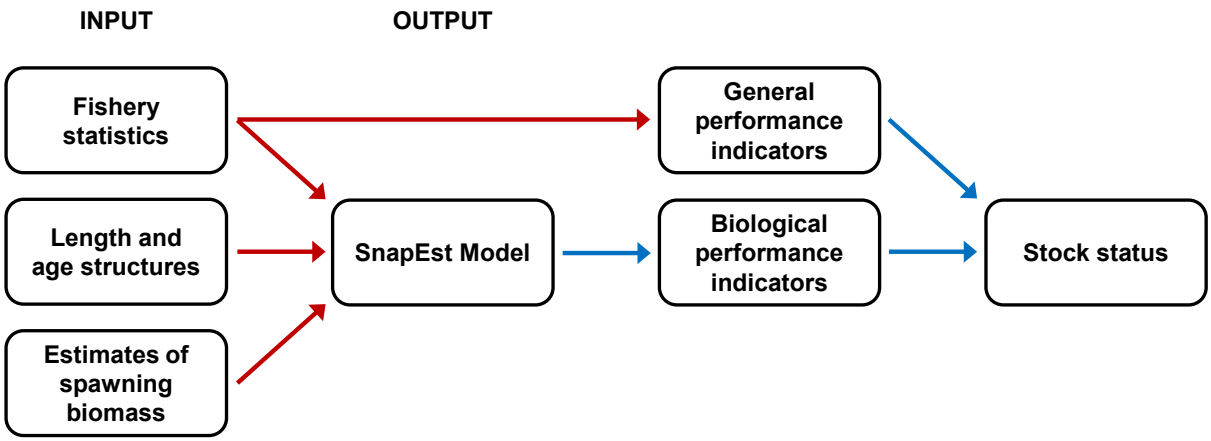

Figure 1-2. Flowchart showing the structure of the report and the weight-of-evidence approach used to determine stock status.

2. FISHERY STATISTICS

2.1. Introduction

Fishery statistics provide a means for measuring the impact of fishing mortality on an exploited fish population. Analysis of trends in fisheries statistics can identify changes in population dynamics, such as an increase in biomass or overfishing. Historic patterns in state-wide annual Snapper catch increased from ~150 t in the early-1950s to ~500 t in the early-1970s (McGlennon 2003). Annual catches then fluctuated between 300 to 500 t between the mid-1970s and mid-1980s (McGlennon 2003). Since 1984, commercial fishers in South Australia's MSF have been required to report monthly catch and effort statistics in logbooks. In 2003, the fishery transitioned to reporting daily catch and effort statistics (but still submit monthly logbook returns) after a suite of management and reporting changes were introduced. Fishery statistics derived from these catch and effort returns are used to assess the fishery against performance indicators relative to reference points prescribed in the Management Plan (PIRSA 2013). Catch and effort information reported in this assessment was also integrated into the fishery assessment model SnapEst (Chapter 5). This assessment is the first in the series of Snapper assessments to have data relating to recreational Snapper catches from mandatory reporting in the SE Region, which have been included in this Chapter and integrated into the SnapEst model.

2.2. Methods

Catch and effort data from 1 January 1984 to 31 December 2021 were examined for the four commercial fisheries with access to Snapper (i.e., MSF, NZRLF, SZRLF, LCF). This included records of location by MFA, gear type and units, daily catch weights, and the number and weight of Snapper caught. At the State-wide scale, annual estimates by calendar year were calculated for total catch, effort, and CPUE, which were differentiated by the two main gear types of handlines (HL) and longlines (LL). For each of the three stocks (i.e., SG/WCS, GSVS, and SE Region), annual estimates of total catch, targeted catch, effort, and CPUE by gear type were calculated, along with the number of fishers taking and targeting Snapper. Catch per unit effort was calculated as kg.fisherday^{-1} for the SG/WCS and GSVS. For the SE Region, an alternative metric (kg/per hook for LL) of CPUE was calculated. All CPUE timeseries are currently unstandardised. Furthermore, following the introduction of daily trip limits in 2012, two additional fishery performance indicators were prescribed in the Management Plan, i.e., the proportions of daily fishing trips where HL and LL catches were ≥250 kg (PIRSA 2013). These indicators were developed to show the proportion of targeted daily catch catches by a gear type that exceeded 250kg per day. Daily catches prior to 2012 were unrestrained by daily catch limits. These performance indicators were later revised to proportions of catches ≥200

kg (i.e., Prop200kgTarHL and Prop200kgTarLL), reflecting the reduction in the daily trip limit for the SG/WCS that was implemented in December 2016 (Fowler et al. 2019). Annual estimates of these two performance indicators were based on the commercial daily catch and effort data available in the MSF Information System (MSFIS) for calendar years from 2004 to 2021. For each year, only the targeted catch data from February to October were considered to remove the influence of the seasonal closure from the data (PIRSA 2013).

Catch and effort data for the SG/WCS and GSVS were assessed at the biological stock level, consistent with previous assessments (Fowler et al. 2019, 2020). The SE Region was assessed at the scale of post-reform management zone to align with TACs set for this region (Figure 2-1, Table 2-1). MFA 44B is included in both the GSVS and SE Region for this assessment, as historically MFA 44B was included in the GSV biological stock and was recently reallocated into the SE Region through the regionalisation of the MSF post reform. Data from the Charter Boat sector operation in the SE Region were obtained from the MSFIS. Daily records of the number and estimated weight of retained Snapper were analysed for the calendar years 2020 and 2021. Monthly recreational catch data were sourced from PIRSA Fishery Compliance. Recreational catch estimates (i.e., total weight) for the SE Region were based on the reported number of Snapper retained multiplied by the average weight for the region (i.e., 1.98 kg, derived from SARDIs Snapper biological information database). Estimates of recreational catches in previous years were sourced from recreational fishing surveys in 2000/01 (Henry and Lyle 2003), 2007/08 (Jones 2009), and 2013/14 (Giri and Hall 2015).

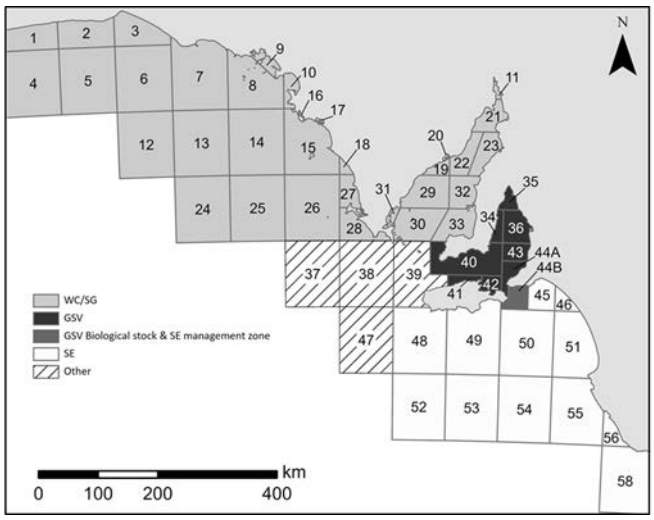

Figure 2-1. Map of the 58 Marine Fishing Areas (MFAs) of South Australia's Marine Scalefish Fishery (MSF), showing the reporting regions for the three Snapper stocks. The Spencer Gulf / West Coast Stock (SG/WCS) and Gulf St Vincent Stocks (GSV) were reported at the biological stock level, and the South-East Region (SE) was reported at the post-MSF reform management zone scale. 'Other' reflects MFA blocks that are not included the biological stocks or management zones used in this assessment.

Table 2-1. The three Snapper stocks considered in this assessment and the Marine Fishing Areas (MFAs) that comprise them. MFA 44B is included in both GSVS and SE Region for this assessment, previously MFA 44B was included in the GSV biological stock and was recently reallocated into the SE management zone through the regionalisation of the MSF post reform.

Stock	Abbrev.	Type	Marine Fishing Areas (MFAs)
Spencer Gulf / West Coast	SG/WCS	Biological	1-33
Gulf St Vincent Stock	GSVS	Biological	34-36, 40-43, 44A, 44B
South-East Region	SE	Management	44B, 45, 46, 48-58
Other			37-39, 47

2.3. Results

2.3.1. State-wide

Estimates of total State-wide commercial catch have fluctuated over varying time scales. Since 2003, State-wide catch increased to a record level of 1,035 t in 2010, before declining to 252 t in 2019 (Figure 2-2). In 2020 and 2021, catch declined to record low levels with all landings coming from the SE Region, due to the fishery closures for SG/WCS and GSVS. Furthermore, catches in the SE Region for 2020 and 2021 were constrained by TACCs (Table 1-2).

Historically, HL was the most significant gear type used to target Snapper, with HL catches accounting for the variation in total catch until 2008. The contribution of LL to total catch increased between 2005 and 2010, when it became the dominant gear type. Both HL and LL catches have declined since 2010. In 2021, 98.7% of the total catch was caught by LL.

From the mid-1980s to 2008 there was a long-term declining trend in total commercial fishing effort for Snapper (Figure 2-2). This was followed by a period of increased effort between 2009 and 2012 that corresponded to the increase in LL effort. LL effort declined from 2010, complementing the declining trend in HL effort since 2002. As such, the total fishing effort of 4,336 fisher-days in 2019 (pre-closure) was the lowest recorded since 1984. In 2021, total fishing effort was 327 fishery-days.

State-wide HL CPUE fluctuated between 1984 and 2007 but demonstrated a general long-term increasing trend (Figure 2-2). From 2007 it decreased considerably, concomitant with the emerging increase in LL effort. In contrast, LL CPUE increased substantially between 2004 and 2015, before declining each year between 2016 and 2019. The LL CPUE for 2021, estimated from the SE Region only, was 101.8 kg.fisher-day^{-1}, which is the third highest on record. The total number of fishers who reported taking Snapper declined consistently from 444 in 1987 to 244 in 2000 (Figure 2-2). Fisher number stabilised for a decade before declining from 260 in 2010 to 164 fishers in 2019. The number of fishers that targeted Snapper varied similarly and fell from 201 in 2009 to 122 in 2019. In 2021, a total of 16 fishers reported taking Snapper, of which 15 fishers reported targeting Snapper. In 2021, 80.7% of the total

commercial catch of Snapper was taken by the MSF, with SZRLF accounting for the remaining 19.3%.

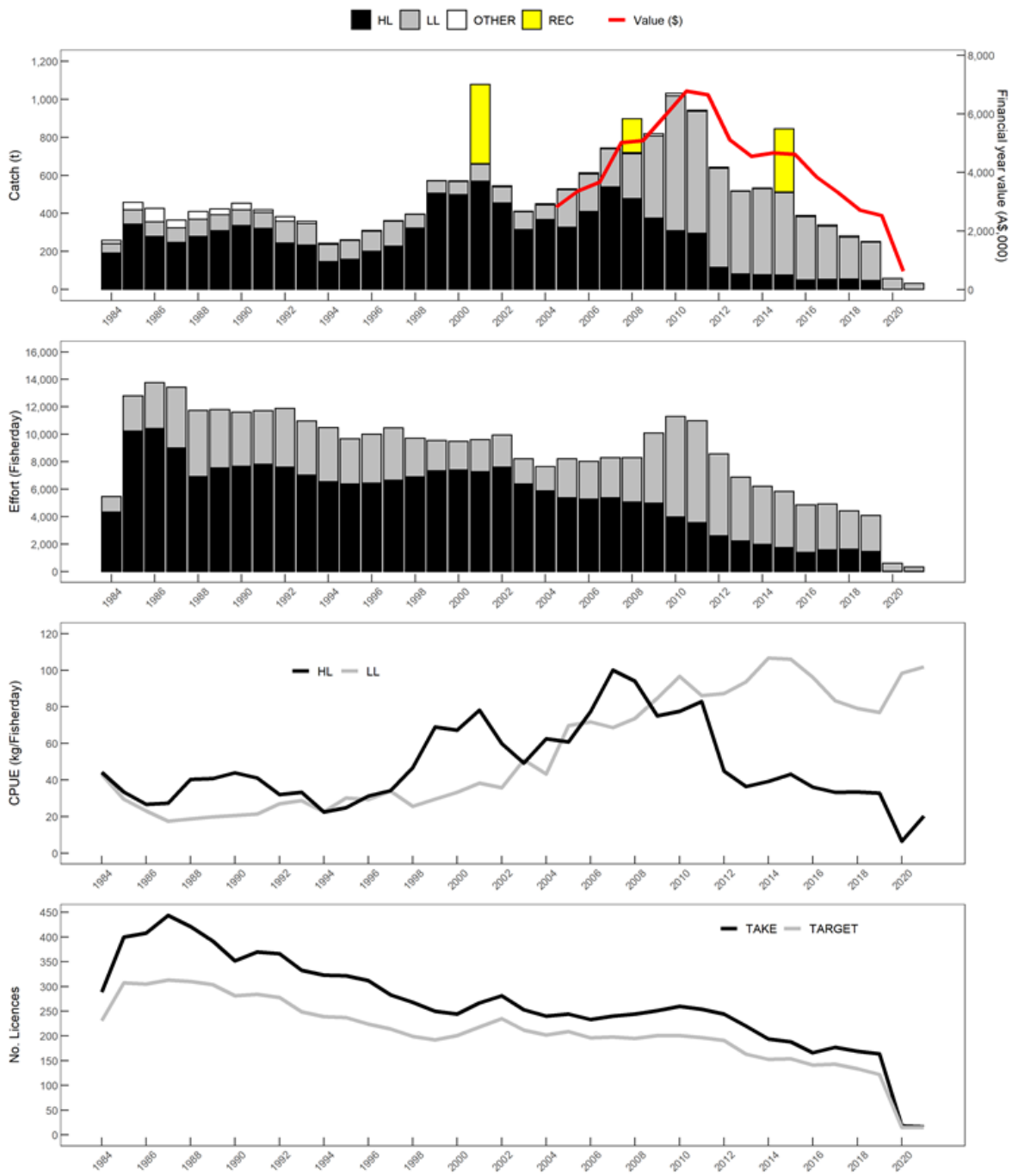

Figure 2-2. Long-term, State-wide trends in: (A) total commercial catch by gear type (handline (HL) and longline (LL)), estimates of recreational catches, and gross production value; (B) total commercial effort for HL and LL; (C) catch per unit effort (CPUE) for HL and LL; and (D) the number of active commercial licence holders taking and targeting Snapper.

2.3.2. Contribution by Stock

The relative contributions of the three stocks to total State-wide catches have changed considerably over time, in response to the change in spatial structure of the fishery between 2008 and 2012, and the fishery closures implemented in late 2019. The SG/WCS provided the highest proportions of annual catches up to 2009, after which they declined to their lowest levels between 2012 and 2019 (Figure 2-3). Catches from the GSVS were generally low until 2004, they then increased gradually until 2007 before increasing further between 2007 and 2010, when catches from this stock became (and 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 declining back to a low level in 2013 where they have remained at a relatively low level up until 2021.

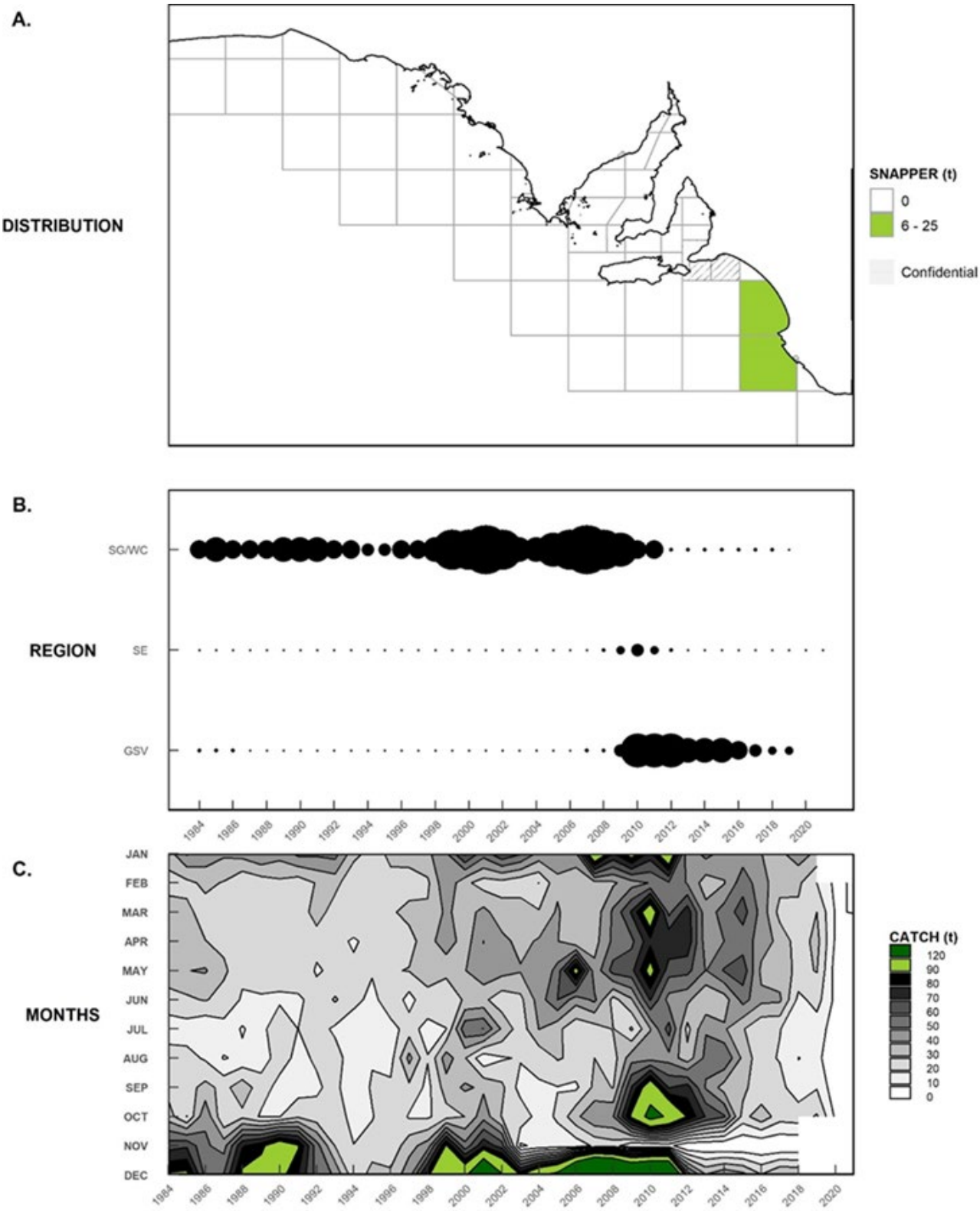

Figure 2-3. (A) Spatial distribution of commercial catch in 2021 by MFA. Long term trends in: (B) the annual distribution of commercial catch among stocks; and (C) months of the year.

2.3.3. Spencer Gulf / West Coast Stock

Prior to the fishery closure, annual catches from the SG/WCS varied cyclically for most of the period from 1984 to 2019, with peaks in 1990, 2001 and 2007, the latter year producing the highest catch of 618.3 t (Figure 2-4A). From 2007 to 2012, annual catches decreased, and subsequently remained relatively stable at a low level. In 2019, the lowest recorded catch of 61.7 t was taken. Catches from the WC component of this stock have ranged from 6.9 t in 1995 to 42.8 t in 2009, with most annual catches < 20 t (Figure 2-4).

Targeted HL catches have varied over time. The highest of 516.4 t was taken in 2001, which has since decreased and in 2019 was the lowest on record at 26.1 t (Figure 2-4B). Targeted HL effort increased between 1984 and 2002 when it was at its highest level of 5,142 fisher-days (Figure 2-4C). Since then, it declined to the lowest level recorded at 459 fisher-days in 2019. Targeted HL CPUE has fluctuated, but showed a long-term increasing trend to 2011, which peaked in 2007 at 138.1 kg.fisher-day^{-1}, before declining to 63.8 kg.fisher-day^{-1} in 2012 and then further to 48.1 kg.fisher-day^{-1} by 2018 (Figure 2-4D). It subsequently increased to 56.8 kg.fisher-day^{-1} in 2019.

The numbers of licence holders who took and targeted Snapper using HL declined slowly through the 1980s and 1990s but the rates of decline increased through the 2000s (Figure 2-4E). Those taking Snapper using HL fell from 219 in 1985 to 92 in 2019, while those targeting Snapper 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 and from 2012 to 2019 were relatively low (i.e., generally <400 catches.yr^{-1}). Estimates of Prop200kgHLTar range from 0.1 to 0.25, but showed no long-term trend.

From 1984 to 2004, targeted LL catch for the SG/WCS was relatively stable before it increased and peaked at 154.2 t in 2006 (Figure 2-4G). Catch declined thereafter and, by 2019 had fallen to 22.9 t, the second lowest annual total. Targeted LL effort peaked at 2,578 fisher-days in 1997, but declined since then (Figure 2-4H). From 2014 to 2018, effort relatively stable before declining to its lowest level of 523 fisher-days in 2019. Highest targeted LL CPUE occurred between 2005 and 2008, peaking at 98.7 kg.fisher-day^{-1} in 2006 (Figure 2-4I). After 2008, it decreased and by 2014 was 33.7 kg.fisher-day^{-1}. It subsequently increased to 52.8 kg.fisher-day^{-1} in 2018, but decreased again to 43.9 kg.fisher-day^{-1} in 2019.

Since 1988, the number of licence holders taking Snapper fell from 118 to 40 while those targeting Snapper fell from 100 to 32 (Figure 2-4J). The numbers of reported daily LL catches fell between 2006 and 2011 and have subsequently remained at the relatively low level of <500 catches.yr^{-1}(Figure 2-4K). The annual estimates of Prop200kgLLTar peaked 0.28 in

2006 and 2007, prior to daily catches being constrained by trip limits. They then declined to approximately 0.1 in 2011 and have since remained around this low level.

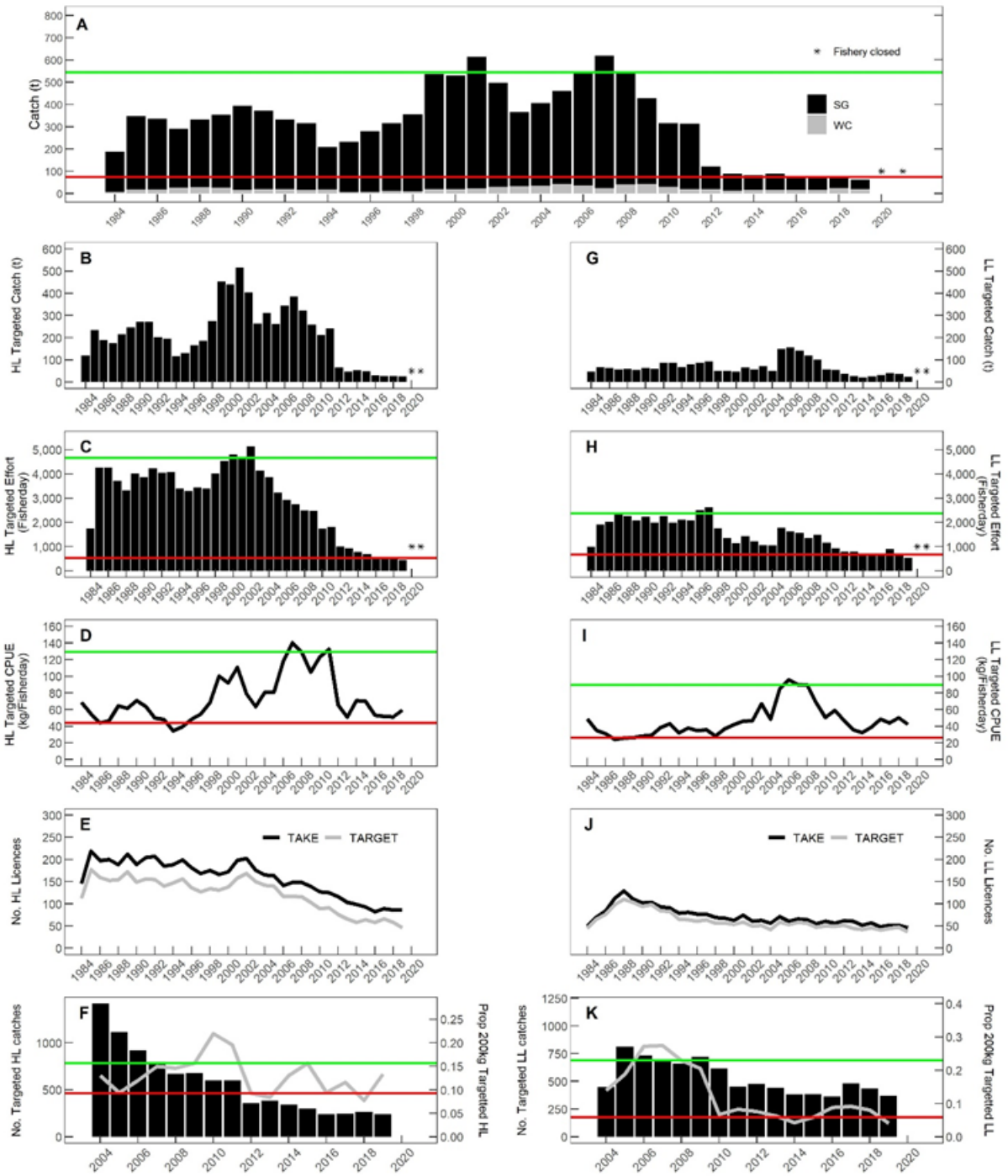

Figure 2-4. Key fishery statistics used to inform the status of the Spencer Gulf / West Coast Stock. Long-term trends in (A) total catch; (B) targeted handline catch; (C) effort; (D) catch rate; and (E) the number of active licence holders taking and targeting the species; (F) number of targeted daily catches and Prop200kgTarHL; (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 6-1.

2.3.4. Gulf St Vincent Stock

Catch and effort statistics for the GSVS are reported and analysed for the period of 1984 to 2019. Between 1984 and 2006, the GSVS produced relatively low catches (Figure 2-5A). 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, which is 37.7% of the highest catch recorded, 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 2-5B, C). 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 (Figure 2-5D). It has subsequently decreased to a moderate level, with 34.6 kg.fisher-day^{-1} recorded in 2019.

The number of licences using handline declined considerably through the 1980s and 1990s. The number that reported taking Snapper declined from 96 in 1984 to 41 in 2019 (Figure 2-5E). Similarly, the number that targeted Snapper reduced from 89 to 28. The number of reported daily handline catches have generally been <300.yr^{-1} since 2004 (Figure 2-5F). 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 rapid increase in total catches from 2008 – 2010 (Figure 2-5G). Between 2008 and 2015, targeted LL catch increased from 46.7 t to 388.2 t (Figure 2-5G). This increase was associated with a 334.1% increase in targeted longline fishing effort from 657 to 2,852 fisher-days. Targeted fishing effort then declined between 2016 and 2019 from 2,558 to 1,487 fisher-days (Figure 2-5H). Between 2000 and 2010, LL CPUE increased considerably, peaking at 145.7 kg.fisher-day^{-1} (Figure 2-5I). From 2015, LL CPUE declined reaching 100.4 kg.fisher-day^{-1} in 2019.

The number of LL licences that took and targeted Snapper peaked in 2012 at 66 and 64, respectively, but since declined to 29 and 28 in 2019 (Figure 2-5J). The number of daily longline catches increased from 2007, peaked in 2012 at 1,448 catches and then declined between 2016 and 2019 to 693 catches (Figure 2-5K). The Prop200kgTarLL was low from 2004 to 2008 (<0.2) but then increased up to 0.57 in 2014. Since then, there has been a general decline to 0.43 in 2019.

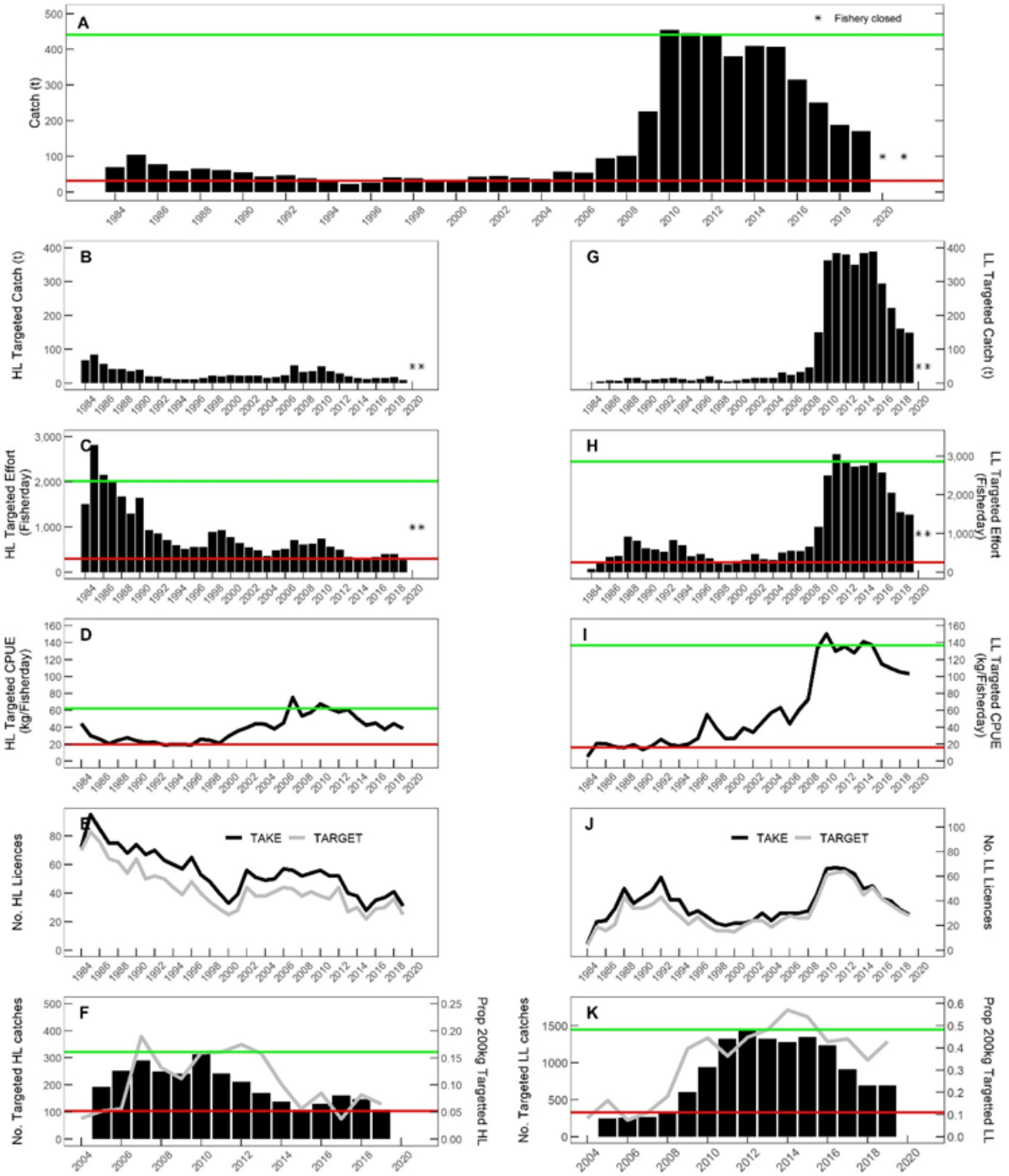

Figure 2-5. Key fishery statistics used to inform the status of the Gulf St Vincent Stock. Long-term trends in (A) total catch; (B) targeted handline catch; (C) effort; (D) catch rate; and (E) the number of active licence holders taking and targeting the species; (F) number of targeted daily catches and Prop200kgTarHL; (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 6-1.

2.3.5. South-East Region

The SE Region has generally produced low catches of Snapper compared to the other stocks. However, from 2006 to 2010 there was an exponential increase in catch that peaked in 2010 at 271.6 t (Figure 2-6A). Catch then declined sharply to 4.8 t in 2016 before moderately increasing to 21.6 t in 2018 and 23.5 t in 2019. Commercial catches then doubled to 57.9 t in 2020, constrained by the TACC of 60.75 t. In 2021, annual total catch was 31.6 t, which was taken over two TACC fishing periods. From 1 July 2021, the MSF quota unit values (and subsequent catch limits) are set by financial year.

In the 2020 fishing season, the total catch across sectors was 59.2 t, which represented 79% of the TAC (Table 2-2). The catch was dominated by the commercial sector (57.9 t), with minor contributions from the Charter Boat (0.35 t) and recreational (0.86 t) sectors. In the 2021 fishing season (i.e., 1 February to 30 June 2021), the total catch was 25.7 t which represented 96% of the TAC. Again, catch was dominated by the commercial sector (22.7 t), with the recreational sector contributing 2.6 t (Table 2-2). The commercial and recreational sectors each caught their respective TACs in March 2021. Catch records for the 2021/22 fishing season are available to 31 December 2021 and are therefore incomplete. Indigenous and traditional fishery catches are not reported and a nominal 1% is allocated to this sector.

Table 2-2. Reported Snapper catch (kg) for the commercial, charter, and recreational sectors in the SE Region during TAC periods from 1 February 2020 and 31 December 2021. *Includes data to 31 December 2021 and not the complete TAC period.

Time period	TAC	Total catch	Commercial	Charter	Recreational
1 Feb 2020 – 31 Oct 2020	75,000	59,150	57,988	350	857
1 Feb 2021 – 30 June 2021	26,667	25,658	22,731	373	2,554
*1 July 2021 – 30 June 2022**	*48,000*	*10,266*	*8,928*	*494*	*844*

Targeted HL catch in the SE Region has always been low, with annual mean catch of 2.7 t (Figure 2-6B). Catch increased between 2006 and 2009, which peaked in 2007 at 14.5 t, but then declined (Figure 2-6B). Such catches reflect low but variable fishing effort, which peaked at 402 fisher-days in 2007 (Figure 2-6C). Until 2003, targeted HL CPUE was generally <20 kg.fisher-day^{-1} (Figure 2-6D). It increased to its highest levels from 2006 to 2009, peaking at 60.6 kg.fisher-day^{-1} in 2008. From then, HL CPUE declined to the lowest level in 2017, before increasing again in 2019.

The numbers of HL fishers that took and targeted Snapper peaked in 2009, at 16 and 14, respectively (Figure 2-6E) before declining to <5 fishers between 2016 and 2021 (except for 2019 (n=9)). Since 2004, the numbers of reported daily catches have been consistently low having declined from a peak of 153 in 2007 to 42 in 2019 (Figure 2-6F). Prop200kgTarHL was

highest from 2008 to 2009 and again in 2019, but in years between has either been close to or zero.

Up to 2007, annual targeted LL catches were generally <5 t. There was then a rapid increase to the highest recorded catch of 251.2 t in 2010 (Figure 2-6G). Longline catches then declined to the lowest catch of 3.2 t in 2016, before increasing to 19.8 t in 2018 and then to 19.5 t in 2019. Targeted LL catch then increased to 54.2 t in 2020 but fell to 30.1 t in 2021, which was constrained by TACC. There was a considerable increase in targeted LL effort from 64 fisher-days in 2003 to its peak at 2,950 fisher-days in 2011 (Figure 2-6H). Effort subsequently declined to only 102 fisher-days in 2016 but increased to 296 fisher-days in 2021. Targeted LL CPUE increased between 2007 and 2010, reaching 87.9 kg.fisher-day^{-1} (Figure 2-6I). It was variable from 2010 to 2016, then increased from 2017 and reached a peak of 101.7 kg.fisher-day^{-1} in 2021. Using an alternative metric, targeted LL CPUE peaked in 2007 at 0.39 of kg/hook as catch and effort increased. This metric then fluctuated, reflecting raw LL CPUE trends (kg.fisher-day^{-1}) until 2011 when it started to decrease, reaching 0.06 kg/hook in 2015. Since then, it has slowly increased to 0.17 kg/hook in 2021. The numbers of LL fishers who took and targeted Snapper increased from 2005 and peaked in 2010 at 47 and 35, respectively, before declining to 16 and 15 in 2021 (Figure 2-6J). The number of daily catches increased from 2007, peaked at 965 in 2010, and then declined to a low of 50 in 2016 (Figure 2-6K). Daily catches then increased to 256 in 2020, reducing back to 158 in 2021. Prop200kgTarLL also peaked in 2010 at 0.47 and then declined to 0.02 in 2016. From 2017, it successively increased reaching its highest level of 0.57 in 2020, then declined to 0.42 in 2021.

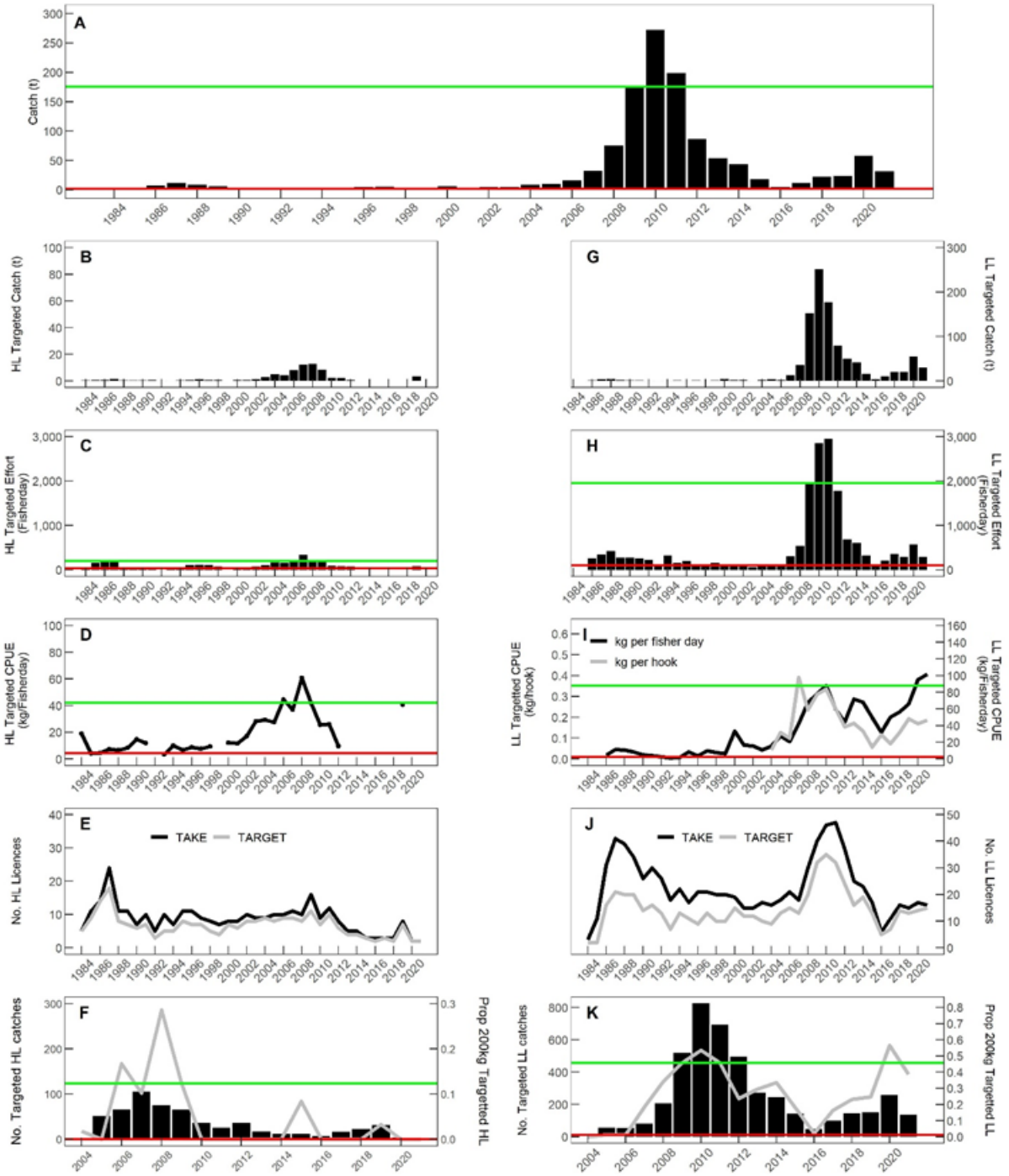

Figure 2-6. Key fishery statistics for the population of Snapper in the South-East Region. Long-term trends in: (A) total catch; (B) targeted handline catch; (C) effort; (D) catch rate; (E) the number of active licence holders taking and targeting the species; (F) number of targeted daily catches and Prop200kgTarHL; (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 6-1.

2.4. Discussion

Since the last Snapper stock assessment in 2020 (Fowler et al. 2020), the SG/WCS and GSVS have been closed to Snapper fishing due to deterioration in stock status since 2012. The fishery closures have resulted in a substantial decline in State-wide catch and effort, with Snapper fishing only permitted in the SE Region in 2020 and 2021.

In 2020, there was a considerable increase in catch and effort in the SE Region since 2019. This coincided with the fishery closures for the SG/WCS and GSVS and can be partly attributed to the displacement of effort. The sharp increase in catch and effort in 2020 resulted in the highest targeted LL CPUE recorded, which was then surpassed in 2021 (Figure 2-6). The notable increase in targeted LL CPUE may be related to an increase in relative abundance of Snapper as a result of the recent recruitment of several strong year classes to the population from the WVS, and the transition of several specialised Snapper fishers into the SE Region.

Longline catch and effort continued to be the dominant gear type used to target Snapper in the SE, while HL accounted for a marginal proportion of catch and effort. The limited HL effort over the past few years necessitated a shift to using an alternate metric of LL CPUE (kg per hook) as an index of relative abundance in the SnapEst model (Chapter 5). This new CPUE time series represents the first step in refining the CPUE metrics that are integrated into fishery assessment models.

The implementation of TACs for all sectors for the SE Region since November 2019 has constrained catches to ensure sustainable levels of exploitation. However, it has created additional challenges in interpreting trends in fishery statistics due to differences in TACs and the duration of fishing seasons (Table 2-1). The TAC periods and sectoral allocations changed following the removal of a temporal spawning closure and the shift from State-wide to regional allocations through the MSF reform (Smart et al. 2022). These changes make inferring trends over an annual scale challenging. These challenges will be resolved for future assessments following a shift to reporting by financial year to align with quota periods. Switching to financial-year reporting may alter some historic trends in catch and effort but will ensure that future changes in fishery dynamics can be clearly interpreted.

A poor understanding of catch and effort from the recreational sector has been a limitation in previous stock assessments for Snapper, whereby recreational catches were estimated from phone and diary surveys undertaken in 2000/01 (Henry and Lyle 2003), 2007/08 (Jones 2009), and 2013/14 (Giri and Hall 2015). The recent management approach of setting a Total Allowable Recreational Catch (TARC) and compulsory catch reporting in the SE Region has provided the most comprehensive understanding of recreational Snapper fishing to date.

3. REGIONAL LENGTH AND AGE STRUCTURES

3.1. Introduction

Appropriate fishery management requires a fundamental understanding of the demographic processes of a fish population, to determine its response to levels of exploitation. Determining the age, along with length, of the fish is essential in determining any rate-based population metric, such as growth, maturity, recruitment, and natural mortality. Tracking a population's demography through time can provide an indication of its response to ecological processes or to varying levels of exploitation, providing valuable information for natural resource management. Age determination of fish is generally done by interpreting growth rings in the structure of their hard anatomical structures, such as their otoliths (Campana 2001).

This chapter describes the annual length and age structures for the six regional populations of Snapper in SA (i.e., WC, NSG, SSG, NGSV, SGSV, and SE) and provides estimates of length-at-age for the three stocks (i.e., SG/WCS, GSVS, and SE Region). This demographic information is a key input into the SnapEst assessment model (Chapter 5), which simulates natural population dynamics and demographic processes.

3.2. Methods

3.2.1. Sample collection

Since 2000, a weekly market sampling program has been undertaken by SARDI researchers to provide biological data for Snapper caught by commercial fishers across SA. Commercial catches are primarily accessed through the SAFCOL fish market in Adelaide and augmented with periodic trips to regional areas and research cruises (e.g., DEPM surveys). This market sampling program has conformed to a two-stage protocol (Fowler et al. 2016a), where each Snapper within a commercial catch is measured (caudal fork length (CFL), and a sub-sample of the catch are processed to collect biological information, including whole weight, sex, gonad weight, reproductive stage, and age (from otolith analysis). The determination of fish age followed a standard protocol that involved the preparation of a transverse section of a single otolith (Fowler et al. 2013). The fisher's details were recorded so that the area of capture and the capture method could be determined from the mandatory daily catch return.

The closure of the SG/WCS and GSVS to fishing since 1 November 2019 precluded the collection of biological information from fishery catches for these stocks. To overcome this, a targeted adult sampling program was implemented to collect representative biological samples and continue the annual time series of regional length and age structures. This involved contracting commercial MSF fishers to collect samples of Snapper from representative regions, i.e., the West Coast of Eyre Peninsula (WC), northern Spencer Gulf (NSG), southern Spencer Gulf (SSG), northern Gulf St Vincent (NGSV), and southern Gulf St Vincent (SGSV). A total of 118 sampling events, across the five regions from January 2020 to January 2022, were undertaken (Table 3-1) (Figure 3-1). The fish sampled were processed at local fish processing facilities, and the fillets donated and distributed through charitable organisations (Food Bank). Biological samples from the SE Region were obtained from the SAFCOL fish market when the fishery was open, and through an onboard research observer program in this region (part of FRDC 2019-044) (Table 3-1).

Table 3-1. Number of Snapper sampled by region between December 2019 and January 2022.

Year	WC	NSG	SSG	NGSV	SGSV	SE	Total
2019		368					368
2020	202	24	274	245	297	568	1610
2021	228	640	186	195	227	422	1898
2022				263	216	98	577

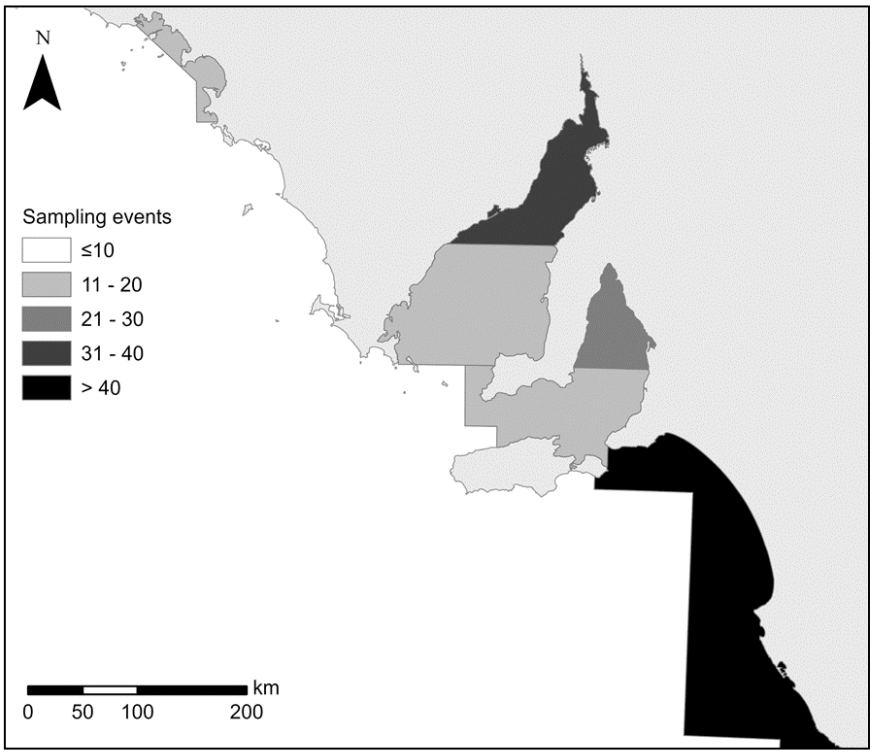

Figure 3-1. Spatial distribution by region showing the number of representative biological samples of Snapper collected between January 2020 and January 2022.

3.2.2. Data processing

The analytical procedures used to develop the annual length and age structures were the same as those used in previous assessments (Fowler et al. 2013, 2016; McGlennon et al. 2000). Length and age structures were developed for the six regions of WC, NSG, SSG, NGSV, SGSV, and the SE. Each size structure was based on all fish measured from handline and longline catches. Age structures were developed by subtracting the estimated age of each fish from its capture date, to determine its year of birth (i.e., year-class) using a nominal birthday of 1 January. Annual weight structures were calculated to show the distribution of biomass harvested in each year across the different size classes. Where size, weight or age structures are not presented for a particular year and region, there were insufficient data available for their development. Four size categories were recognised when considering the annual size structures: (i) 'small' fish that were 30 – 39.9 cm CFL; (ii) 'medium' fish that were 40 – 59.9 cm CFL; (iii) 'large' fish that were 60 – 79.9 cm CFL; (iv) and 'very large' fish that were ≥80 cm CFL. For each of the three stocks, length-at-age data collected between 2000 and 2022 were used to generate von Bertalanffy growth curves. The R software package (R Core Team 2022) was used to fit the non-linear equation to the data and generate estimates of L_∞, K, and t_0.

3.3. Results

3.3.1. Northern Spencer Gulf

For Snapper in NSG, the sample sizes of fish collected during market sampling and the resulting size structures changed markedly through the period of 2010 to 2021 (Figure 3-2). Length and age structures developed prior to 2010 are presented in previous stock assessments (Fowler et al. 2010, 2013). Initially, the sample sizes were large due to the high catches from this region, but then declined, particularly after 2012, reflecting the lower catches and fewer fish from this region that passed through the SAFCOL fish market until the fishery closed in 2019.

From 2010 to 2015, the size distributions were generally broad, although dominated by fish in the ‘small’ and ‘large’ size categories (Figure 3-2). The weight distributions were dominated by the ‘large’ or ‘very large’ size categories. The size distribution changed in 2016, and the size and weight distributions were dominated by the ‘small’ size category. This continued through 2017 to 2019, although with higher representation from the ‘medium’ size category in both the size and weight distributions. In 2021, the size distribution was uniform across the ‘small’, ‘medium’ and ‘large size categories, whilst the latter size class dominated the weight distribution.

The age structures provide insight into the demographic processes that were responsible for the changes in the size and weight distributions. From 2010 to 2015, the age structures were relatively broad and characterised by numerous year classes (Figure 3-3). Nevertheless, through this period the age structures were dominated by the 1997 and 1999-year classes. By 2017, these two strong year classes were depleted, and the age structures were more characterised by fish that had recruited between 2012 and 2016. Between 2018 and 2021, it was evident that the 2014-year class showed moderate strength compared to recent years, whilst in the latter two age structures the 2016-year class emerged as above-average.

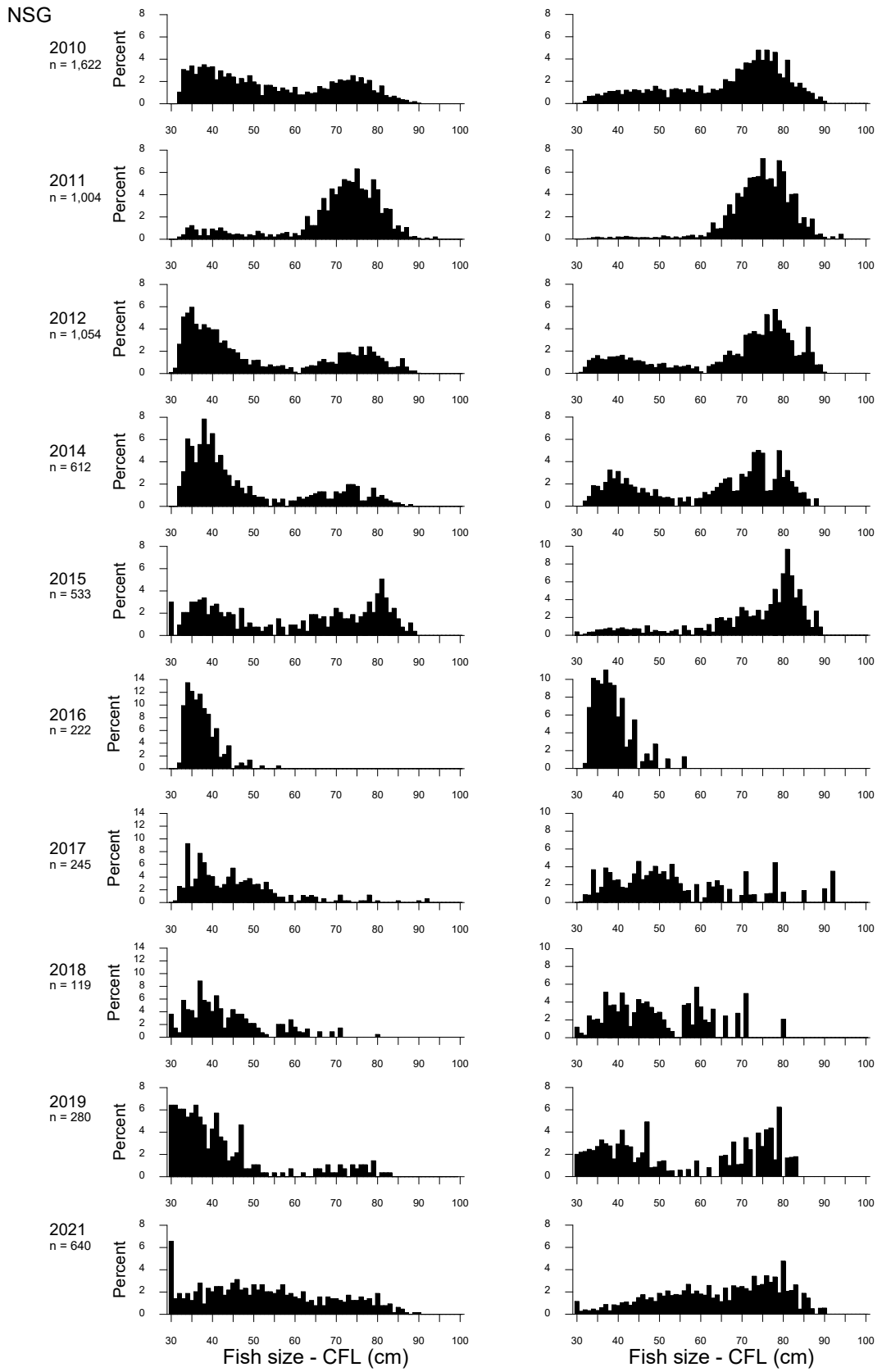

Figure 3-2. Estimated annual size and biomass distributions for Snapper caught in northern Spencer Gulf (NSG) from 2010 to 2021. Left hand graphs show the size structures. Right hand graphs show the percentage of biomass accounted for by each size class.

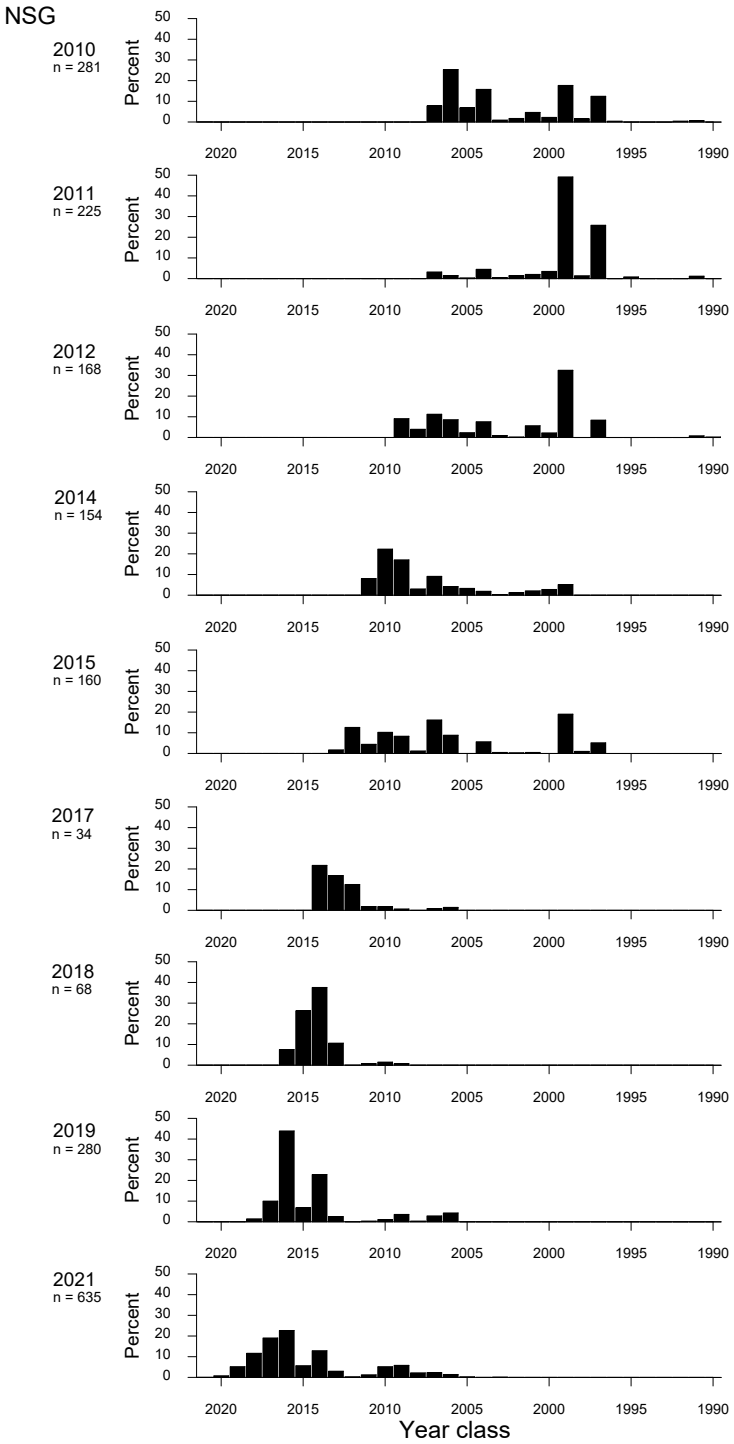

Figure 3-3. Estimated annual age structures for fish caught in northern Spencer Gulf (NSG) between 2010 and 2021. For each year, data are presented as the relative percentage of total catch accounted for by each year class, i.e., the years in which they were spawned.

3.3.2. Southern Spencer Gulf

Population size structures for SSG are available for most years from 2010 to 2021 (Figure 3-4). Length and age structures developed prior to 2010 are presented in previous stock assessments (Fowler et al. 2010, 2013). Sample sizes were highly variable amongst years, but generally declined over time. They were particularly low from 2015 to 2019, prior to a recent increase that related to the targeting of adult Snapper by SARDI in 2020 and 2021 as part of its fishery-independent sampling program. The annual size structures reflected broad size ranges of fish but generally involved modes of ‘small’ and ‘large’ fish, the relative sizes of which varied between years (Figure 3-4). The weight distributions were generally dominated by ‘large’ fish. However, in 2021 ‘large’ fish were notably absent, and so the size and weight distributions were dominated by ‘small’ and ‘medium’ fish.

There were sufficient fish sampled from this region to develop age structures for some years between 2010 and 2021. The age distributions in 2010, 2011 and 2014 were dominated by the 1997 and 1999-year classes, i.e., relatively old fish (Figure 3-5). By 2015, both of these year classes had become depleted and the subsequent age distributions were dominated by fish that had recruited during the 2000s. The age structure in 2020 was broad and characterised by strong year classes that had recruited in 2006, 2007, 2009 and 2014. The age structure in 2021 further supported the significance of the 2014-year class, and also revealed the emergence of a potentially significant 2016-year class for this region.

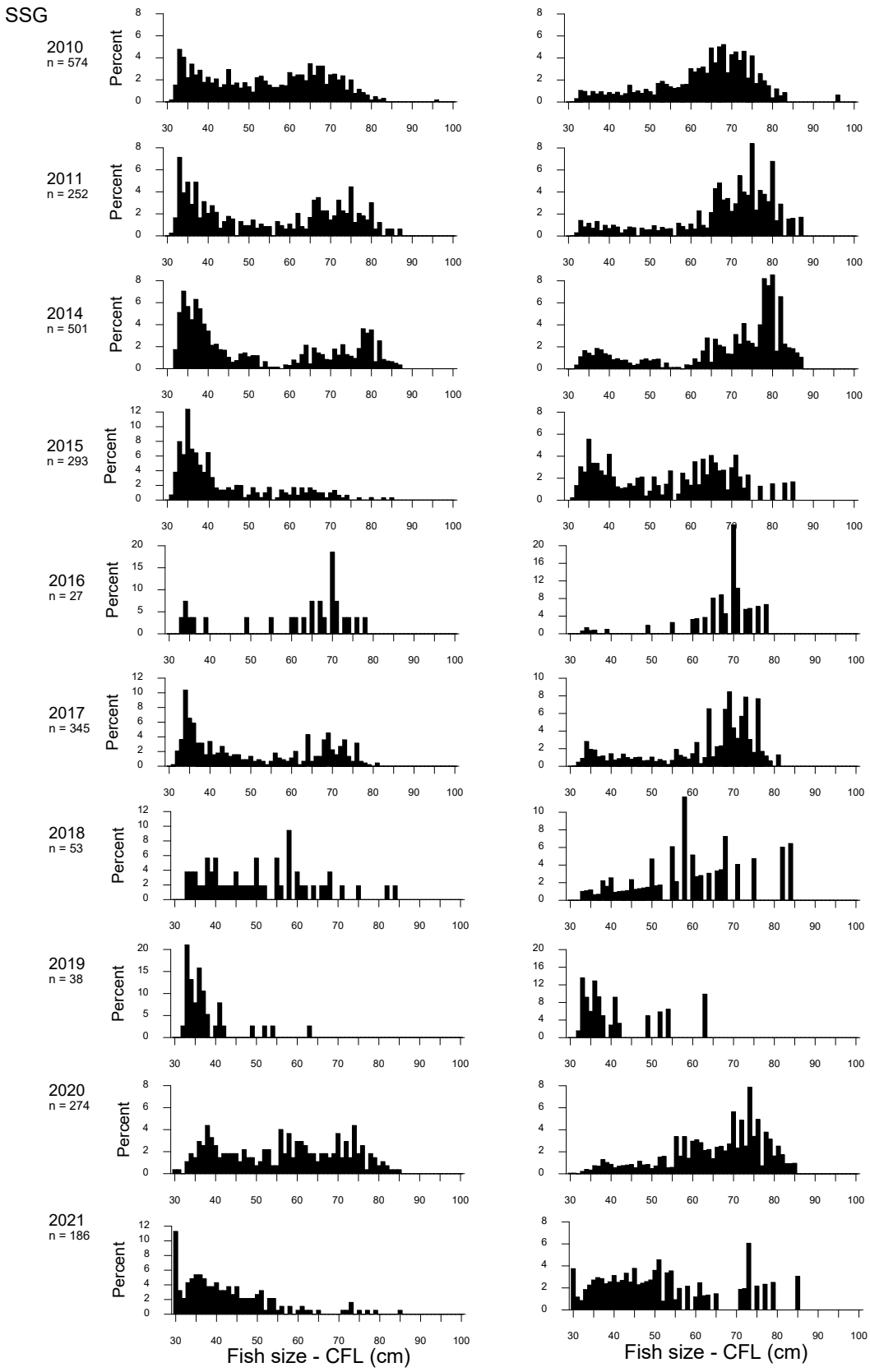

Figure 3-4. Estimated annual size and biomass distributions for Snapper caught in southern Spencer Gulf (SSG) from 2010 to 2021. Left hand graphs show the size structures. Right hand graphs show the percentage of biomass accounted for by each size class.

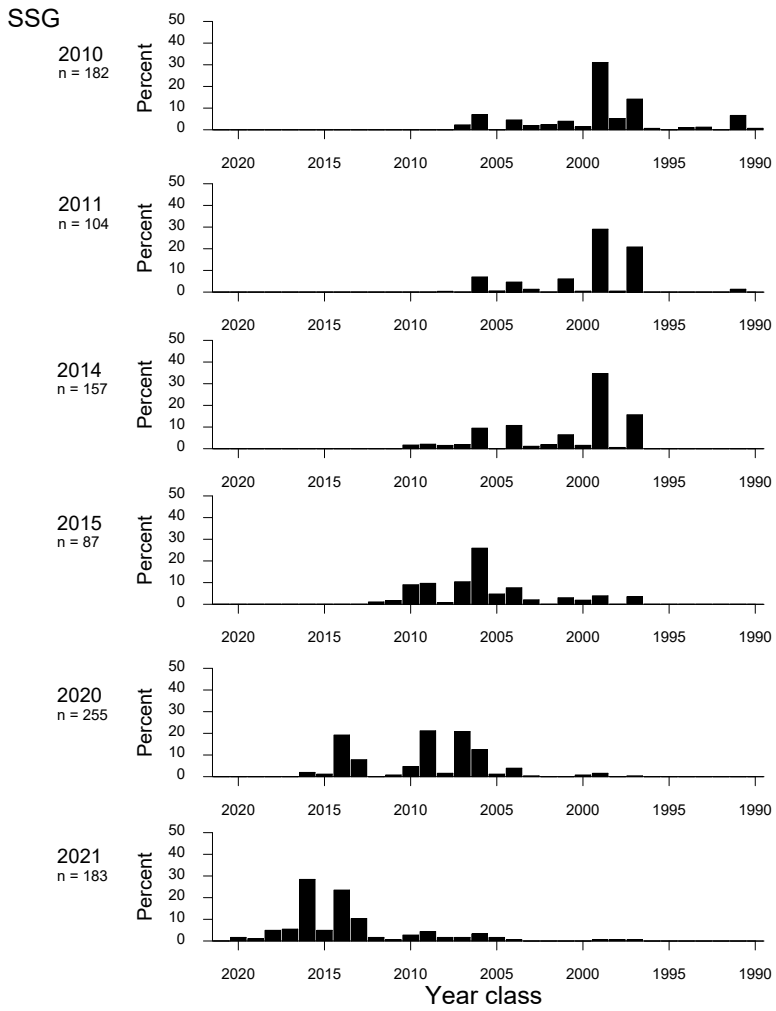

Figure 3-5. Estimated annual age structures for fish caught in southern Spencer Gulf (SSG) between 2010 and 2021. For each year, data are presented as the relative percentage of total catch accounted for by each year class, i.e., the years in which they were spawned.

3.3.3. Northern Gulf St Vincent

Since 2008, this region has provided high catches of Snapper. Up to 2017, relatively high numbers of fish were sampled during market sampling, but from 2018 onwards, sample sizes were much lower. In most years, all four size categories were well represented in the size structures, with some variation existing among years in their relative contributions (Figure 3-6). In 2010, the 'large' and 'very large' fish were most numerous. In 2011 and 2012, there was no modal structure evident in the size structures, indicating that all size categories contributed to the catches. Then, from 2016 to 2022, the annual size structures were dominated by 'large' fish, with relatively few 'small' and 'medium' fish. Furthermore, there were also few 'very large' fish compared to previous years. This suggests some recent contraction in the size structures, which was particularly evident in the annual weight distributions. From 2010 to 2015, the weight distributions were unimodal and involved 'large' and 'very large' fish, and then from 2016 onwards, they were dominated by the 'large' fish.

For NGSV, sufficient otolith samples were collected in most years from 2010 to 2022 to develop population age structures (Figure 3-7). Age structures developed prior to 2010 are presented in previous stock assessments (Fowler et al. 2010, 2013). Recent age structures 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 structure in 2017 first showed the emergence of the 2014-year class, that year class has remained evident as a moderate year class through to 2022. Furthermore, the age structures in 2020, 2021 and 2022 indicated the potential significance of the 2017-year class as another above average year.

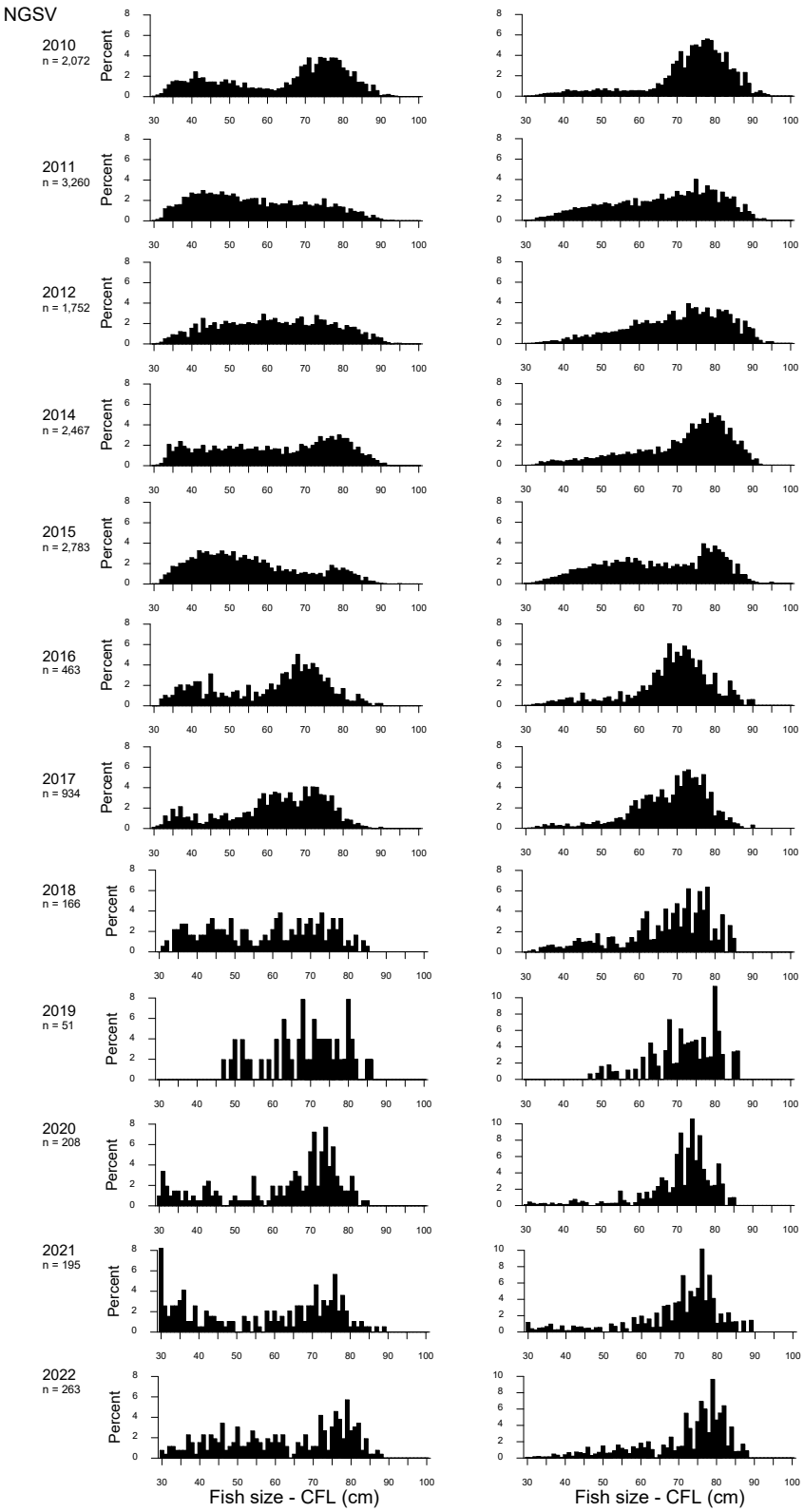

Figure 3-6. Estimated size and biomass distributions for Snapper caught in northern Gulf St Vincent (NGSV) from 2010 to 2022. Left hand graphs show the size structures. Right hand graphs show the percentage of biomass accounted for by each size class.

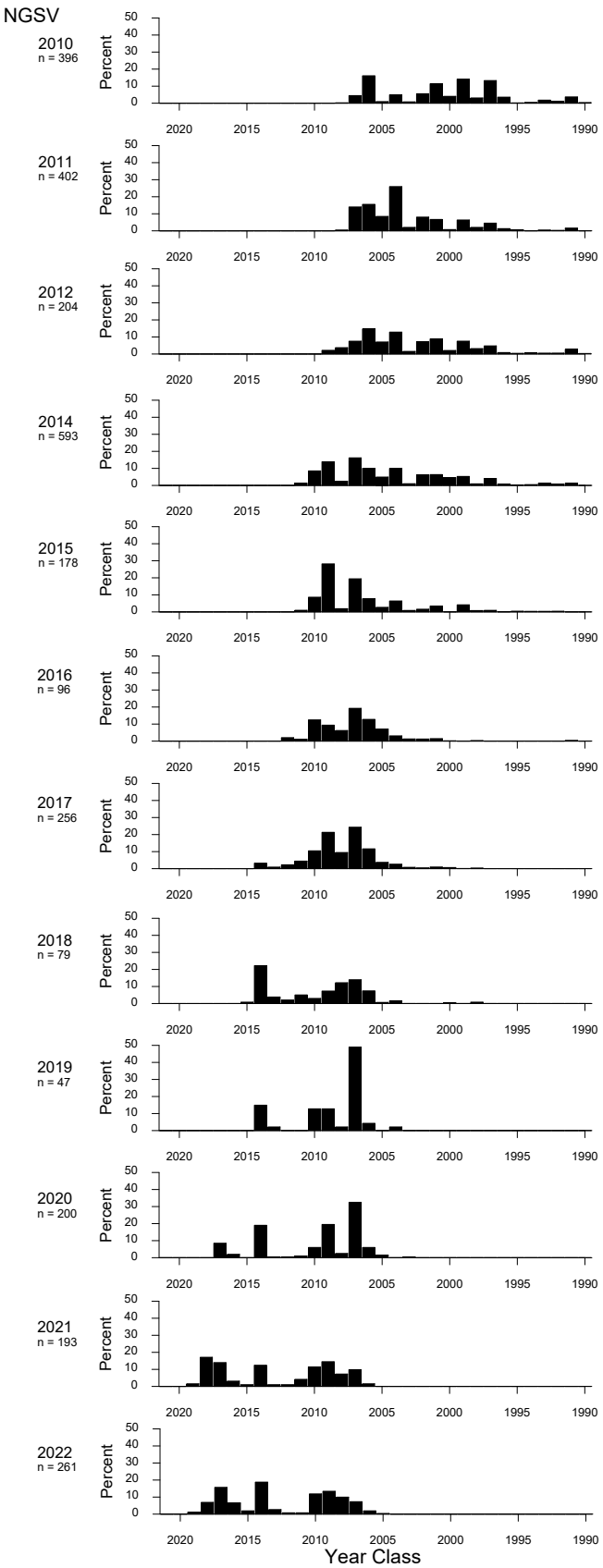

Figure 3-7. Estimated annual age structures for fish caught in northern Gulf St Vincent (NGSV) between 2010 and 2022. For each year, data are presented as the relative percentage of total catch accounted for by each year class, i.e., the years in which they were spawned.

3.3.4. Southern Gulf St Vincent

Sample sizes of fish measured from SGSV were relatively high until 2016 and declined thereafter. The consistent sample sizes in 2020, 2021 and 2022 reflected SARDI's targeted adult sampling program. The size structures from 2010 to 2015 were generally multi-modal but were dominated by 'small' and 'medium' sized fish (Figure 3-8). 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 suggest relatively higher numbers of 'large' fish. In contrast, in 2020, 2021 and 2022, the size distributions were broad, but were nevertheless dominated by the 'medium' fish. Generally, the weight distributions reflect the relatively high contributions of both the 'medium' and 'large' fish.

Age structures developed prior to 2010 are presented in previous stock assessments (Fowler et al. 2010, 2013). The age structures for SGSV from 2010 onwards involved very few fish that had recruited throughout the 1990s (Figure 3-9). Those from 2010 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, but also reflected the emergence of the relatively strong 2014-year class. This was reinforced by the age structures for 2019 and 2020 that were dominated by the 2007, 2009 and 2014-year classes. The age structures for 2021 and 2022 were dominated by the 2014-year class, and also showed the emergence of the 2017-year class as another possible above average year class.

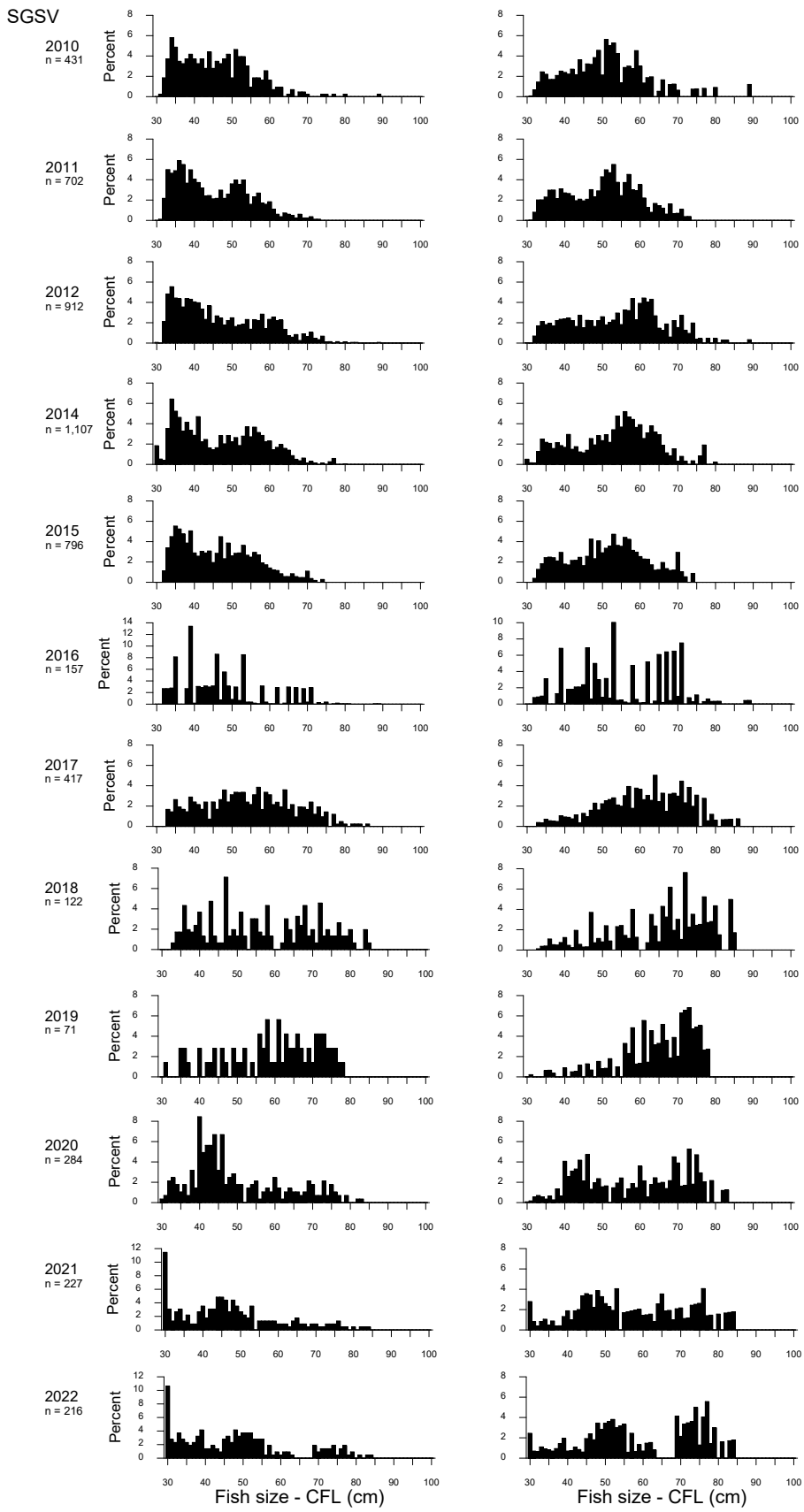

Figure 3-8. Estimated annual size and biomass distributions for Snapper caught in southern Gulf St Vincent (SGSV) from 2010 to 2022. Left hand graphs show the size structures. Right hand graphs show the percentage of biomass accounted for by each size class.

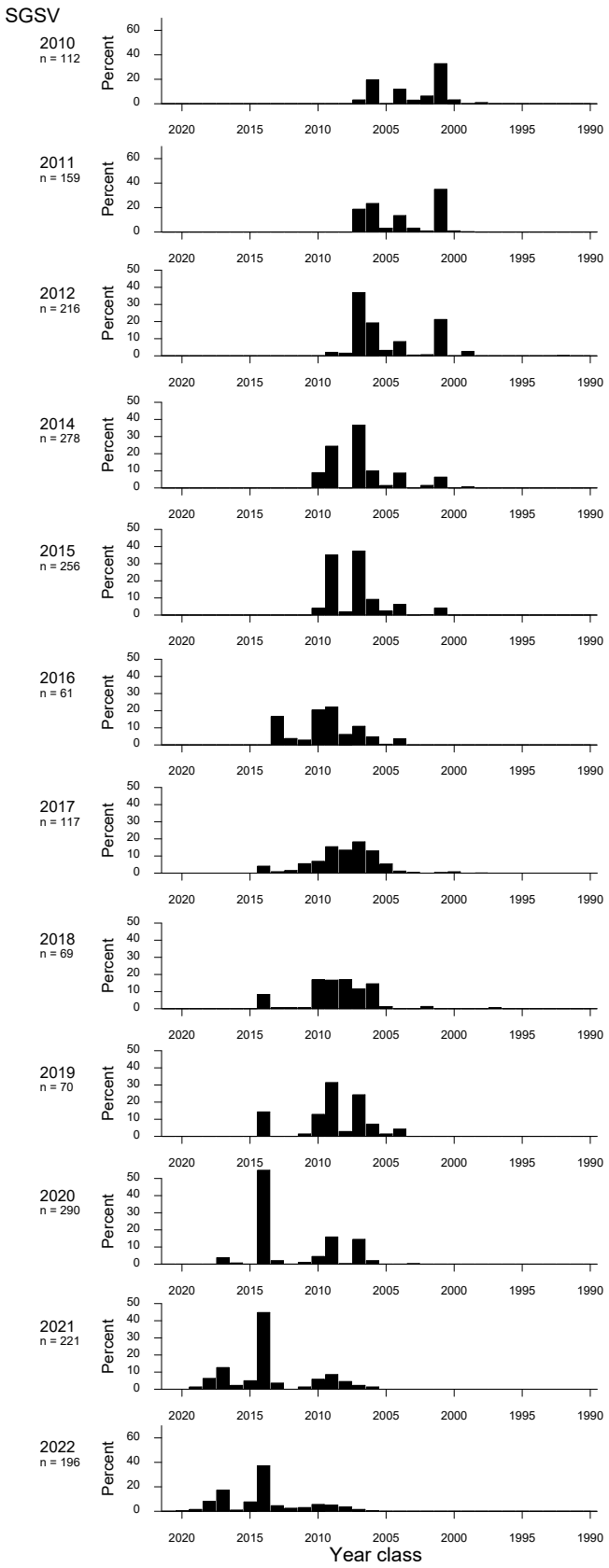

Figure 3-9. Estimated annual age structures for fish caught in SGSV between 2010 and 2022. For each year, data are presented as the relative percentage of total catch accounted for by each year class, i.e., the years in which they were spawned.

3.3.5. South-East

For the SE Region, size distributions are available for a number of years from 2010 to 2022 (Figure 3-10). Up to 2014, the sample sizes were relatively high, but in 2017 and 2019 they were very low. Sample sizes increased considerably in 2020 and 2021, due to a SARDI observer program on commercial fishing vessels. All size structures for this region rarely involved fish >60 cm CFL and so were dominated by 'small' and 'medium' fish (Figure 3-10). For several years until 2017, there was a proportional increase in the representation of 'large' fish. The weight distributions from 2010 to 2017 were dominated by 'medium' fish but also contained some 'large' fish. The size distributions for 2020 and 2021 had large modes of 'small' fish as well as some 'medium' fish. The weight distributions for these years were bimodal, characterised by fish in both size categories.

Up to 2014, the age structures were dominated by the 2001 and 2004-year classes, whose relative contributions changed throughout this period (Figure 3-11). In 2015, the age structure 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. The age structures for 2021 and 2022 were dominated by the 2013 and 2014-year classes.

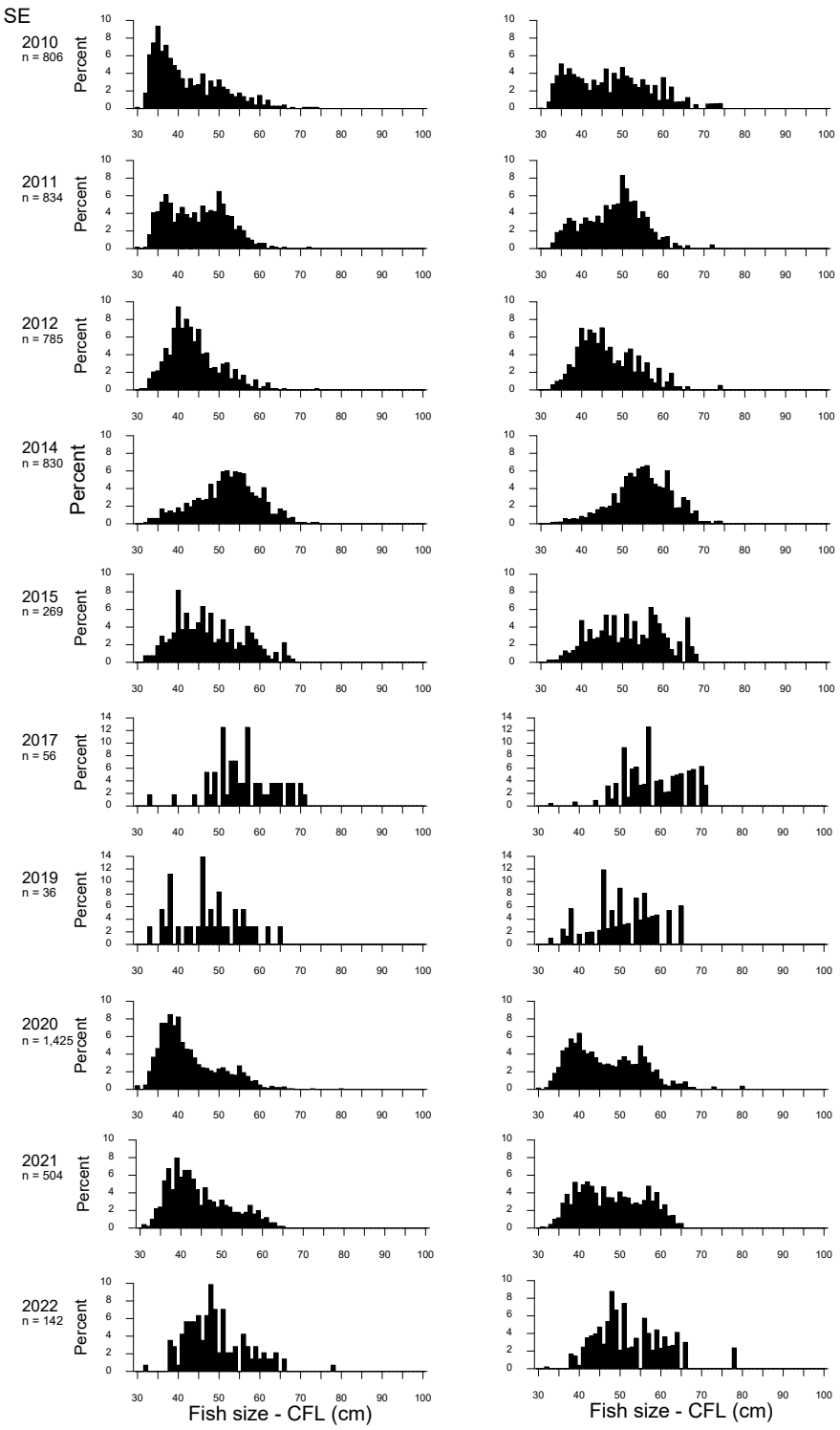

Figure 3-10. Estimated annual size and biomass distributions for Snapper caught in the South-East (SE) from 2010 to 2022. Left hand graphs show the size structures. Right hand graphs show the percentage of biomass accounted for by each size class.

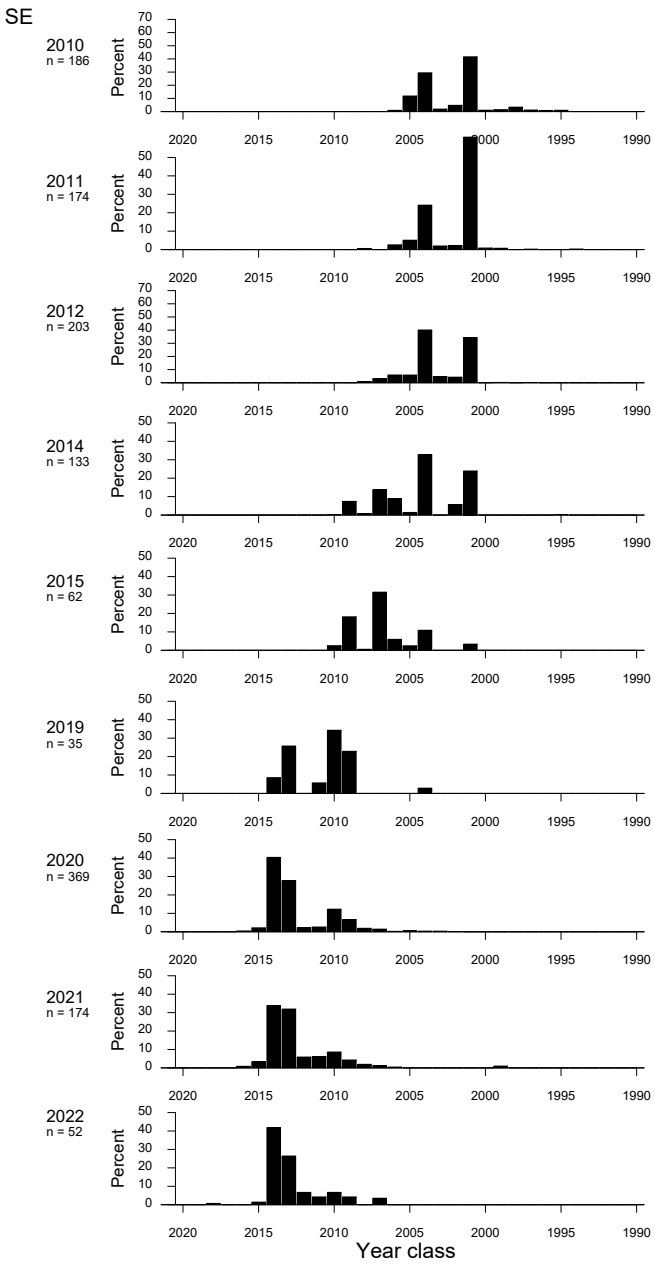

Figure 3-11. Estimated annual age structures for fish caught in the South-East (SE) between 2010 and 2022. For each year, data are presented as the relative percentage of total catch accounted for by each year class, i.e., the years in which they were spawned.

3.3.6. West Coast

Historical estimates of length and age structures from Snapper from the WC are not available due to difficulties consistently accessing fish at Adelaide's SAFCOL fish market. However, targeted adult sampling by SARDI in 2019, 2020, and 2021 enabled the development of contemporary length and age structures that can be used as baselines for future assessments. In these years, the size and weight distributions were dominated by 'large' fish with few fish in the other size categories evident (Figure 3-12). The fish covered a broad age range, including one captured in 2020 that was 27-years old. The age structures for each of the three years were dominated by several strong year classes (Figure 3-13). In 2019, these were the 2006, 2008, 2012 and 2014-year classes. Those in 2020, were the 2009, 2012 and 2014-year classes. In 2021, the 2014-year class was the clear dominant age class.

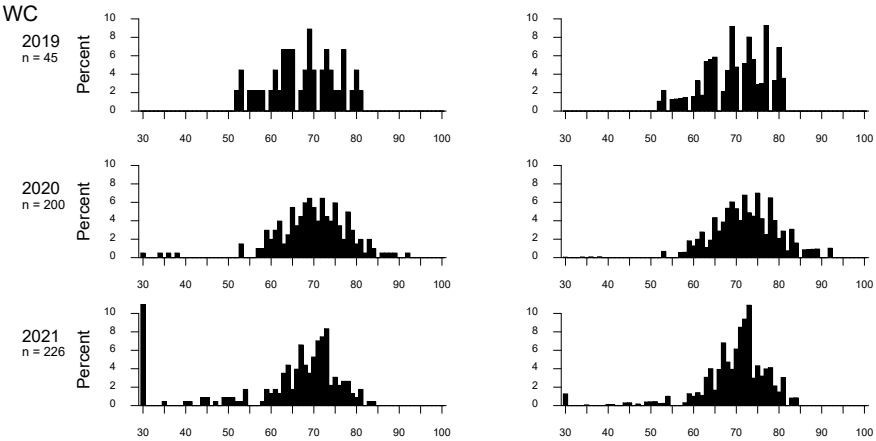

Figure 3-12. Estimated annual size and biomass distributions for Snapper caught on the West Coast of Eyre Peninsula (WC) from 2019 to 2021. Left hand graphs show the size structures. Right hand graphs show the percentage of biomass accounted for by each size class.

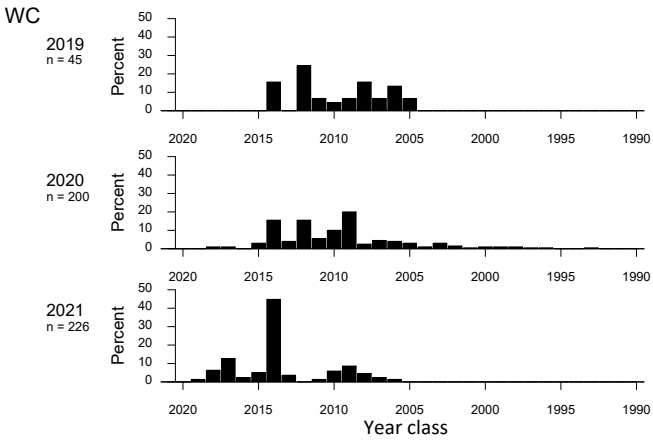

Figure 3-13. Estimated annual age structures for fish caught on the West Coast of Eyre Peninsula (WC) between 2019 and 2021. For each year, data are presented as the relative percentage of total catch accounted for by each year class, i.e., the years in which they were spawned.

3.3.7. Growth: Length-at-age

Previous analysis of growth for Snapper in SA (McGarvey and Feenstra 2004, Fowler et al. 2013) showed high regional variation in mean Snapper length-at-age, with SSG having fish with smaller mean lengths at age than NSG. Compared to the northern gulfs, Snapper are also slower growing in the SE Region. In this assessment, Snapper growth estimates were updated using the revised model regional breakdown aligned with the three stocks, SG/WCS, GSVS, and SE Region.

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 3-14). Results for mean growth were more uncertain for the aggregated SG/WCS. Large and small sized Snapper were evident in the scatterplot of individual samples for the SG/WCS (Figure 3-14), 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 for this stock likely reflects the presence of two distinct groups with different growth characteristics. It is likely that this bimodal separation reflects the slower growth of Snapper from SSG compared with those NSG (Fowler 2016).

Overall variation in Snapper growth among individual fish in all regions is high (Figure 3-14). In the two gulfs, a 10-year-old Snapper can vary in total length from 35 to 90 cm CFL. 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 numerous years to reach the minimum legal length, the faster growers reaching 38 cm around age 3, and slower growers recruiting around age 12 in GSV, with others exceeding 12 years to recruit into the fishery in the slow-growing region of SSG. One important advantage of the slice-partition modelling method employed in SnapEst to assess the three Snapper 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 than fitting to the age proportions, mean lengths, catch totals, and catch rates, all of which strongly depend 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 3-14 and their confidence intervals, are provided in Appendix 9.1.

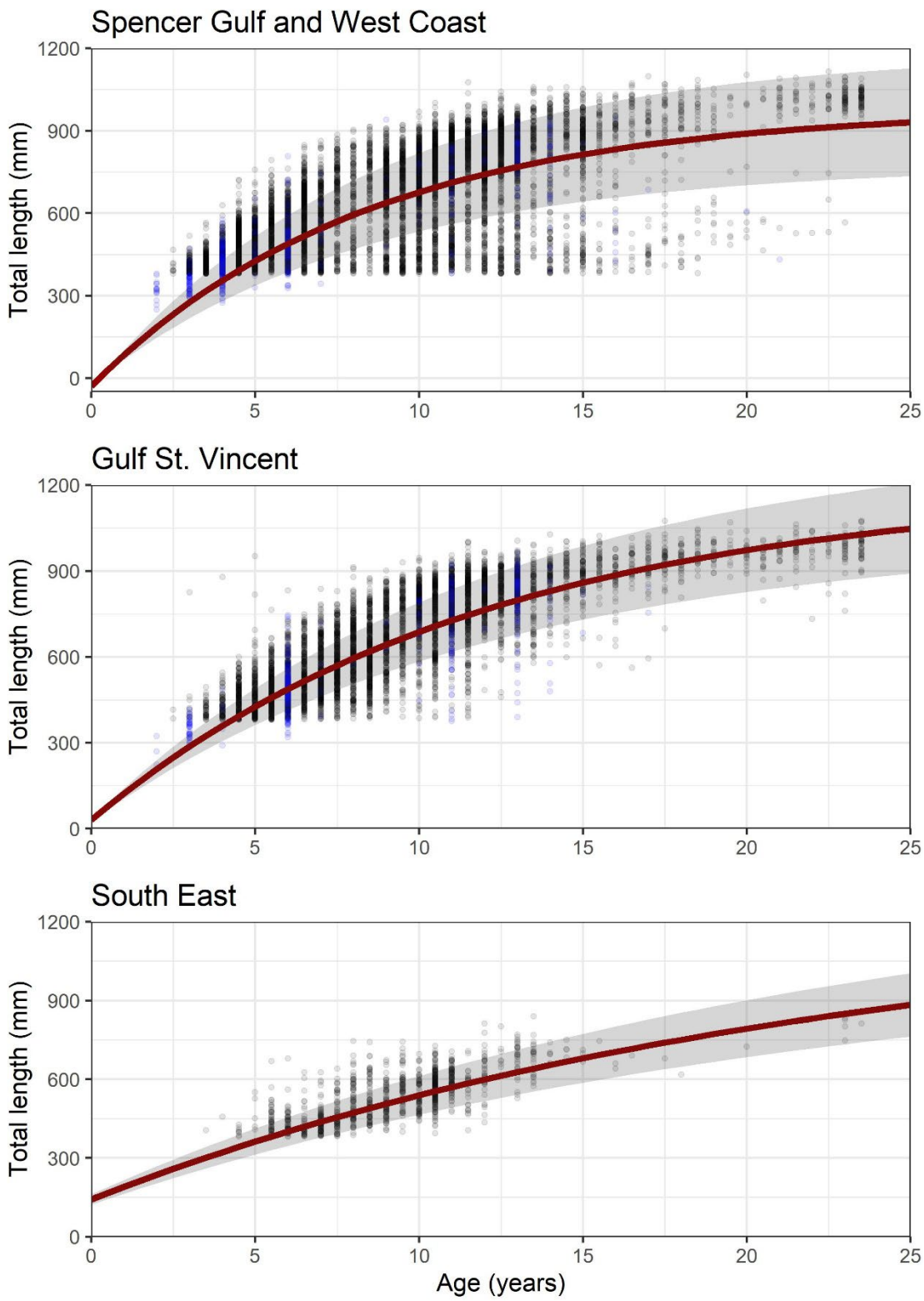

Figure 3-14. Fitted von Bertalanffy growth (VBG) curves for Snapper length-at-age for the three SA stocks. Red line: fitted mean length at age. Shaded bands: 95% confidence intervals. Grey points: commercial, primarily SAFCOL market, samples. Blue points: age-length samples taken in the two gulfs by SARDI researchers that were not subject to the cut-off at the legal minimum length of 380 mm TL. Data points are translucent to show density of samples.

3.4. Discussion

Length-at-age data are fundamental to understanding the regional population dynamics of Snapper in SA and are an integral input into the SnapEst model (Chapter 5). This demographic information also provides an important empirical data source that is needed to determine stock status.

The closure of the SG/WCS and GSVS necessitated a targeted adult sampling program to be implemented to obtain regional biological information on age, growth and reproduction. The adult sampling program addressed two key objectives: (i) to develop annual length and age structures for each region that could be used in SnapEst; and (ii) to collect reproductive information for Snapper in SG and GSV that was used to estimate spawning biomass using the daily egg production method (DEPM) (Chapter 4). The targeted sampling also provided highly resolved spatial information on catches over a broad geographic range.

Consistent with previous assessments, the length and age structures in 2020 and 2021 showed variation among regions (Fowler et al. 2016, 2019, 2020). The general trend in the regional comparisons of population dynamics is that the northern region of each gulf displays faster growth rates and achieves larger maximum lengths than those in the southern regions (Fowler et al. 2013). Furthermore, the differences in regional age structures reflected the influence of highly variable inter-annual recruitment of age-0 juveniles to regional nursery areas, which is a phenomenon that is well documented for populations of Snapper in South Australia (Fowler and Jennings 2003, Saunders 2009, Fowler et al. 2010, 2017a). The occasional strong year class has previously sustained the fishery through periods characterised by poor to average recruitment (McGlennon et al. 2000, Fowler et al. 2017a).

For SG, length and age structures in 2021 continued to show evidence of truncation and were characterised by a high proportion of small, young fish. Although a higher proportion of older fish were sampled in 2020 and 2021 compared to previous years, the majority of fish sampled were ≤8 years of age. Older fish (>15 years) have been mostly absent in the age structures for several years despite the potential for Snapper to reach >30 years of age (McGlennon et al. 2000). The age structures for each of NSG and SSG throughout the 2000s and 2010s indicated the lack of strong year classes since the late 1990s, and, consequently, a prolonged period of poor recruitment. The 2014 and 2016-year class was well represented in the age structures for NSG in 2019 and 2021, and for SSG in 2021, which indicated that it could be a better than average year class. Nevertheless, the relative strength of the 2016-year class needs to be interpreted with caution because this cohort is yet to be subjected to the prolonged fishing mortality experienced by other year classes.

Comparatively, the length and age structures for GSV were much broader and contained a higher proportion of large, old fish. For NGSV, the strong 2007- and 2009-year classes remained present in the recent age structures and have persisted through the unprecedented levels of catch and effort sustained by the GSVS since the late-2000s (Fowler et al. 2013, 2016a, 2019). The 2014-year class was well represented in recent age structures, particularly for SGSV where it was the dominant year class in 2020, 2021, and 2022. Similarly, the 2017-year class is emerging as a better than average year class.

The population structures for the SE Region in 2020 and 2021 were characterised by small to medium sized fish (35–60 cm CFL), which primarily related to the growth of fish from the 2013- and 2014-year classes. There was also a small proportion of larger fish that persisted from the 2009- and 2010-year classes. Each of these strong year classes corresponded to years of high 0+ recruitment in Port Phillip Bay (PPB), Victoria, which supports the hypothesised density dependent movement of Snapper from PPB into the SE Region (Fowler 2016, Fowler et al. 2017a). Recruitment of 0+ juveniles in PPB was the highest on record in 2018, and therefore it is expected that the 2018-year class will begin to enter the fishable biomass in the SE Region from late-2022.

The development of length and age structures for the WC was facilitated by the targeted adult sampling program in this region. The size structures in 2020 and 2021 were characterised by a high proportion of large fish (60–80 cm CFL), which related to a broad range of year classes between 2005 and 2014. The relative strength and persistence of the 2014-year class in the annual age structures suggested it was an above-average recruitment year. Length and age structures differed between the WC and SG, although a very small proportion of Snapper on the WC persist from the strong 1997 and 1999-year classes in SG. Given the lack of strong recruitment to SG since 1999 and the concurrent depletion of the population (Fowler et al. 2016a, 2019, 2020), it is unlikely that the WC population has been replenished through the emigration of juveniles from SG in recent years.

4. REGIONAL ESTIMATES OF SPAWNING BIOMASS

4.1. Introduction

The spawning biomass of Snapper in Spencer Gulf (SG) and Gulf St Vincent (GSV) was estimated using the daily egg production method (DEPM) in December 2021 and January 2022. The estimates of spawning biomass from the DEPM are one of several fundamental inputs into the SnapEst model (Chapter 5).

The DEPM was originally developed for stock assessment of Northern Anchovy (*Engraulis mordax*) (Parker 1980, Lasker 1985) and has been applied to more than 20 species of small pelagic fishes worldwide (e.g., Stratoudakis et al. 2006, Neira et al. 2009, Ward et al. 2021). The DEPM has been used to estimate the spawning biomass of Snapper in New Zealand (Zeldis and Francis 1998), South Australia (McGlennon 2003) and Western Australia (Jackson et al. 2012). However, difficulties in confidently differentiating eggs of Snapper from other teleost species based on their morphology previously caused uncertainty in the estimates of spawning biomass and precluded the ongoing application of DEPM for Snapper. This issue was recently addressed through the development of a molecular technique to validate the identity of Snapper eggs (Oxley et al. 2017, Steer et al. 2017).

The underlying principle of the DEPM is that spawning biomass can be determined from the relative density of pelagic eggs per unit area (i.e., total daily egg production) divided by the mean number of eggs produced per unit mass of adult fish (i.e., mean daily fecundity) (Parker 1980, Lasker 1985). Total daily egg production is the product of mean daily egg production (P_0) and total spawning area (A). Mean daily fecundity is calculated by dividing the product of mean sex ratio (by weight, R), mean spawning fraction (S), and mean batch fecundity (F) by mean female weight (W). Spawning biomass (SB) is calculated according to the equation:

$$SB = \frac{P_0 \cdot A}{\left(\frac{R \cdot S \cdot F}{W}\right)} \qquad \text{(Equation 4.1)}$$

The DEPM surveys in 2021 were the fourth undertaken for each gulf since 2013, and the third since 2018. The initial surveys in 2013 (NSG) and 2014 (GSV and Investigator Strait) were completed as part of the project that adapted the DEPM for Snapper in South Australia (FRDC 2014-019; Steer et al. 2017). The surveys in 2018, 2019, and 2021 were undertaken to calculate fishery-independent estimates of spawning biomass for each region, and those results contributed to the assessments of stock status (Fowler et al. 2019, 2020).

4.2. Methods

4.2.1. Total daily egg production

Ichthyoplankton surveys to sample Snapper eggs were done in SG from 10 to 18 December 2021 and in GSV and Investigator Strait (IS) from 8 to 15 January 2022 (hereafter the 2021 surveys). Since 2013, DEPM surveys have been done in December and January to align with the peak spawning period for Snapper (McGlennon 2003, Saunders 2009, Fowler unpublished data) (Table 4-1). All ichthyoplankton surveys were undertaken from *MRV Ngerin*.

The spatial coverage of the surveys has varied between years to ensure that each survey covered as much of the spawning area as possible, and in response to severe weather. Sampling stations were confined to waters >10 m deep and conformed to a grid of low (4 × 4 nm), medium (2 × 4 nm), or high (2 × 2 nm) intensity. For SG, the surveys in 2013 and 2018 covered northern and central parts of the gulf (Figure 4-1). In 2019, an additional 81 stations were sampled, which expanded the survey into the southern part of the gulf (i.e., northwards of a line from Port Victoria to Tumby Bay). The same 272 stations were sampled in 2021.

For GSV and IS, the survey in 2014 involved 216 stations that covered most gulf waters and extended to the north coast of Kangaroo Island (Figure 4-1). In 2018, the survey was reduced to 138 stations as a result of 48 hours of severe weather and subsequent time constraints. In 2019, 49 stations were added to the original survey area at the western end of IS and through Backstairs Passage. A total of 270 stations were sampled in 2021, which included the 265 stations sampled in 2019 and an additional five stations along the western side of GSV.

The total area surveyed was determined using the Voronoi natural neighbour (VNN) method (Watson 1981), where the survey area was divided into a series of contiguous polygons that were centred around each sampling station, with the boundary of each polygon the equidistant midpoint between surrounding stations. This was done using the geographic information software ArcGIS. The area (km^2) represented by each polygon was then determined, and the total survey area was calculated as the sum of the area represented by all stations sampled.

Table 4-1. Summary of ichthyoplankton surveys for Snapper eggs between 2013 and 2021.

Gulf	Year	Dates	No. stations	Survey area (km^2)	Reference
Spencer Gulf	2013	10-16 December 2013	188	4,610	Steer et al. (2017)
	2018	10-16 December 2018	191	4,792	Fowler et al. (2019)
	2019	4-15 December 2019	272	9,235	Fowler et al. (2020)
	2021	10-18 December 2021	272	9,235	This report
Gulf St Vincent	2014	11-16 December 2014	216	8,022	Steer et al. (2017)
	2018	17-19 December 2018	138	5,059	Fowler et al. (2019)
	2019	13-20 January 2020	265	10,245	Fowler et al. (2020)
	2021	8-15 January 2022	270	10,381	This report

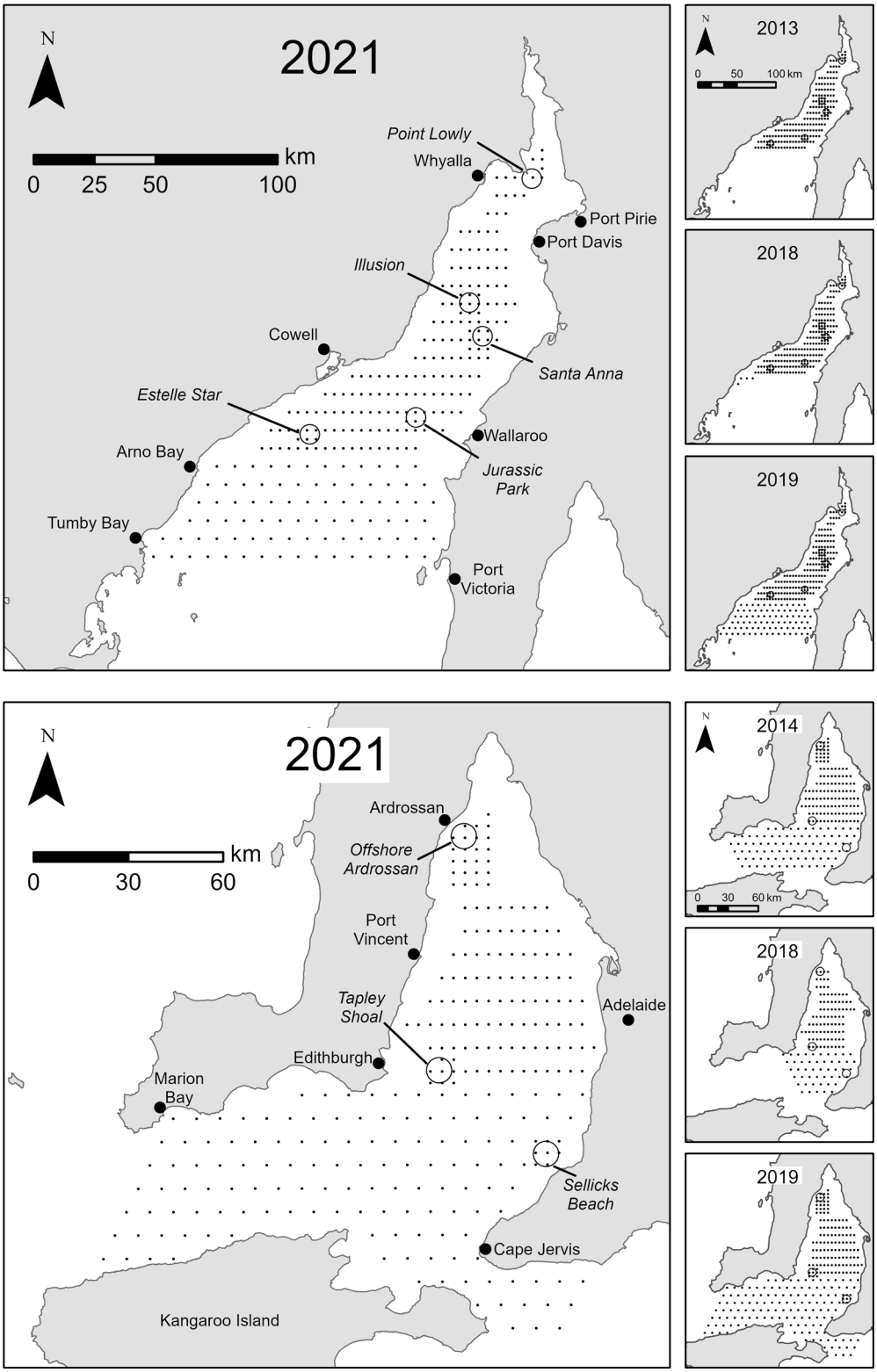

Figure 4-1. Location of stations (●) sampled during ichthyoplankton surveys for Snapper eggs in Spencer Gulf (top) and Gulf St Vincent and Investigator Strait (bottom) from 2013 to 2021. Circles identify previous spatial spawning closures.

Plankton sampling

A plankton sample was collected at each station using paired bongo nets. Each net was 2.5 m long, had an internal mouth diameter of 0.57 m, and was constructed of 500 µm mesh with plastic cod-ends. A large aluminium paravane was attached to the bottom of the net frame to control its trajectory. An oblique tow was done at each station which sampled throughout the water column. The net was deployed over the stern and descended at an angle of ~45° as the vessel proceeded at ~3 knots. Once the net was within 5 m of the seabed, the vessel returned to neutral, and the net was retrieved at ~45° at a constant speed of ~1 $m.s^{-1}$. On retrieval, the nets were rinsed with seawater and the plankton samples in each cod-end were combined into a one litre container. Plankton samples were preserved in 95% ethanol and refrigerated at 4°C prior to processing. A datalogger (Sensus Ultra, ReefNet™, Canada) attached to the net frame recorded the trajectory and maximum depth of the tow. The distance travelled and volume of water sampled differed between tows due to the depth, prevailing wind, tide, and sea conditions. A flowmeter (General Oceanics™ 2030; Florida, USA) was positioned in the centre of each net mouth to estimate the distance travelled for each tow. The mean value of the two flowmeters was used to estimate the distance travelled using factory-calibrated coefficients. Where there was a difference of >500 units between flowmeter readings, the relationship between wire length and flowmeter units was used to determine the most accurate value, and that value was used for both nets.

A Sea-Bird conductivity-temperature-depth (CTD) profiler (SBE 19plus V2 SeaCAT, Sea-Bird Scientific; Washington, USA) was used to record oceanographic parameters (i.e., temperature, salinity, fluorescence) at a subset of stations in each gulf (n = 43 in SG, n = 47 in GSV and IS). The CTD was deployed and retrieved vertically using a hydrographic winch to within 5 m of the seabed. Oceanographic parameters were recorded at 1 m intervals. Water temperatures were interpolated through the survey area using ArcGIS (Appendix 9.2).

Egg identification and staging

Reference samples of Snapper eggs were collected from broodstock maintained at the South Australian Aquatic Sciences Centre (SAASC), West Beach. The broodstock were collected from various sites in SG and GSV between December 2019 and April 2020. Spawning was induced via the implantation of a slow-release cholesterol/cellulose LHRHa pellet (see Ham and Hutchison 2003) and the resulting eggs were incubated at ambient temperature (~18 to 22°C) in a flow through system. Egg samples were collected periodically throughout the embryonic development period (~36 hours) and were preserved in 95% ethanol and refrigerated at 4°C. The samples covered the nine-stage (I-IX) development series described by Steer et al. (2017) and were used to aid the identification of Snapper eggs from mixed plankton samples. Diagnostic characteristics of Snapper eggs include: (i) spherical shape

ranging from 0.78 to 0.90 mm diameter; (ii) smooth chorion; (iii) narrow perivitelline space (0.01 to 0.15 mm); (iv) prominent, unsegmented yolk; (v) single oil globule (0.15 to 0.30 m diameter); and (vi) development of melanophores on the embryo and oil globule.

The protocol used to identify Snapper eggs from mixed plankton samples involved a multi-stage process that applied both morphological and molecular techniques (Figure 4-2). All egg samples were retained in fresh 95% ethanol and refrigerated at 4°C throughout the sorting process. First, each mixed plankton sample was sorted using a modified Sedgewick-Rafter sorting tray under a stereo dissecting microscope. All teleost eggs were removed from each mixed sample and were separated into two categories based on their diameter: 'possible' Snapper eggs which ranged from 0.75 to 0.90 mm diameter; and 'other' eggs that were <0.75 mm or >0.90 mm. The 'other' egg samples were sorted twice to ensure that all 'possible' Snapper eggs were identified. Next, the 'possible' eggs were refined to remove those that were obviously not Snapper. Each egg was categorised based on its morphological characteristics, and the 'possible' eggs were separated into two categories: 'potential' eggs were 0.78 to 0.90 mm diameter and had morphological characteristics similar to Snapper; and 'unlikely' eggs had morphological features obviously different to Snapper (e.g., oil globule absent, wide perivitelline space, excessive pigmentation). The 'potential' egg samples were then examined to identify and quantify Snapper eggs.

Snapper eggs were identified from the 'potential' egg samples and assigned to a development stage (I-IX), based on the morphological descriptions by Steer et al. (2017). Each sample was examined by two scientists and the number of eggs at each stage in each sample were counted. Next, Snapper eggs were identified from the 'potential' egg samples using the *in situ* hybridisation (ISH) molecular technique, which uses an oligonucleotide probe that binds specifically to the mitochondrial 16S ribosomal RNA of Snapper and generates a blue colour change through oxidation with a horseradish peroxidase reactive substrate (Oxley et al. 2017) (Figure 4-3). The chorion of each 'potential' egg was mechanically pierced with a 25-gauge hypodermic needle to expose the embryonic tissue to the molecular probe. Following the ISH process, each sample was examined by two scientists and the number of Snapper eggs at each stage in each sample were counted. The counts of Snapper eggs from the ISH process were used to estimate daily egg production. The counts of Snapper eggs based on morphological characteristics provided a contingency if there was uncertainty with the ISH molecular validation technique.

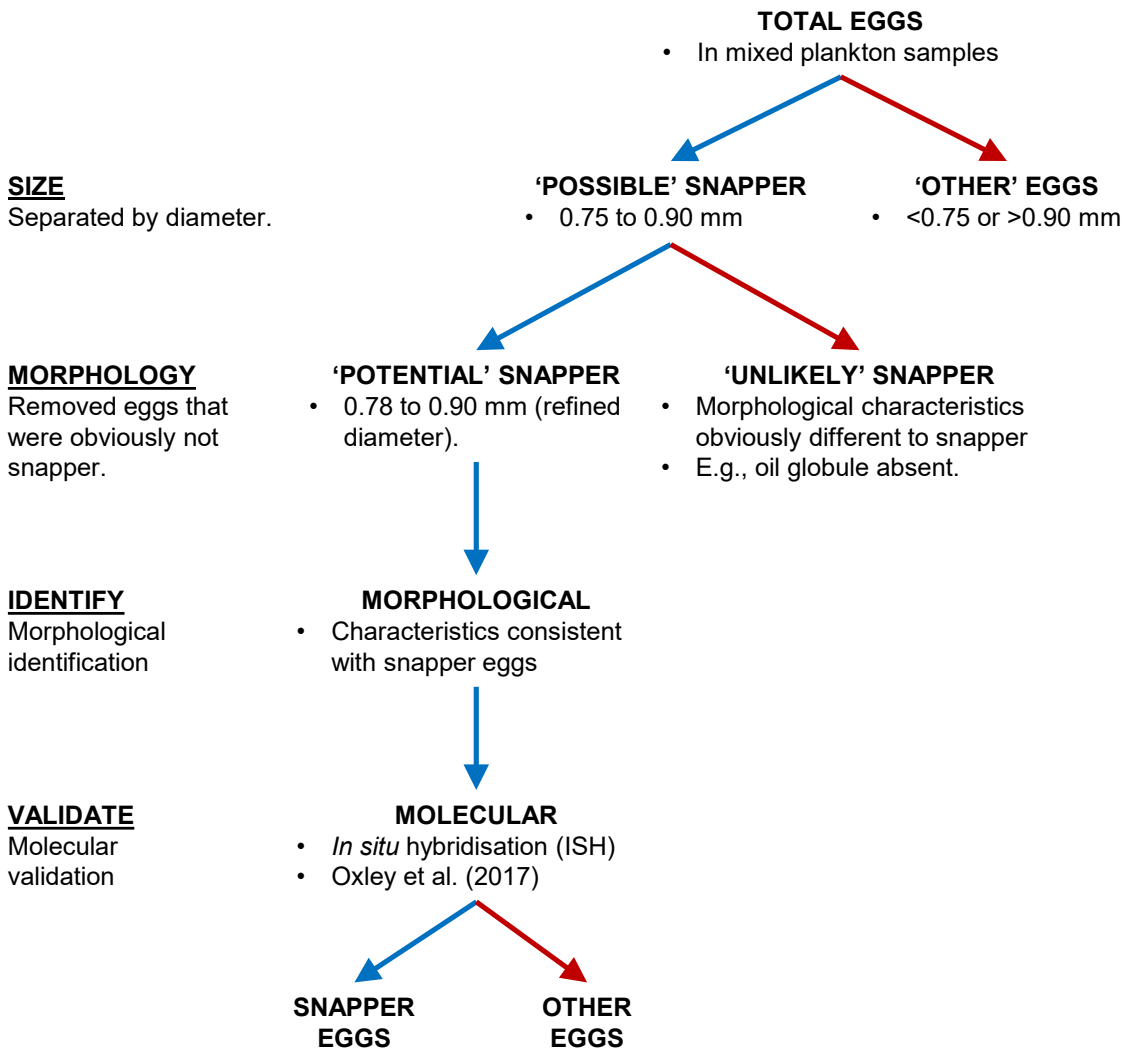

Figure 4-2. Flowchart of the stepwise process used to identify Snapper eggs from mixed plankton samples.

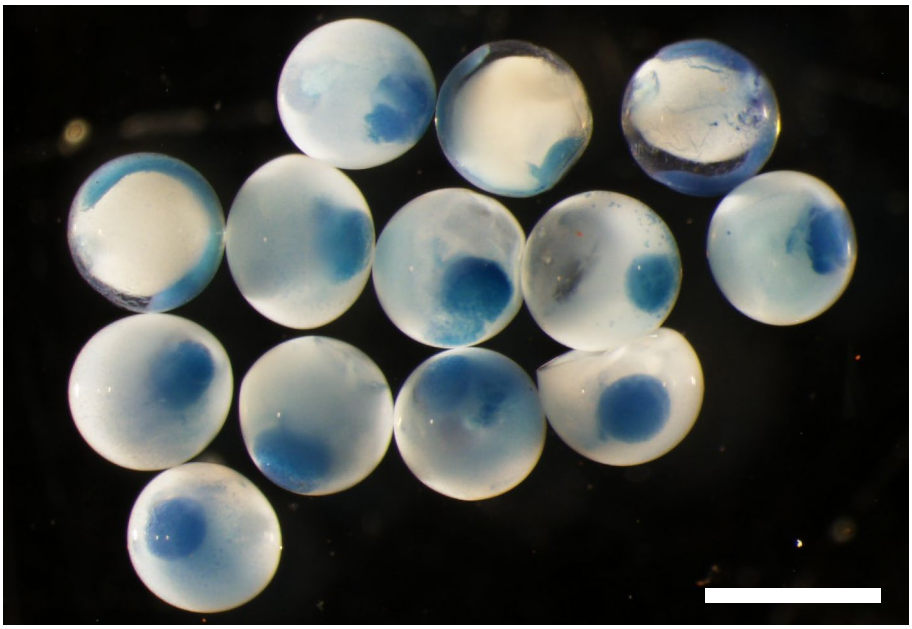

Figure 4-3. A sample of Snapper eggs following validation using the *in situ* hybridisation (ISH) molecular technique (Oxley et al. 2017). The scale bar is 1 mm.

Egg ageing and density

After they were staged, each Snapper egg was assigned a mean age in hours using the temperature dependent embryonic development relationship developed by McGlennon (2003):

$$y_{i,t} = 36.158 \,.\, e^{(-0.12t)}\, i^{0.827} \qquad \text{(Equation 4.2)}$$

where $y_{i,t}$ is the mean age of the *i*th stage at temperature *t*°C. Egg stages were transformed to the nine-stage development series by calculating the mean of the equivalent development stage described by McGlennon (2003) (see Steer et al. 2017). The time of spawning for each egg was determined by subtracting its mean age from the time it was sampled.

The density of Snapper eggs at each station was estimated by:

$$P_s = \frac{C \times D}{V} \qquad \text{(Equation 4.3)}$$

where P_s is the density of eggs in each sample (eggs.m^{-2}), *C* is the number of eggs in each sample, *D* is the maximum depth to which the net was deployed (m), and *V* is the volume of water filtered (m^3). *V* was calculated as the surface area of the mouth of the paired nets ($2 \times \pi r^2$) multiplied by the mean distance travelled from the flowmeters.

Spawning area (*A*)

The spawning area (*A*) for each survey was determined using the Voronoi natural neighbour method (Watson 1981) in ArcGIS. The georeferenced point data (eggs.m^{-2} at each station) were interpolated to predict the intermediate values through a Gaussian process governed by prior covariances. A minimum egg density of 0.1 eggs.m^{-2} was used to define the outer boundary of spawning activity. The spawning area for each survey was calculated as the total area contained within this outer boundary.

Mean daily egg production (P_0)

Mean daily egg production (P_0) was determined using the stage-based egg density estimator developed by McGarvey et al. (2018). This method is an improved approach to determine egg production for demersal fish species such as Snapper that spawn at much lower egg densities than small pelagic fishes. Briefly, the expected egg duration (i.e., time from spawning to hatch) and the ages of eggs at each station was determined as a function of temperature. The density of Snapper eggs at each station was calculated as the sum of the egg density for each development stage (I-IX). Mean daily egg production for each survey was estimated as the sum of P_0 at each station divided by the number of stations where Snapper eggs were sampled. The estimates of P_0 were weighted to account for the stratified sampling design. The advantage of this approach is that instantaneous daily egg mortality (*Z*) is specified *a priori* rather than estimated, and the method accounts for the natural mortality of eggs from the time

of spawning to the time of sampling for each egg development stage (McGarvey et al. 2018). However, as Z is specified *a priori*, sensitivity analyses were conducted over a range of values (0.2 – 0.6 day^{-1}) to ensure that the estimate of P_0 was not overly influenced by this value. A value of Z = 0.4 day^{-1} was used in all calculations of P_0.

4.2.2. Mean daily fecundity

Adult sampling and processing

Targeted sampling of adult Snapper was conducted by six commercial MSF fishers with SARDI observers present. Adult samples were collected concurrently with the ichthyoplankton surveys, with sampling undertaken in SG from 11 to 17 December 2021, and in GSV and IS from 7 to 15 January 2022. The samples were collected using standard commercial fishing methods, i.e., demersal longlines and hand lines. Adults were sampled across a broad geographic range to assess spawning condition throughout each survey area. A total of 23 samples were collected in SG and 20 samples in GSV and IS (Figure 4-4). For NGSV, this included 17 longline sets that were aggregated into three samples due to their close proximity (i.e., <5 km).

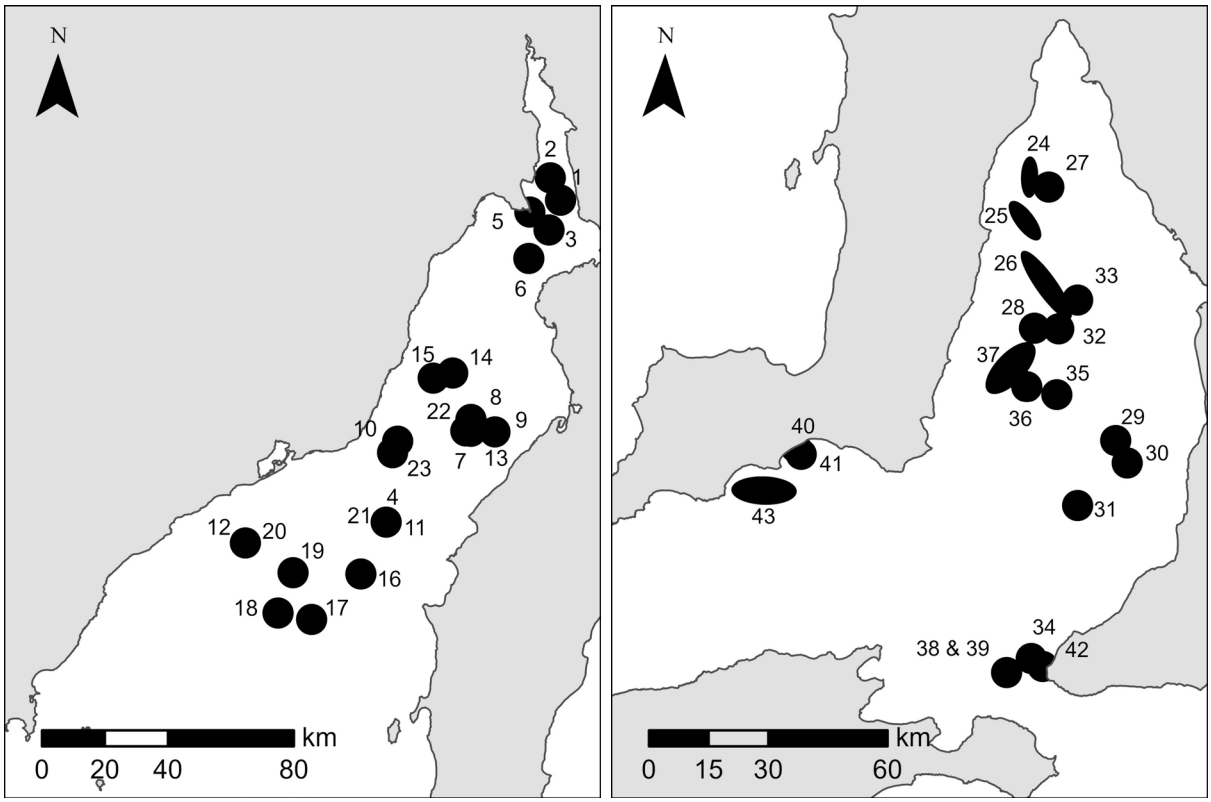

Figure 4-4. Locations of adult samples of Snapper in Spencer Gulf (left) and Gulf St Vincent (right). Black ellipses identify the locations where fish were sampled. Numbers correspond to the sample details in Table 4-4.

All fish were processed within 24 hours of capture at certified fish processing facilities, that were located in Wallaroo for the samples from SG and in Adelaide for the samples from GSV. Each fish was measured for caudal fork length (CFL) to the nearest mm and weighed to the nearest gram. The sagittal otoliths were removed, cleaned, and stored in resealable plastic bags for later ageing using a standard protocol based on the preparation and interpretation of a transverse section of a single otolith (Fowler et al. 2013). Each fish was gutted, the gonads removed and weighed to the nearest 0.01 g, and sex and stage of reproductive development determined through macroscopic visual examination (Saunders et al. 2012). All ovaries were classified macroscopically to one of five stages of development based on the presence, size, colour and visibility of oocytes (Appendix 9.2) (Saunders et al. 2012). Ovaries classified to Stage 3 (developed) and ambiguous ovaries were used in histological analysis to determine if spawning had occurred within the last 24 hours or if spawning was likely to occur in the next few hours. For these ovaries (n = 105), a small section (~5 mm thick) was removed from the centre of one lobe and preserved in formalin, acetic acid, calcium chloride (FAACC) and seawater solution. Ovaries of females classified to Stage 4 (gravid) were used to estimate batch fecundity. For these fish, a single ovary was weighed (nearest 0.01 g), opened longitudinally with a scalpel, and the oocytes washed from the connective tissue through a 500 µm sieve using filtered seawater. The retained oocytes were preserved in a 1 L solution of seawater and 10% formalin.

Female weight (*W*)

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 to 13,000 grams. The proportions of fish in each weight bin were then included as an input into the size-based spawning biomass equation as described by McGarvey et al. (2021). A multinomial error distribution was applied to determine the uncertainty for the proportion of fish in each weight bin (McGarvey et al. 2021). The midpoint of each weight bin was included in the estimate of spawning biomass.

Sex ratio (*R*)

The weight of mature (gonad Stages ≥2) males and females in each sample were used to estimate the female sex ratio (*R*) by:

$$R = \frac{\overline{W^{\text{fem}}}}{\overline{W^{\text{tot}}}} \qquad \text{(Equation 4.4)}$$

where $\overline{W^{\text{fem}}}$ and $\overline{W^{tot}}$ are the respective total weights of mature females and all mature fish across samples. Standard errors were calculated using a mean ratio estimator (Rice 1995).

Spawning fraction (*S*)

The estimates of spawning fraction (*S*) in each gulf were calculated using a combination of macroscopic staging and histological analysis. Macroscopic gonad stages for Snapper are: Stage 1 - immature, Stage 2 - developing, Stage 3 - developed, Stage 4 – gravid, and Stage 5 – regressing (Appendix 9.2) (Saunders et al. 2012). The differentiation of females into mature and immature fish was based on the macroscopic stage, i.e., Stages 2 to 5 were considered mature. Differentiating whether a mature fish was spawning or not was straight-forward for those fish classified to Stages 2, 4, and 5. Those at Stages 2 and 5 would not have spawned in the current 24-hour period, whereas those at Stage 4 would have spawned within the next few hours. However, for those at Stage 3 or any ambiguous ovaries (n = 105), there was uncertainty about spawning status. This uncertainty was resolved through analysis of the microscopic characteristics of the ovaries from histology. Histological sections were prepared from the FAACC preserved ovarian tissue samples. Ovarian tissue was embedded in paraffin wax, sectioned at 6 µm, and stained with haematoxylin and eosin (prepared by University of Adelaide Histology Services). Several sections from each ovary were viewed at 100× magnification under a dissecting microscope and examined for: the most advanced stage of oocyte development; the level of atresia; and the presence/absence of post-ovulatory follicles (POFs) (Farley and Davis 1998, Fowler et al. 1999, Saunders et al. 2012). Ovaries at macroscopic Stage 3 or were ambiguous were considered to be spawning if: (1) there were migratory nucleus oocytes present, which represents the start of the hydration process; or (2) if there were new POFs present (<24hrs old). These two features indicate that spawning would have occurred at some time during the current day or that spawning had occurred in the recent few hours, respectively (Matsuyama et al. 1988). The mean spawning fraction of each gulf was then calculated as a ratio estimate over all mature females:

$$S = \frac{\overline{N^{\text{fem,sp}}}}{\overline{N^{\text{fem}}}} \qquad \text{(Equation 4.5)}$$

Where, $\overline{N^{\text{fem,sp}}}$ is the mean number of spawning female fish across samples, and $\overline{N^{\text{fem}}}$ is the mean number of mature females across samples. Standard errors were calculated using a mean ratio estimator (Rice 1995).

Batch fecundity (*F*)

Batch fecundity (*F*) was estimated from ovaries containing hydrated oocytes (i.e., Stage 4). Formalin preserved oocytes were rinsed in a 350 µm sieve to remove the preservative, then transferred into a beaker which was filled with water to a standard volume of 1 L. The sample was thoroughly mixed to ensure the oocytes were evenly distributed throughout the solution. Ten 1 mL sub-samples were pipetted into a Sedgewick-Rafter tray and examined under a

dissecting microscope. The hydrated oocytes (>0.7 mm) in each sub-sample were counted. The estimate of batch fecundity (F^{Batch}) for each fish was calculated as:

$$F^{Batch} = \left[\frac{E_{sub} \text{ x } 1{,}000}{W_{Ovary}}\right] \text{ x } W_o \qquad \text{(Equation 4.6)}$$

Where E_{sub} is the mean count of hydrated oocytes per mL, W_{ovary} is the weight of the single ovary, and W_o is the whole weight of the paired ovaries.

The relationship between female weight (*W*) and batch fecundity (*F*) was determined using an allometric function. The relationship was developed from samples collected from 2013 to 2021. The allometric function for fecundity against weight was taken as a continuous variable, where α and β are allometric coefficients:

$$\hat{F}(W) = \alpha \cdot W^{\beta} \qquad \text{(Equation 4.7)}$$

A maximum likelihood estimator that accounted for heteroscedasticity in the spread of the residuals was used in the model fit to estimate the parameters α and β (McGarvey et al. 2021). Estimates of weight-dependent batch fecundity were calculated for the mid-point of each weight bin using the allometric relationship with normally distributed error:

$$F_w = \alpha \cdot \breve{w}_w{}^{\beta} \qquad \text{(Equation 4.8)}$$

These were included in the size-dependent estimate of spawning biomass (McGarvey et al. 2021).

4.2.3. Spawning biomass

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

$$SB = \left(\frac{P_0 \cdot A}{S \cdot R \cdot \sum_{w=1}^{\omega} F_w \cdot p_w^N}\right) \cdot \sum_{w=1}^{\omega} p_w^N \cdot \breve{w}_w \qquad \text{(Equation 4.9)}$$

Where P_0 is mean daily egg production, A is spawning area, S is spawning fraction, R is sex ratio by weight, w is weight class number, ω 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^{N} is the proportion of females in weight class w (Steer et al. 2017; McGarvey et al. 2018; McGarvey et al. 2021). The variance of each of these parameters was estimated using the delta approximation method (Casella and Berger 2002), where the overall variance of spawning biomass was estimated as:

$$V(SB) = \frac{1}{R^4 \cdot S^4 \left(\sum_{w=1}^{\omega} F_w \cdot p_w^{\mathrm{N}}\right)^4} \cdot \left\langle \begin{array}{l} \left[P_0^2 \cdot R^2 \cdot S^2 \cdot V(A) + A^2 \cdot R^2 \cdot S^2 \cdot V(P_0) + A^2 \cdot P_0^2 \cdot S^2 \cdot V(R) + A^2 \cdot P_0^2 \cdot R^2 \cdot V(S)\right] \cdot \\ \left\{\left(\sum_{w=1}^{\omega} F_w \cdot p_w^{\mathrm{N}}\right)^2 \cdot \left(\sum_{w=1}^{\omega} p_w^{\mathrm{N}} \cdot \breve{w}_w\right)^2\right\} + \\ A^2 \cdot P_0^2 \cdot R^2 \cdot S^2 \cdot \left\{\left(\sum_{w=1}^{\omega} V(F_w) \cdot \left(p_w^{\mathrm{N}}\right)^2\right) \cdot \left(\sum_{w=1}^{\omega} p_w^{\mathrm{N}} \cdot \breve{w}_w\right)^2\right\} + \\ A^2 \cdot P_0^2 \cdot R^2 \cdot S^2 \cdot \left\{\sum_{i=1}^{\omega}\left[V(p_i^{\mathrm{N}}) \cdot \left(\breve{w}_i \cdot \sum_{w \neq i}^{\omega} F_w \cdot p_w^{\mathrm{N}} - F_i \cdot \sum_{w \neq i}^{\omega} p_w^{\mathrm{N}} \cdot \breve{w}_w\right)^2\right]\right\} \end{array} \right\rangle \qquad \text{(Equation 4.10)}$$

It is recognised that daily egg production methods can have large imprecision as a result of combining several parameters that are themselves imprecise. While it is acknowledged that DEPM estimates are considered unbiased and are capable of detecting changes in spawning biomass, this imprecision requires sensitivity analyses to determine how variance for each parameter could influence the estimate of spawning biomass. For each gulf, sensitivity analyses were done for egg density (P_0), spawning area (A), spawning fraction (S), and sex ratio (R). The analyses are presented in Appendix 9.3.1. All DEPM calculations were produced using the 'DEPM' package in the R programming environment (Smart et al. 2020; R core Team 2019).

4.3. Results

4.3.1. Total daily egg production

Total area sampled

The total area sampled during the ichthyoplankton surveys has generally increased through time to ensure that each survey covered as much of the spawning area as possible. For SG, the surveys in 2019 and 2021 covered approximately double the area of the surveys in 2013 and 2018 (Table 4-2), which reflected the addition of ~80 sampling stations in the southern part of the gulf. For GSV and IS, the surveys in 2019 and 2021 were the most extensive to date and encompassed the entire spatial coverage of the GSV Stock. The increased survey area was double the area of the 2018 survey and represented a 27.7% increase compared to the survey in 2014. For both gulfs, the survey area and sampling design in 2019 and 2021 was consistent.

Table 4-2. Total survey area and estimated spawning area (*A*) for Snapper DEPM surveys in Spencer Gulf and Gulf St Vincent between 2013 and 2021.

Gulf	Year	No. stations	No. stations w. Snapper eggs	Survey area (km^2)	Spawning area (*A*) (km^2)	*A* as % of survey area
Spencer Gulf	2013	188	94	4,610	2,800	60.7
	2018	191	156	4,792	3,979	83.0
	2019	272	97	9,235	4,773	51.7
	2021	272	53	9,235	2,411	26.1
Gulf St Vincent	2014	216	167	8,022	6,435	80.2
	2018	138	111	5,059	3,126	61.8
	2019	265	173	10,245	10,112	98.7
	2021	270	89	10,381	4,293	41.4

Spawning area (*A*)

Estimates of spawning area have varied between years in response to changes in survey area and the relative abundance and distribution of Snapper eggs. For SG, the estimated spawning area of 2,411 km^2 in 2021 was 26.1% of the survey area and represented a 49.5% reduction compared to the previous survey in 2019 (Table 4-2). The large reduction in spawning area between years reflected the absence of Snapper eggs at stations in northern SG (i.e., near Whyalla) in 2021 and the low number of eggs sampled in southern SG (Figure 4-5). Similarly, for GSV, the estimated spawning area of 4,293 km^2 in 2021 was 41.4% of the survey area and represented a 57.5% reduction compared to the survey in 2019. The significant reduction in spawning area between years was largely driven by the low number of Snapper eggs sampled in southern GSV and IS (Figure 4-5). For both gulfs, the proportion of spawning area to total survey area in 2021 was the lowest recorded (Table 4-2).

Distribution and abundance of eggs

The distribution and abundance of Snapper eggs in SG varied considerably between surveys. In 2021, a total of 104,436 teleost eggs were collected from the 272 stations sampled. From these, a total of 222 Snapper eggs were identified at 53 stations (19% of total stations) using the ISH molecular technique. In general, Snapper eggs were distributed as several discrete patches throughout the survey area, with the highest abundances in the central part of SG adjacent to Lucky Bay, north of the *Santa Anna* wreck, and south of *Jurassic Park* (Figure 4-5). The density of Snapper eggs throughout these areas was generally 0.1 to 3.0 eggs.m^{-2}, with the highest density of 6.8 eggs.m^{-2} recorded near Lucky Bay. There was also a patch of Snapper eggs sampled at low densities (i.e., <1.0 egg.m^{-2}) toward the southern extremity of the survey area offshore from Balgowan. Unlike previous surveys in SG, there were no Snapper eggs sampled in the northern part of the gulf (i.e., near Whyalla and Port Pirie), and very few eggs were sampled on the western side of the gulf south of Cowell. The spatial distribution and abundance of Snapper eggs in SG was consistent with the distribution of spawning fish observed in adult samples throughout the survey area (see Section 4.3.2).

The spatial distribution of Snapper eggs in 2021 was noticeably contracted compared to the survey in 2019, despite sampling an almost identical number of eggs (Figure 4-5). In 2019, a total of 223 Snapper eggs were sampled at 97 stations (36% of total stations), which were scattered in small patches at low densities (<1.0 egg.m^{-2}) throughout the survey area. Comparatively, in 2018 Snapper eggs were broadly distributed at low densities (0.1 to 3.0 eggs.m^{-2}) at most stations sampled (156 of 191 stations; 82%), with the highest density (14.3 eggs.m^{-2}) recorded north of the *Santa Anna* wreck. In 2013, relatively high densities of eggs (5 to 15 eggs.m^{-2}) were sampled adjacent to the *Illusion* wreck and near Point Lowly, although egg densities were generally low and patchy throughout the survey area.

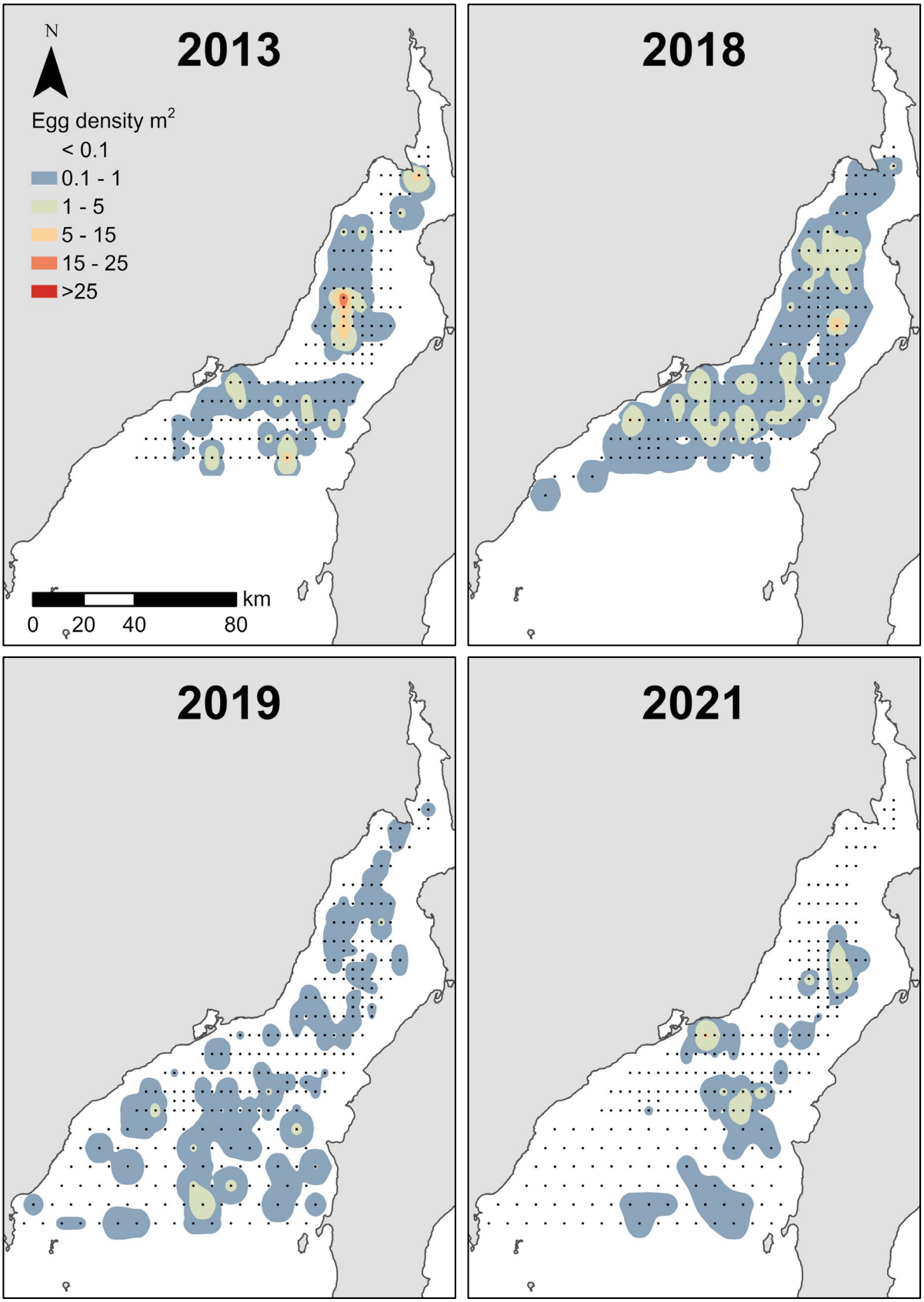

Figure 4-5. Spatial distribution and abundance (eggs.m^{-2}) of Snapper eggs sampled from surveys in Spencer Gulf between 2013 and 2021. Symbols (●) identify the stations sampled in each survey.

For GSV and IS in 2021, a total of 92,748 teleost eggs were collected from the 270 stations sampled. From these, 785 Snapper eggs were identified at 89 stations (33% of total stations). The abundance of Snapper eggs varied spatially throughout the survey area. The majority of Snapper eggs were concentrated in central-western GSV, which had a large area of moderate to high egg densities (i.e., 5 to 25 eggs.m^{-2}) that extended from Port Vincent to Tapley Shoal (Figure 4-6). There were numerous stations within this large area that had egg densities of 8 to 20 eggs.m^{-2}. This was the only area in the survey with egg densities of >5 eggs.m^{-2}. There were also several smaller patches of Snapper eggs at lower densities (1 to 5 eggs.m^{-2}) throughout the survey area including near the Ardrossan barge, offshore from Sellicks Beach, adjacent to Cape Jervis, and in IS (Figure 4-6). The distribution and abundance of Snapper eggs in GSV and IS was consistent with the distribution of adult samples and the reproductive status of fish in the different areas. That is, there were numerous schools of large spawning fish noted by SARDI researchers during the adult sampling on the western side of GSV that aligned with the area of moderate to high egg densities, and there were other schools of spawning fish in areas where the localised patches of snapper eggs were sampled (see Section 4.3.2).

The distribution and abundance of Snapper eggs throughout GSV and IS has varied considerably between surveys. Similar to SG, the spatial distribution of eggs in 2021 was contracted compared to the survey in 2019 (Figure 4-6). In 2019, a total of 1,586 Snapper eggs were sampled at 173 stations (65% of total stations), which were broadly distributed throughout the survey area and included several large areas of low to moderate egg densities (i.e., 1 to 10 eggs.m^{-2}). The relatively reduced abundance and distribution of eggs in 2021 was particularly evident in southern GSV and IS, where large numbers of eggs were sampled in 2019. In 2018, Snapper eggs were broadly distributed throughout the survey area, although most of IS was not sampled due to poor weather. The highest abundances of eggs were sampled in northern GSV (i.e., north of a line from Adelaide to Port Vincent), adjacent to Black Point. In 2014, the highest abundances of eggs were sampled near Tapley Shoal.

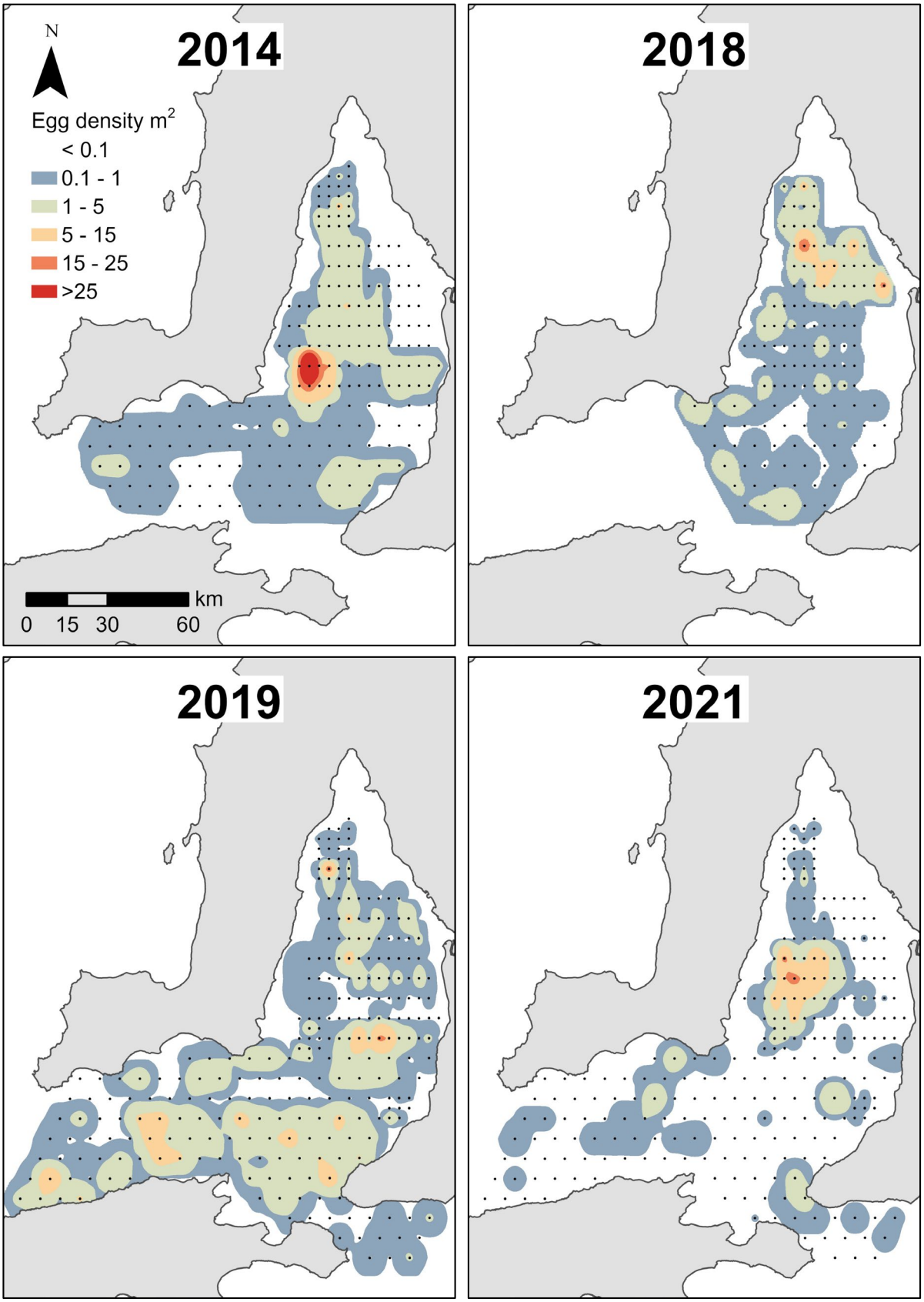

Figure 4-6. Spatial distribution and abundance (eggs.m^{-2}) of Snapper eggs sampled from surveys in Gulf St Vincent and Investigator Strait between 2013 and 2021. Symbols (●) identify the stations sampled in each survey.

Mean daily egg production (P_0)

The time of the day when each egg was spawned was calculated by subtracting its estimated age from the time it was sampled. The distribution of spawning times indicated that the majority (79.7%) of eggs were spawned between 18:00 and 03:00 hours, with a peak at 23:00 hours (Figure 4-7). This pattern was consistent in both gulfs. For SG, the increase in spawning activity from the evening to early morning coincided with a rising tide that peaked at ~03:30 hours. For GSV, it aligned with a high tide on dusk (20:00 to 22:00 hours) and a low tide in the early morning (02:00 to 03:30 hours).

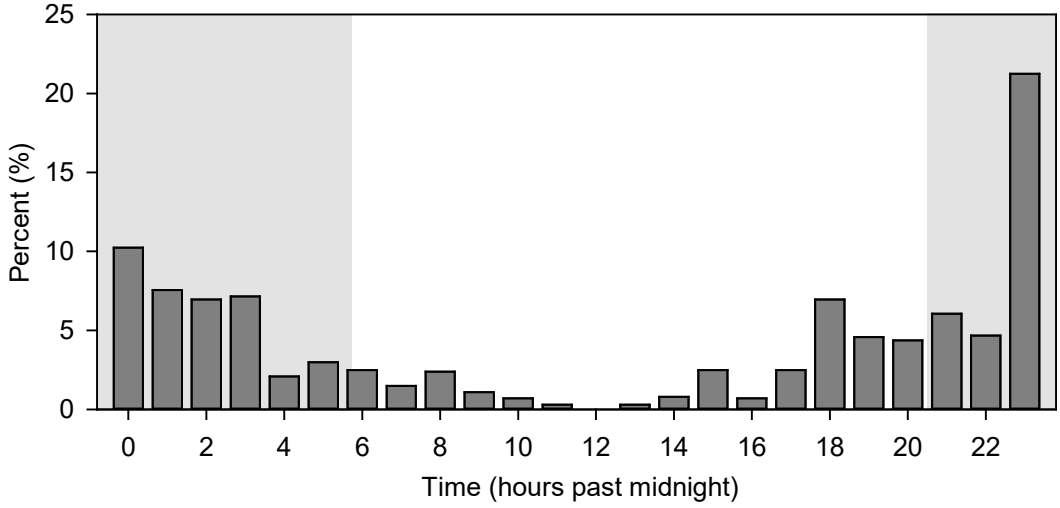

Figure 4-7. Daily cycle of spawning for Snapper in Spencer Gulf and Gulf St Vincent based on the back-calculated spawning time of eggs sampled in 2021. There was considerable similarity in the distribution of spawning times for the two gulfs and therefore the data were combined (n = 1,007). Shaded areas – night; light area – day.

For SG, the estimated mean (± SE) daily egg production (P_0) in 2021 was 0.68 (± 0.13) eggs.m^{-2} (Table 4-3). This represented a 44.7% increase compared to the previous survey in 2019 (P_0 = 0.47 ± 0.06 eggs.m^{-2}), which largely resulted from sampling a similar number of eggs from relatively fewer stations. The estimates of P_0 in 2021 and 2019 were both lower than in 2018 and 2013. For GSV and IS, estimated mean (± SE) daily egg production in 2021 was 2.12 (± 0.39) eggs.m^{-2}, which was similar to P_0 in 2019 (2.24 ± 0.21 eggs.m^{-2}) (Table 4-3). The similarity in P_0 between years reflected a proportional decrease in the number of eggs collected and the number of stations with Snapper eggs. The estimates of P_0 in 2021 and 2019 were similar to 2018, but considerably less than 2014.

The assumed rate of natural egg mortality (Z = 0.4 day^{-1}) did not significantly affect the estimates of P_0 in either gulf. Sensitivity analyses showed that altering Z within reasonable bounds (i.e., 0.2 to 0.6 day^{-1}) changed P_0 by <10% (Appendix 9.2).

Table 4-3. Estimated mean daily egg production (P_0) for surveys in Spencer Gulf and Gulf St Vincent between 2013 and 2021. All estimates of P_0 were determined using an instantaneous egg mortality rate (Z) of 0.4 day^{-1}. SE – standard error.

Gulf	Year	No. stations	No. stations with eggs	No. snapper eggs	P_0 (eggs.m^{-2})	SE
Spencer Gulf	2013	188	94	101	2.37	0.54
	2018	191	156	1,204	0.99	0.10
	2019	272	97	223	0.47	0.06
	2021	272	53	222	0.68	0.13
Gulf St Vincent	2014	216	167	203	11.56	4.31
	2018	138	111	1,222	1.83	0.26
	2019	265	173	1,586	2.24	0.21
	2021	270	89	785	2.12	0.39

4.3.2. Mean daily fecundity

Distribution of adult samples

For SG, a total of 505 Snapper were sampled from 23 locations that were distributed from Fitzgerald Bay near Whyalla to the Steamer Channel south of Cape Elizabeth (Figure 4-4). Fish ranged in size from 16.6 to 90.2 cm CFL and in weight from 0.08 to 11.19 kg (Table 4-4). The ages of fish varied between sampling locations and ranged from 1 to 24 years. In general, the samples from northern and central SG were characterised by medium to large fish (i.e., 50 to 80 cm CFL), whereas those from the south had a higher proportion of small to medium fish (i.e., 30 to 50 cm CFL). Snapper from the south were generally older for their size than those from the north.

For GSV and IS, a total of 479 Snapper were sampled from 20 locations that were distributed throughout GSV, including near Cape Jervis and Foul Bay (Figure 4-4). Most fish were sampled from northern and central GSV. The fish ranged in size from 14.0 to 87.8 cm CFL and in weight from 0.25 to 10.79 kg (Table 4-4). The ages of fish ranged from 2 to 17 years.

Table 4-4. Summary of sample details for adult Snapper collected in Spencer Gulf and Gulf St Vincent as part of the 2021 DEPM. Map code corresponds to Figure 4-4. Length (cm), weight (kg), and age (years) values are total range (min.–max.). CFL – caudal fork length.

Map code	Sample code	Total no. fish	No. males	No. females	Length (CFL) (cm)	Weight (kg)	Age (years)
	Spencer Gulf	**505**	**239**	**266**	**16.6 – 90.2**	**0.08 – 11.19**	**1 – 24**
1	PP12/2101	92	40	52	33.9 – 73.0	0.78 – 5.69	3 – 8
2	PP12/2102	37	21	16	42.0 – 86.2	1.28 – 10.21	3 – 12
3	PP12/2103	8	1	7	44.0 – 81.3	1.72 – 8.21	3 – 12
5	PP12/2104	40	23	17	38.9 – 79.1	1.14 – 8.01	3 – 14
6	PP12/2105	24	11	13	29.0 – 60.2	0.52 – 4.10	1 – 7
22	MB12/2101	22	14	8	27.7 – 85.4	0.88 – 9.25	3 – 15
13	PB12/2101	29	13	16	27.1 – 90.2	0.48 – 10.39	2 – 13
14	PB12/2102	12	5	7	71.9 – 85.7	6.23 – 9.71	6 – 15
15	PB12/2103	25	11	14	33.3 – 88.8	0.92 – 11.19	3 – 15
7	WA12/2101	28	16	12	56.3 – 83.3	3.34 – 9.28	6 – 16
8	WA12/2102	23	9	14	34.1 – 86.5	0.77 – 10.53	3 – 15
9	WA12/2103	7	2	5	32.2 – 45.3	0.74 – 1.74	3 – 8
10	WA12/2104	9	4	5	27.6 – 40.6	0.47 – 1.37	2 – 8
12	ES12/2101	18	8	10	33.3 – 70.0	0.72 – 5.34	4 – 12
20	ES12/2102	24	15	9	23.7 – 85.4	0.28 – 4.14	3 – 7
4	JP12/2101	13	5	8	27.8 – 82.5	0.50 – 9.64	3 – 15
21	JP12/2102	18	7	11	23.7 – 85.4	0.30 – 9.16	5 – 15
19	MT12/2101	21	9	12	16.7 – 85.0	0.08 – 9.12	1 – 13
23	WA12/2105	2	2	0	22.9 – 32.4	0.28 – 0.69	2 – 5
11	WA12/2106	12	4	8	31.1 – 76.3	0.75 – 7.80	5 – 13
17	WA12/2107	24	11	13	16.7 – 62.6	0.10 – 3.78	1 – 16
16	WA12/2108	12	7	5	29.4 – 79.2	0.58 – 7.61	3 – 24
18	WA12/2109	5	1	4	16.6 – 56.4	0.11 – 2.99	1 – 7
	Gulf St Vincent	**479**	**276**	**203**	**14.0 – 87.8**	**0.25 – 10.79**	**1 – 17**
24	AR01/2201	42	21	21	38.4 – 86.6	1.32 – 9.21	6 – 17
25	AR01/2202	34	20	14	36.9 – 84.7	1.08 – 9.62	3 – 16
26	AR01/2203	56	34	22	32.4 – 79.8	0.63 – 7.57	4 – 13
27	AR01/2204	11	8	3	37.3 – 80.8	0.97 – 9.44	4 – 15
32	NH01/2201	30	16	14	33.5 – 83.1	0.83 – 8.53	5 – 15
33	NH01/2202	20	13	7	53.2 – 84.3	2.61 – 8.85	5 – 16
35	NH01/2203	16	8	8	25.4 – 48.1	0.38 – 2.12	3 – 8
28	ST01/2201	9	6	3	31.6 – 81.0	0.67 – 7.71	4 – 15
36	ST01/2202	25	18	7	36.5 – 82.0	1.04 – 8.73	4 – 15
37	ST01/2203	20	7	13	57.0 – 87.8	3.19 – 10.79	8 – 16
34	CJ01/2201	53	29	24	24.5 – 76.4	0.36 – 8.11	3 – 13
38	CJ01/2202	52	33	19	30.6 – 60.8	0.61 – 3.82	4 – 13
39	CJ01/2203	18	12	6	36.4 – 54.8	0.87 – 3.09	4 – 8
42	CJ01/2204	3	1	2	30.1 – 37.5	0.56 – 1.10	4 – 5
43	FO01/2201	1	0	1	73.8	6.78	13
40	FO01/2202	2	0	2	39.5 – 49.4	1.25 – 2.30	5 – 9
41	FO01/2203	20	13	7	56.8 – 84.3	3.03 – 9.39	8 – 15
29	NH01/2204	40	23	17	21.7 – 82.7	0.25 – 9.31	3 – 14
30	NH01/2205	10	5	5	26.0 – 76.3	0.39 – 6.72	3 – 15
31	NH01/2206	17	9	8	14.0 – 79.1	0.28 – 8.14	1 – 16

Mean female weight (*W*)

For SG in 2021, small to medium sized Snapper of <4 kg were prominent in the samples and accounted for 64% of fish captured (Figure 4-8). The mean (± SE) female weight was 3.72 ± 0.15 kg. In comparison to 2018 and 2019, there was a substantial decrease in the proportion of small fish <2 kg and an increase in the proportions of medium (2–4 kg) and large (5–11 kg) fish. For GSV and IS in 2021, the distribution of female weight was bimodal with peaks at ~1 kg and 7–8 kg (Figure 4-8). The mean (± SE) female weight for GSV was 3.78 kg ± 0.19. In comparison to 2019, there was a marginal increase in the proportion of small to medium Snapper (<4 kg) which accounted for 65% of the fish sampled. Overall, the distributions of female weight for GSV have remained relatively consistent between surveys.

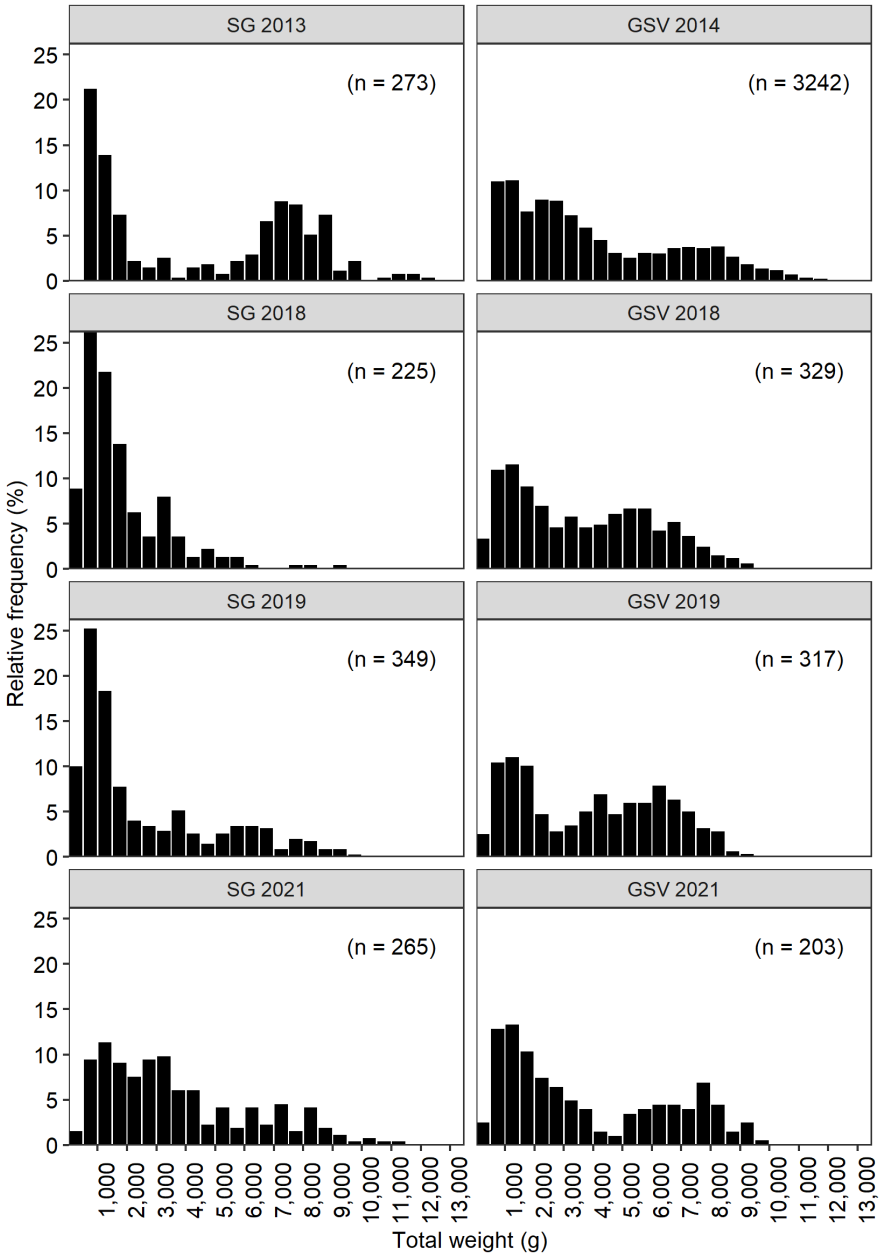

Figure 4-8. Proportional female weight distributions of Snapper for Spencer Gulf (SG) and Gulf St Vincent (GSV) between 2013 and 2021. The number of samples is shown in parentheses.

Sex ratio (*R*)

Female sex ratios by weight were calculated from the 505 mature Snapper sampled from SG and 479 from GSV. For SG, the sex ratio was slightly biased towards females at 56%. In contrast, the sex ratio for GSV and IS was slightly biased toward males, with a female sex ratio of 45%. For both gulfs, the estimates of female sex ratio in 2021 were within the upper and lower estimates of previous surveys (i.e., between 0.40 and 0.60) (Table 4-5).

Table 4-5. Population sex ratio (*R*) by weight for Spencer Gulf and Gulf St Vincent for DEPM surveys between 2013 and 2021. SE – standard error.

Gulf	Year	Female sex ratio	SE
Spencer Gulf	2013	0.57	0.06
	2018	0.40	0.06
	2019	0.55	0.04
	2021	0.56	0.02
Gulf St Vincent	2014	0.59	0.04
	2018	0.40	0.03
	2019	0.54	0.03
	2021	0.45	0.03

Spawning fraction (*S*)

There was a considerable difference in spawning fractions between gulfs. For SG, the overall spawning fraction (*S*) was 0.44, which was 18.9% higher than the estimate of 0.37 in 2019 (Table 4-6). There was significant spatial variation in the estimates of *S* between samples (i.e., 0.00 to 1.00), with *S* highest in central SG (0.64 to 1.00) and lowest in the north (0.00 to 0.31) (Appendix 9.2). For GSV and IS, the estimated spawning fraction of 0.83 was similar to *S* in 2019. Spawning fractions in GSV were consistently high among samples (0.57 to 1.00).

Table 4-6. Estimated spawning fraction (*S*) for Spencer Gulf and Gulf St Vincent for DEPM surveys between 2013 and 2021. Data source identifies whether S was calculated from concurrent adult sampling or published literature. SE – standard error.

Gulf	Year	Spawning fraction (*S*)	SE	Data source
Spencer Gulf	2013	0.72	0.05	Saunders (2009)
	2018	0.72	0.05	Saunders (2009)
	2019	0.37	0.05	Concurrent sampling
	2021	0.44	0.11	Concurrent sampling
Gulf St Vincent	2014	0.72	0.05	Saunders (2009)
	2018	0.72	0.05	Saunders (2009)
	2019	0.85	0.10	Concurrent sampling
	2021	0.83	0.04	Concurrent sampling

Batch fecundity (*F*)

The relationship between batch fecundity (*F*) and total female weight (*W*) was best described by an allometric linear regression that accounted for the increasing variance in *F* as fish weight increases (Figure 4-9). There were no statistical differences 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.

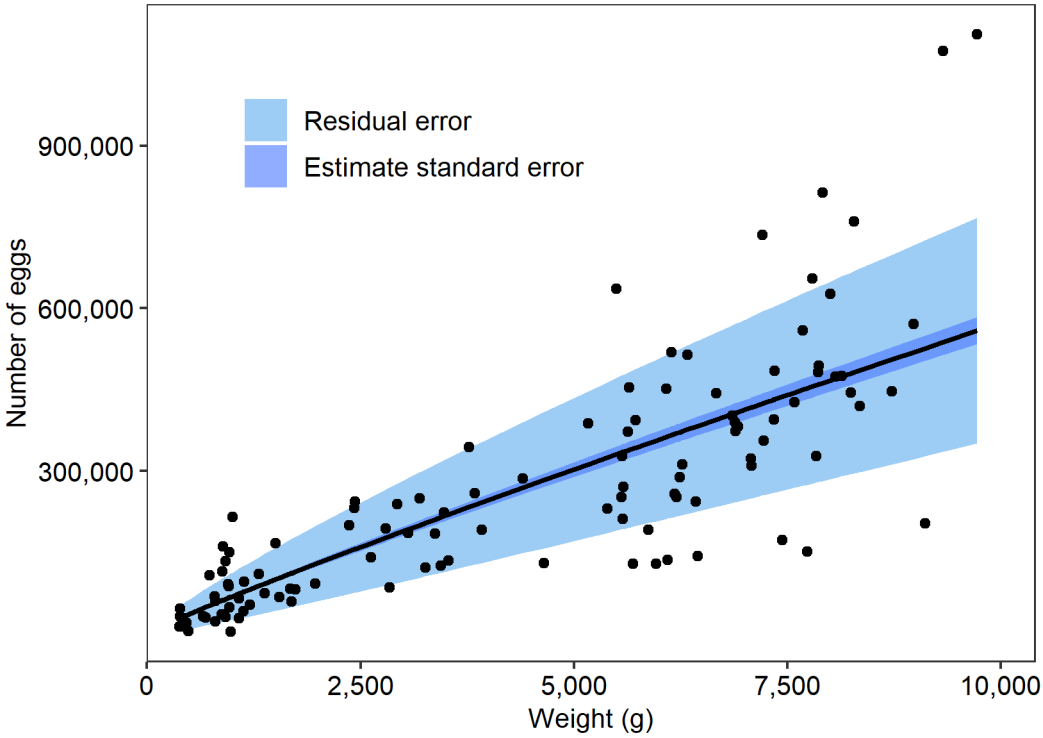

Figure 4-9. Relationship between batch fecundity (F) (number of eggs per batch) and total female body weight (W) (g) for Snapper from South Australia's gulfs (n = 110). The error bands show the standard error (SE) of the residuals (light blue) and the SE of the F_w estimates (dark blue).

4.3.3. Spawning biomass

The estimated spawning biomass of Snapper in each gulf has differed between surveys. It is important to recognise that each estimate of spawning biomass relates to the area covered by each survey, and therefore it is difficult to directly compare between years when the survey areas were different (i.e., 2013 and 2018). The differences in survey design are accounted for in the estimates of fishable biomass from the SnapEst model.

The estimated spawning biomass of Snapper for SG in 2021 was 108 ± 65 t. This represents a 39.0% decrease compared to the previous survey in 2019 (177 ± 34 t), which largely resulted from a 49.5% reduction in spawning area and an 18.9% increase in spawning fraction (Table 4-7). The estimates of spawning biomass in 2019 and 2021 were both lower than those in 2013 and 2018, despite expanding the survey area by double in the most recent surveys.

The estimated spawning biomass of Snapper for GSV and IS in 2021 was 404 ± 124 t. This was a 50.2% decrease compared to the 2019 survey (811 ± 125 t) (Table 4-7), which primarily related to the reduction (57.5%) in spawning area observed. The difference in the estimates of spawning biomass between the 2018 and 2019 surveys related to the reduced survey area in 2018.

Table 4-7. Summary of parameters and the estimated spawning biomass (*SB*) (t) of Snapper for DEPM surveys in Spencer Gulf and Gulf St Vincent between 2013 and 2021. P_0 – mean daily egg production (eggs.m^{-2}); *A* – spawning area (km^2); *W* – mean female weight (kg); *R* – sex ratio; *F* – batch fecundity of *W*; *S* – spawning fraction; SE – standard error.

Gulf	Year	P_0	*A*	*W*	*R*	*F*	*S*	*SB* (t)	SE
Spencer Gulf	2013	2.37	2,800	3.69	0.57	222,146	0.72	280	152
	2018	0.99	3,979	2.44	0.40	146,334	0.72	192	63
	2019	0.47	4,773	2.37	0.55	142,088	0.37	177	34
	2021	0.68	2,411	3.71	0.56	223,501	0.44	108	65
Gulf St Vincent	2014	11.56	6,434	4.25	0.59	256,110	0.72	2,780	1,444
	2018	1.83	3,126	4.27	0.40	257,323	0.72	343	130
	2019	2.24	10,111	3.91	0.54	235,489	0.85	811	125
	2021	2.12	4,293	3.78	0.45	227,876	0.83	404	124

4.4. Discussion

The DEPM has become an important component of the stock assessment for Snapper in South Australia, primarily as an integral input into the SnapEst model. The regional estimates of spawning biomass for SG and GSV/IS in 2021 were the fourth in a time-series since 2013 and represent the third application of DEPM for Snapper in both gulfs since 2018 (i.e., 2018, 2019, and 2021). In 2021, the estimates of spawning biomass for both SG and GSV/IS were lower than previous estimates in 2019 (reduced 39% and 50%, respectively). Several hypotheses may account for the decreases in spawning biomass estimated from the DEPM.

4.4.1. Interpretation of results

Applications of the DEPM for multiple fish species have demonstrated that estimated spawning biomass is strongly correlated with spawning area (Gaughan et al. 2004, Ward et al. 2021). In 2021, the spawning area in each gulf decreased by approximately half compared to the surveys in 2019 (SG decreased 49.5%, GSV/IS decreased 57.5%). The reductions in spawning area were the primary factor in the proportional decreases in spawning biomass. The reductions in spawning area may relate to changes in fish behaviour in response to the closures of the fisheries. Snapper form dense aggregations throughout its protracted spawning period which results in the localised concentration of adult fish (Wakefield 2006, 2010, Saunders 2009). In 2021, the fish had been relatively undisturbed for the two years between the 2019 and 2021 DEPM surveys. Therefore, it is likely that they formed aggregations during spring and summer that concentrated the spatial distribution of adults and subsequently reduced the spawning area. Such behaviour is supported by the concentrated area of moderate to high egg density in central GSV that coincided with several schools of spawning fish, and the localised patches of moderate egg densities in central SG which resulted in a higher estimate of P_0 in 2021. In contrast, the extensive spatial distributions of eggs in previous surveys (i.e., before the gulfs were closed to fishing) suggests that the adult fish had been broadly dispersed throughout both gulfs. Previous studies that monitored fine-scale movement using acoustic telemetry have demonstrated that Snapper in SA's gulfs generally operate over a small home range but can disperse rapidly in response to disturbance, such as targeted fishing (Fowler et al. 2017b, Fowler unpublished). Consequently, it is likely that the contracted spawning area in 2021 was associated with the concentration of adult fish, reflecting the lack of disturbance from fishing over the past two years.

The spatial distribution and abundance of Snapper eggs in SG and GSV were strongly correlated with the distribution of spawning adults (Figure 4-10). For SG, samples of Snapper were broadly distributed throughout the survey area, with the majority of spawning fish located

in the central region. There were also numerous sites that have traditionally supported schools of Snapper during the spawning period that did not support an aggregation at the time of the survey, including the wrecks of the *Illusion* and *Estelle Star*. Furthermore, there was significant latitudinal variation in reproductive activity that aligned with the spatial distribution of eggs (Figure 4-10). There were no Snapper eggs sampled in northern SG (i.e., north of Port Davis) and the adults in this region had a low spawning fraction, which indicated that very few fish (5%) were spawning at the time of the survey. The majority of eggs were sampled in the central part of SG (i.e., from Port Davis to Wallaroo) where the adults had a much higher spawning fraction of 0.81. Similarly, south of Wallaroo, the localised patches of eggs corresponded to adult samples with spawning fractions of >0.75.

For GSV and IS, the area of moderate to high egg densities on the western side of GSV was concurrently occupied by several large schools of Snapper that had high spawning fractions (0.88 to 1.00) (Figure 4-10). The localised patches of Snapper eggs near Ardrossan, Sellicks Beach, and Cape Jervis each corresponded to samples of adult fish with high spawning fractions (i.e., 0.79 to 1.00). Consequently, the strong association between the spatial distribution of Snapper eggs and spawning adults in both gulfs supports the hypothesis that the fish were aggregated at the time of the surveys.

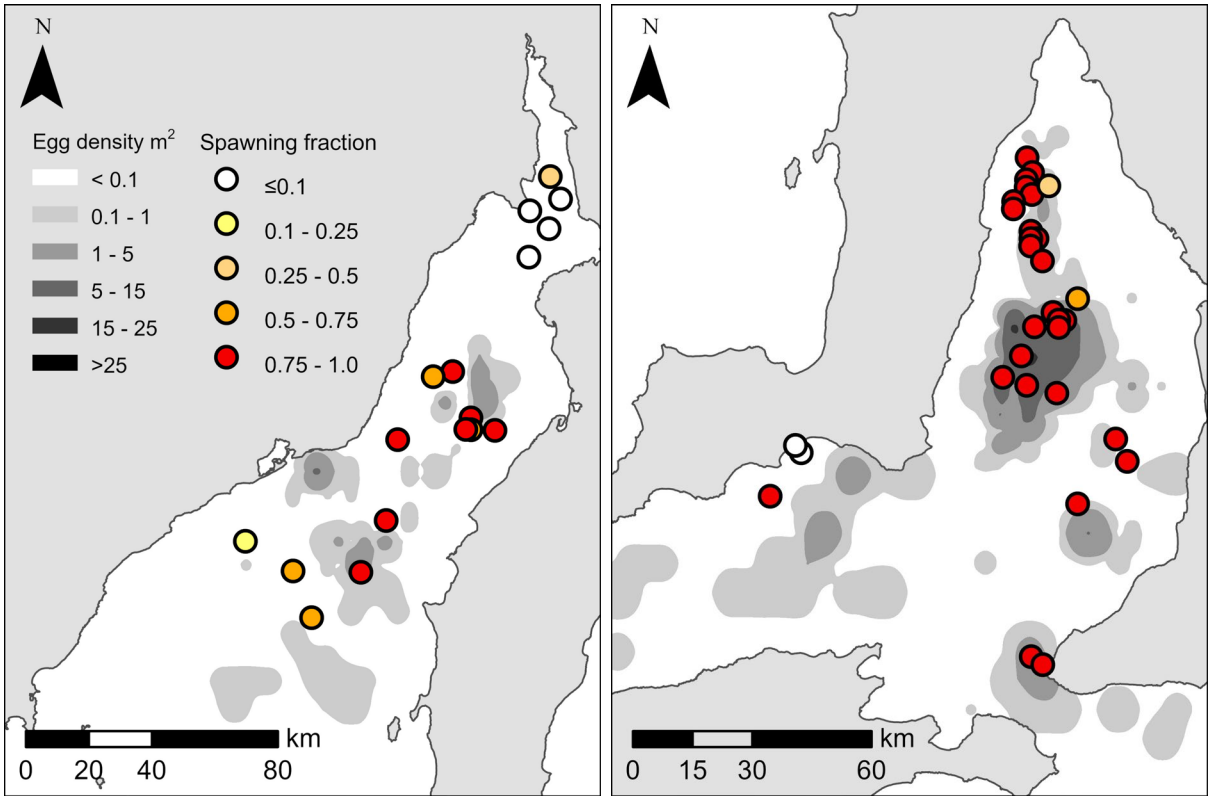

Figure 4-10. Spatial distribution of Snapper eggs (eggs.m^{-2}) and adult samples from Spencer Gulf (left) and Gulf St Vincent (right). Adult samples are colour-coded by spawning fraction. Data are presented in Appendix 9.2.

Latitudinal variation in the timing of spawning is a characteristic of Snapper throughout its geographic distribution and is primarily associated with regional differences in seasonal water temperature regimes (Sheaves 2006, Wakefield 2006, Wakefield et al. 2015). The optimal temperature range for the spawning of Snapper and the survival of eggs and larvae is 18 to 22°C (Pecl et al. 2014). The water temperature in both gulfs during the 2021 surveys was highest in the north and decreased southward, ranging from 18.7 to 20.7°C in SG and 17.9 to 21.8°C in GSV and IS (Appendix 9-2). The water temperature in northern SG was 20.1 to 20.7°C, which is in the middle of the optimal temperature range for spawning, and therefore it is unlikely that water temperature was directly responsible for the lack of reproductive activity.

4.4.2. Improvements, sensitivity analyses, and method development

Two integral components of the DEPM methodology were improved in 2021 to address uncertainties identified in previous assessments (Fowler et al. 2019, 2020). The first was the development of a geographically extensive adult sampling program that was undertaken concurrently with the ichthyoplankton surveys to improve the estimates of adult parameters. This was achieved through collaboration with commercial MSF fishers and provided comprehensive data on the spatial distribution of spawning activity throughout SG and GSV/IS. Accurate estimates of adult parameters are fundamental for estimating the spawning biomass of Snapper using the DEPM. Secondly, the protocol to identify Snapper eggs in plankton samples was refined using a multi-stage process that applied both morphological and molecular techniques (Oxley et al. 2017). This approach provides the highest level of confidence for egg identification.

Multiple sensitivity analyses were undertaken on the estimates of spawning biomass (Appendix 9.3). Key among these were testing the influence of (i) inclusion/exclusion of northern SG from the estimate of spawning biomass because very few fish in this area contributed to the spawning component of the population at the time of the survey (Appendix 9.3.2), and (ii) exploring geostatistical methods to calculate total daily egg production (i.e., $P_0 \times A$) to account for the correlation between the two independent parameters (Appendix 9.3.3). The inclusion of several samples of adults with very low spawning fractions in northern SG considerably reduced the overall population estimate of spawning fraction (from 0.71 to 0.44). Such samples have previously been excluded in applications of the DEPM for other species (e.g., Redbait; Grammer et al. 2022). To investigate the influence of removing adults with low spawning fractions in NSG, two estimates of spawning biomass were generated that accounted for the removal or inclusion of these fish (Appendix 9.3.2). There was a 38% decrease in spawning biomass when the adult parameters were re-estimated after the removal of these fish. Nevertheless, there was a negligible (<1%) difference in the subsequent estimate of fishable biomass from SnapEst (Appendix 9.5.3).

It is evident that spawning area has a significant effect on the estimates of spawning biomass of Snapper using the DEPM. Traditionally, spawning area (A) and daily egg production (P_0) are considered independent parameters in the DEPM equation (Eq. 4.1), despite being inherently correlated, calculated using the same dataset, and are therefore not truly independent. Therefore, we considered the application of geostatistical methods to estimate total daily egg production (i.e., $P_0 \times A$) as a logical approach to overcome this issue and to reduce uncertainty in the estimate of total egg production. Subsequently, several geostatistical methods were explored during the DEPM analysis and incorporated into SnapEst as sensitivity analyses to understand their effect on subsequent estimates of fishable biomass (Appendix 9.3.3). The traditional approach (i.e., independent parameters) was retained in this assessment because the application of geostatistical methods for DEPM remains largely untested and requires further refinement.

The estimate of total egg production could also be refined by empirically assessing the effectiveness of different sampling methodologies to collect Snapper eggs. This could be achieved through a targeted study to understand the vertical distribution of Snapper eggs in the water column and could incorporate a hydroacoustic survey to better understand the aggregating behaviour of Snapper during the spawning period. Reducing the error of each parameter by refining the methodology would contribute to reducing the cumulative error of estimated spawning biomass, and ultimately, the estimates of fishable biomass from SnapEst.

5. STOCK ASSESSMENT MODEL – SNAPEST

5.1. Introduction

The SA Snapper fishery stock assessment model, SnapEst, was developed with FRDC support as a dynamic, spatial, age- and length-structured model (McGarvey and Feenstra 2004). 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 6-1). A significant change was incorporated into the model for the 2020 Snapper assessment (Fowler et al. 2020), which involved updating the spatial resolution of the model to conform with the revised understanding of stock structure (Fowler et al. 2017a). Previously, SnapEst had separate model regions for the northern and southern parts of the two gulfs. In the updated 2020 model, the spatial structure was modified to estimate outputs for the three stocks (i.e., SG/WCS, GSVS, and the SE Region), which were assumed to be independent, i.e., there was no movement or reproductive exchange between them. For this assessment, the model estimates for the SG/WCS and GSVS were based at the biological stock scale to remain consistent with the previous assessment (Fowler et al. 2020). However, for the SE Region, the model estimates were aligned to the post-MSF reform management zone to inform the setting of total allowable catches (TACs) in this region.

As in previous assessments, the estimates of spawning biomass from the DEPM for SG and GSV are a primary data source for SnapEst. When spawning biomass estimates are not available, notably for the SE Region and prior to DEPM surveys in the gulfs, a CPUE derived from the commercial fishery is fitted as the index of relative abundance. Length-at-age samples, and catch totals including (commercial, recreational and charter) are the other primary SnapEst data sources. Major changes to the model runs in this assessment include: (1) adding new estimates of spawning biomass from the 2021 DEPM in the two gulfs; (2) switching the CPUE index fitted in the SE Region to LL kg.fisher-day^{-1} (previously, HL kg.fisher-day^{-1}) and, for 2003 onward, using kg/hook (rather than kg/fisher-day^{-1}).

5.2. Methods

Seven datasets were used as inputs to the SnapEst model:

1) Total commercial catch (t) by region from 1984 to 2022;
2) Handline CPUE derived from commercial catch and effort data as an index of abundance for SG/WCS and GSVS prior to the first DEPM spawning biomass estimates in 2013, and longline CPUE for all years in the SE Region;
3) Estimates of recreational catch data from the telephone and diary surveys undertaken in 2000/01 (Henry and Lyle 2003), 2007/08 (Jones 2009) and 2013/14 (Giri and Hall 2015), recreational catch was interpolated for years outside of surveys based on data from the 2007/08 survey;
4) Retained charter boat catch from logbooks data since September 2005;
5) Annual age structure proportions from commercial catch sampling from years 1999 to 2003, 2004 to 2011, 2013 to 2021 in SG, years 1999 to 2003, 2005 to 2011, 2013 to 2021 in GSV, and years 1999, 2002, 2006 to 2014, 2016, 2019 to 2021 in SE;
6) Snapper length-frequency data from commercial catch sampling; and
7) DEPM estimates of spawning biomass from 2013 for SG, 2014 for GSV, and 2018, 2019 and 2021 for both gulfs.

Commercial catch totals for HL and LL were augmented for this assessment by the inclusion of catch weights from Snapper that were collected through the adult sampling program from January 2020 to January 2022. Whilst Snapper was prohibited from being targeted and taken in the SG/WC and GSV stocks, it is likely that a degree of post-release mortality resulting from incidental catches of Snapper may have occurred. This, however, was unsubstantiated and not considered in this assessment.

Handline CPUE (kg.fisher-day^{-1}) was used as the preferred fishery-based index of Snapper abundance for the gulfs as HL fishing practices for these two regions have not changed as much over time compared to LL. The catch rates by LL have shown evidence of hyperstability (Fowler and McGlennon 2011, Fowler et al. 2019). Further to this, LL equipment used to target Snapper has improved considerably through time, which has resulted in the increased efficiency of this gear type. 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. For the SE Region, the 2022 assessment model now fits to LL CPUE (kg.fisher-day^{-1}) data for all time steps, replacing HL CPUE as effort with this gear type is now infrequently used. An updated LL CPUE series is used in the SE Region model, where effort is measured in the total number of hooks set rather than fisher-days for 2003 onward.

SnapEst runs on a half-yearly time step, fitting to data from each summer (October-March) and winter (April-September), from October 1983 to March 2022. Including the last summer half year (Oct 2021-Mar 2022) allowed the inclusion of the most recent DEPM spawning biomass estimates.

The Snapper model employs the slice-partition method to estimate fish population numbers by both age and length slice-within-age (McGarvey et al. 2007). Model catch was incorporated from six fishery sectors (handline, longline, hauling net to 1993, all other commercial gears combined, charter boat, and other recreational). Target type was not differentiated, and non-target catches of Snapper were, for the most part, low. The model does not attempt to discern a stock-recruitment relationship and the yearly recruitment number for each stock was freely estimated. 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 data-fitting likelihood function, are given in Appendix 9.4.

SnapEst integrates the seven input data sets to produce maximum likelihood estimates of four yearly biological indicators of fishery performance by stock or region: (i) fishable biomass (all Snapper above legal size); (ii) number of recruits; (iii) harvest fraction; and (iv) model-estimated 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 each summer model half year that commences in October) are dated in output figures by cohort year class from the summer when spawned. Harvest fraction is the yearly total catch divided by the annual fishable biomass. Egg production is computed as the estimated total number of eggs produced in each summer spawning season, assuming a 50:50 sex ratio, a fecundity-versus-length formula and all Snapper aged 2 years and older are mature. For comparison against a trigger reference point of 20%, egg production is scaled as a percentage of ‘pristine’, i.e., unexploited 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 model run was set to the average over 1982-2009, which covers years prior to the more recent stock declines. More recent years are excluded because they potentially reflect recruitment reduced by high exploitation or longer-term environmental change and so may not be typical of pre-fishing levels.

Since the 2020 Snapper assessment, SnapEst was catch-conditioned, rather than effort-conditioned. This means 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 Pope approximation was used in the catch-conditioning (Pope 1972).

For each of the two gulfs, the fishery-independent DEPM estimates of spawning biomass were fitted for those summer model half-years (October to March) during which the DEPM plankton surveys were undertaken, namely 2013/14 for SG, 2014/15 for GSV, and 2018/19, 2019/20 and 2021/22 for SG and GSV.

Catch rates for Snapper are known to be relatively uncertain as an index of abundance. Catch conditioning permits the model to use only the measure of CPUE giving most confidence as an index of abundance and only over the years that were deemed informative. In the two gulfs, CPUE was used up to the start of the DEPM, but not thereafter. 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.

In fitting to estimates of spawning biomass from the DEPM, in some years it was necessary to account for the spatial differences between model regions (i.e., SG/WCS and GSVS) and the total area covered by the DEPM surveys. In 2019 and 2021, the survey area covered about half of the SG/WCS. To extrapolate beyond the SG/WC survey area, commercial catch by MFA block was used. Model biomass for the whole region was scaled downward before fitting to the estimate of spawning biomass using the proportion of catch in the DEPM-surveyed MFA blocks divided by catch in each entire model region (Table 5-1). The half-yearly summer catches spanning the first four surveys were used. For fitting to the results from the surveys in 2019 and 2021, when the two gulfs were closed to fishing from November 2019, the 2018/19 spatial catches by MFA block (Oct-Mar) were also used to calculate the 2019 and 2021 survey spatial catch proportions. Because the GSVS was completed covered by the 2019 and 2021 DEPM surveys, this extrapolation using catch by MFA was not needed in GSV for those years.

Table 5-1. Proportion of commercial catch (Prop_DEPM) taken from the DEPM survey area (catch – DEPM) relative to the overall stock (total catch) for the Spencer Gulf / West Coast Stock (SG/WCS) and Gulf St Vincent Stock (GSVS). Note that the catch proportions for 2018 were used for the 2019 and 2021 surveys because of the fishery closures from 1 November 2019.

Stock	DEPM Year	Catch period	Total catch (kg)	Catch – DEPM (kg)	Prop_DEPM
SG/WCS	2013	Oct 2013 – Mar 2014	36,523	14,674	0.4018
	2018	Oct 2018 – Mar 2019	36,889	16,304	0.4421
	2019	Oct 2018 – Mar 2019	36,889	20,083	0.5445
	2021	Oct 2018 – Mar 2019	36,889	20,083	0.5445
GSVS	2014	Oct 2014 – Mar 2015	217,873	216,934	0.9950
	2018	Oct 2018 – Mar 2019	108,013	101,662	0.9412
	2019	Oct 2018 – Mar 2019	108,013	108,013	1.0000
	2021	Oct 2018 – Mar 2019	108,013	108,013	1.0000

Because the spatial structure of SnapEst was modified for the 2020 stock assessment, two basic body size relationships, length-at-age and weight-at-length were re-estimated at the appropriate spatial scale for incorporation into the model. Weight-at-length was found to be similar among the three stocks/regions, so a single relationship was derived and applied. Estimates of length-at-age varied among the three stocks/regions and so different growth relationships were derived for each. The von Bertalanffy formula was fitted to estimate mean length-at-age (Eq. 9.1.2), and an allometric function (McGarvey and Fowler 2002; Eq. 9.1.3) was fitted to estimate the spread (standard deviation) of lengths-at-age. The growth parameters of these formulas were estimated by maximum likelihood (Eq. 9.1.4) using a method that accounts for the truncation in observed lengths-at-age (McGarvey and Fowler 2002) that occurs because Snapper below legal minimum length are not seen in catch market sampling. Full details of these growth (length-at-age) and length-weight sub-models are provided in Appendix 9.1. This growth estimation is integrated into the overall SnapEst assessment model (McGarvey and Feenstra 2004).

SnapEst sensitivity analyses are presented in Appendix 9.5. For the SG/WCS and GSVS, they include testing different methods to compute the combination of total daily egg production and examine the impact of removing the most recent or all of the DEPM spawning biomass estimates. For SG, a sensitivity analysis was conducted for two spatial scales; (1) full DEPM survey area; (2) survey area with the northern component removed, where no snapper eggs were identified, and spawning fraction of sampled adults was (0.05). For the SE Region, sensitivity analyses tested the different timeseries of commercial CPUE as the fitted index of abundance.

5.3. Results

5.3.1. Spencer Gulf / West Coast Stock

For the SG/WCS, notable features of the SnapEst model estimates of fishable biomass are: (i) the decline through the 1980s and early 1990s to a minimum in 1993; (ii) the subsequent recovery through the mid-1990s to a peak fishable biomass in 2005; (iii) the steep and long reduction in fishable biomass to 2020, to the very low level at which the fishery was closed; and (iv) a flat to marginal increase in fishable biomass over the last two years. The reduction in fishable biomass since 2005 is consistent with very low model-estimated recruitment in conjunction with continued removal of fish over that time. 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 5-1). Such poor recruitment is consistent with the age structures for NSG, which demonstrate the significance of the 1997- and 1999-year classes as well as the lack of larger, older fish in the population of NSG after 2015 (Figure 3-2).

From 1984 to 2014, estimates of harvest fraction generally varied between 0.1 and 0.2. These increased from 2014 and reached 0.45 in 2019 (Figure 5-1). This recent peak is a consequence of continued fishing of the rapidly declining biomass. Since November 2020, a closed fishery implies zero harvest fraction. The low harvest fraction for 2020 reflects one month of fishing (October 2019 prior to fishery closure) in that summer half-yearly model time step (Figure 5-1). The trends in model-estimated egg production over time largely mirror those of fishable biomass, showing a steep and large decline from 2005 onward.

Three sensitivity analyses were run for the SG/WCS (Chapter 9.5.1). Comparing three geostatistical interpolation methods to estimate total daily egg production and fitting to the different estimates of spawning biomass produced negligible variation in the four output biological performance indicators (Figure 9-5.1). This implies that the model outputs are insensitive to the range of different spawning biomass estimates from the geostatistical methods. Next, we tested the effect of removing the most recent or all estimates of spawning biomass from the model. The SG/WCS model estimates did not achieve reasonable levels, giving much higher values of biomass when all DEPM estimates were excluded (Figure 9-5.2). However, when only the 2021 spawning biomass was omitted, the model gave similar outcomes (Figure 9-5.2). Therefore, the model could not provide realistic SG/WCS outputs without the addition of at least some DEPM spawning biomass estimates to anchor the levels of absolute abundance. But fitting to all except the 2021 DEPM is sufficient to infer the very large decline in stock biomass. Overall, consistency for a wide range of tested assumptions and model variants implies relatively strong robustness of model biological performance indicators for SG/WCS.

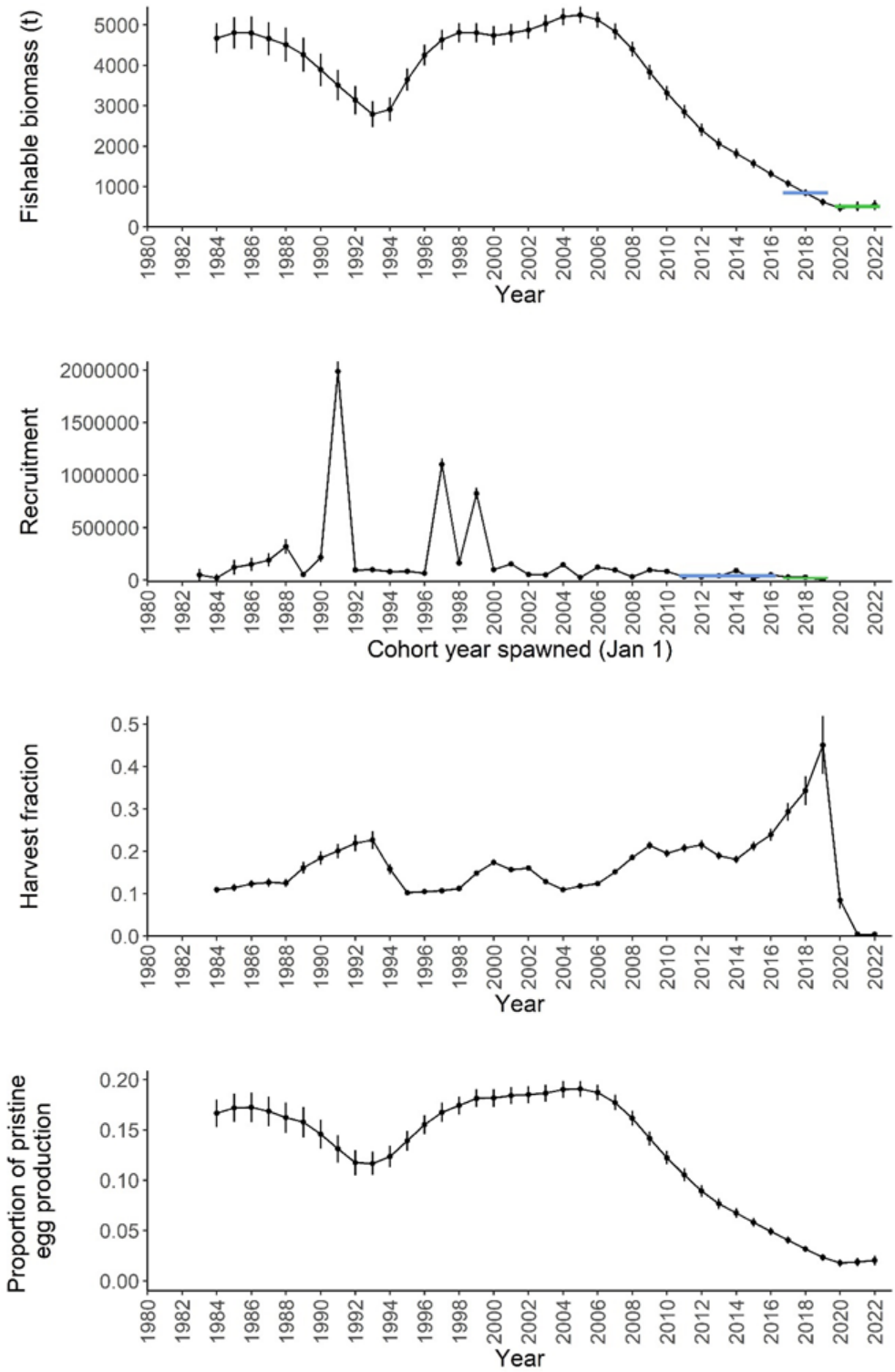

Figure 5-1. Time series of the four annual biological performance indicators from the SnapEst fishery assessment model for the Spencer Gulf/West Coast Stock. Error bars show 95% confidence intervals. For comparing these indicators with Management Plan trigger reference points given in Table 6-1, green 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.

5.3.2. Gulf St Vincent Stock

For the GSVS, estimates of fishable biomass from SnapEst declined marginally from 1984 to 1994, but then rose to a record peak in 2011 (Figure 5-2). From 2011, fishable biomass declined, particularly from 2015 onwards, dropping to an unprecedented low level in 2020. These increasing and then declining trends are consistent with the variation in fishery catches (Figure 2-4). The fits to DEPM estimates of spawning biomass from 2014, 2018 and 2019 have informed this recent decline.

The increase in fishable biomass from 1994 to 2011 reflects the numerous strong year classes that recruited to the population from 1991 to 2009, particularly those of 2001, 2004, 2007 and 2009 (Figure 5-2). These year classes were persistent in the age structures for NGSV (Figure 3-6). However, the ensuing decline in fishable biomass from 2011 onwards reflects relatively low recruitment since 2010. Model estimates of recruitment averaged 26,345 fish per year between 2011 and 2017, which was 9.7% of the average of 270,589 per year between 2001 and 2010.

From 1984 to 2010, the estimates of harvest fraction were <0.15, and lower still from 1999 to 2006 (<0.05) when most Snapper fishing effort occurred in SG (Figure 5-2). Harvest fraction increased from 2008 to a maximum of 0.74 in 2019, the last year prior to the closure. This rapid rise in exploitation reflected fishery catches and effort declining more slowly than biomass, resulting in the proportion of biomass that was removed increasing over time. The non-zero harvest fraction shown for model year 2020 is from fishing in October 2019 prior to the closure in November 2019 (Figure 5-2). The trend in egg production strongly mirrors the variation in fishable biomass over time.

Sensitivity analyses of SnapEst for the GSVS indicated that model outputs were highly robust to model assumptions and DEPM inputs (Chapter 9.5.2). As for the SG/WCS, model outcomes were not affected by different methods for estimating total daily egg production (Po * A; Figure 9-5.4). However, unlike SC/WCS, for the GSVS there remained strong agreement between the model run that excluded DEPM biomass as input and the DEPM estimates themselves. Therefore, the model estimates were almost entirely unaffected by different methods to estimate DEPM biomass, and even to whether or not they were included. This close agreement of model estimates of fishable biomass and DEPM estimates of spawning biomass, both in absolute biomass and in the trend from 2014 to 2019, provides strong mutual validation of this large biomass decline. These results add confidence in the outputs derived from both SnapEst and the DEPM.

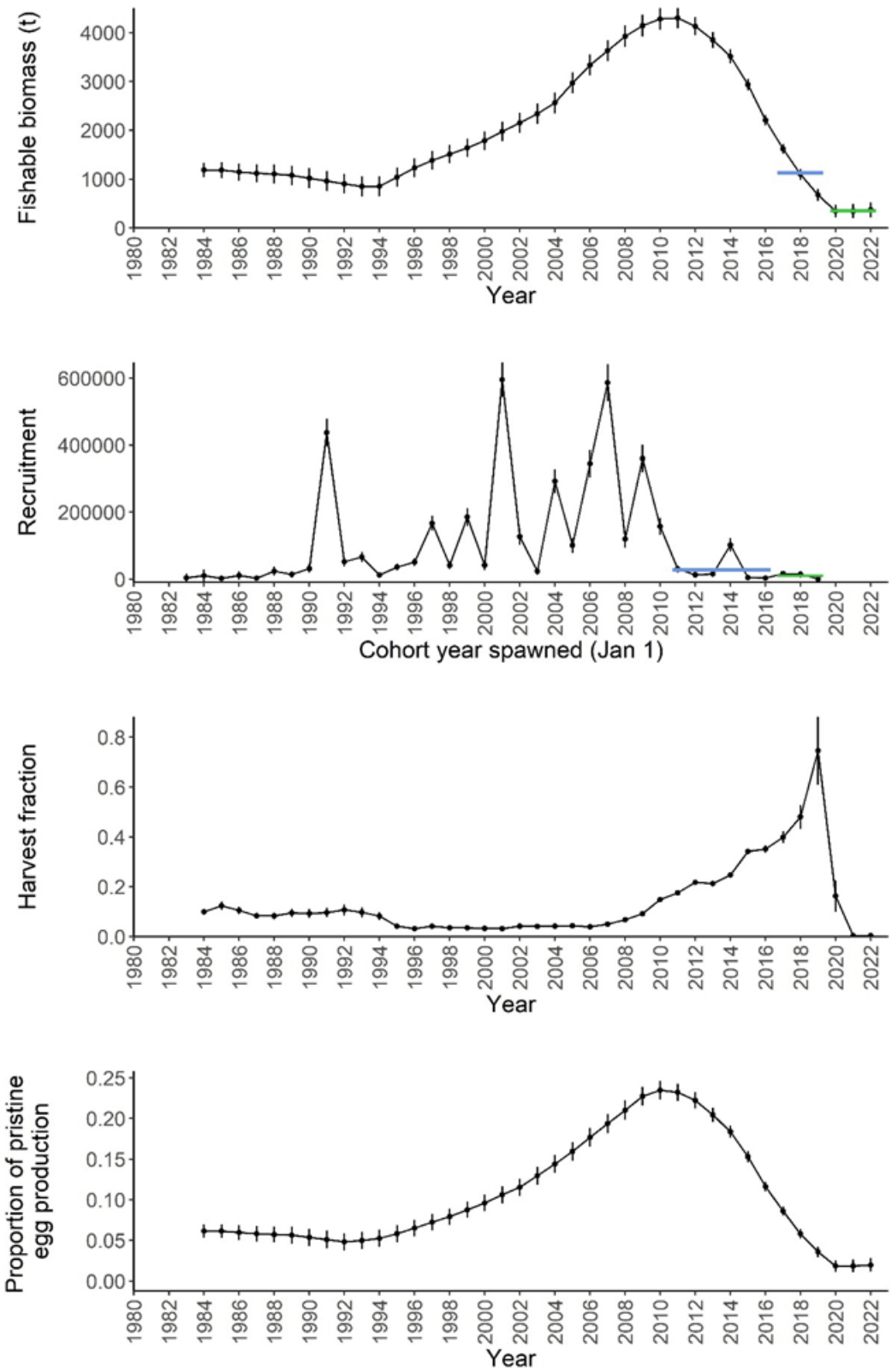

Figure 5-2. Time series of the four annual biological performance indicators from the SnapEst model for Gulf St Vincent Stock. Error bars show 95% confidence intervals. For comparing these indicators with Management Plan trigger reference points given in Table 6-1, green lines show averages over the last three years compared with blue lines giving averages over the preceding three years for biomass and preceding six years for recruitment.

5.3.3. South-East Region

From 1984 to 2004, the model estimates of fishable biomass for the SE Region were relatively low and flat. Biomass increased from 2004 to a record level in 2009 before declining again over several years (Figure 5-3). Estimated fishable biomass decreased to its lowest recent record in 2016 before increasing back to relatively high levels in recent years. The large peak in fishable biomass from 2005 to 2013 was related to the recruitment of two strong year classes in 2001 and 2004, which were persistent in the population age structures (Figure 3-10). The trend of increasing biomass in recent years appears to be driven primarily by recruitment from the strong 2013- and 2014-year classes.

The harvest fraction for Snapper in the SE Region varied around 0.20 until around 2004 before it increased through the period of 2007 to 2011, reaching a maximum of 0.56 (Figure 5-3). This related to an increase in fishing effort driven by the higher biomass in the SE Region (and decreasing biomass in SG). As for the SG/WCS and GSVS, egg production mirrors the temporal variation in fishable biomass in the SE Region (Figure 5-3).

The majority of Snapper in the SE Region are likely to have originated from Port Phillip Bay (PPB), Victoria, and emigrated to the SE Region (Fowler et al. 2017a). Inter-annual variation in recruitment success to PPB has been monitored since 1993 through an annual survey for 0+ Snapper, providing a time-series of relative abundance that is used as an index of recruitment (Hamer and Conron 2016).

This yearly index of recruitment to PPB was compared with the model estimates of SE Region recruitment numbers (Figure 5-4). We include model years of estimated recruitment only up to the 2016 cohort, as Snapper in the SE Region take ~5 years to reach legal size. It is still too soon to detect whether the highest record of Snapper recruitment in 2018 to PPB will recruit to the SE Region (Figure 5-4). There was a significant correlation ($R = 0.60$, $p = 0.002$) between the two timeseries, which provides evidence that the variation in recruitment and biomass in the SE Region is strongly correlated to annual recruitment to PPB.

The two measures of CPUE used as an index of abundance were compared, which showed large divergence among model estimates (Section 9.5.3, Figure 9-5.6). Compared to the new baseline measure of LL CPUE (kg/hook), the HL index predicts much lower biomasses in recent years, whereas the raw LL index (kg/fisher-days) predicts much higher biomasses (Figure 9-5.6). This high model uncertainty is shown by the much wider confidence intervals on the baseline estimates in recent years (Figure 5-3). This higher model uncertainty reflects the modest sizes of length and age samples from 2014 to 2019 and higher relative level of uncertainty in using CPUE as an index of stock biomass (compared to using DEPM estimates of spawning biomass).

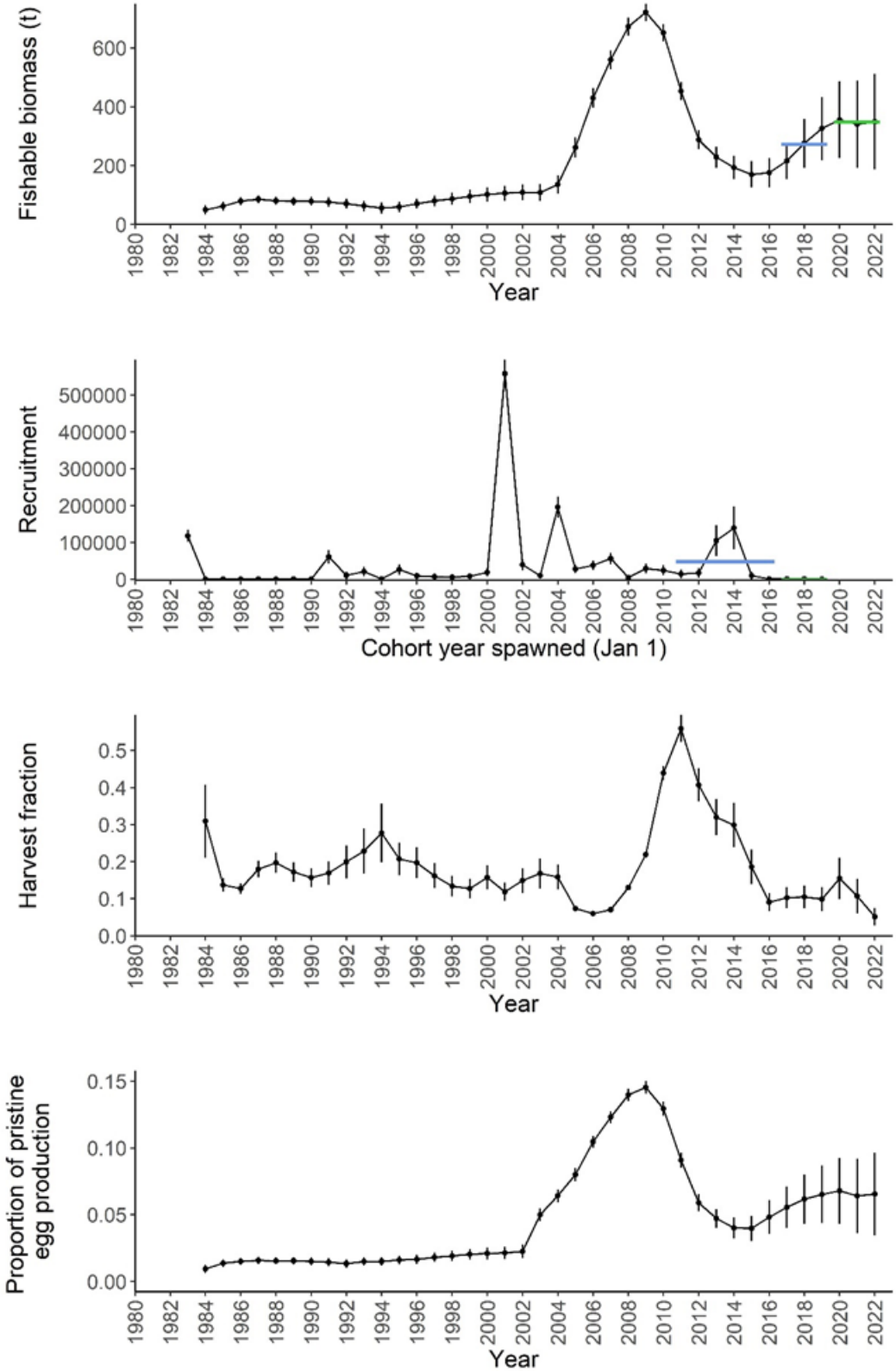

Figure 5-3. Estimates of the four annual biological performance indicators from the SnapEst fishery assessment model for the South-East Region. Error bars show 95% confidence intervals. For comparing these indicators with Management Plan trigger reference points given in Table 6-1, green lines show averages over the last three years, and blue lines show averages over the preceding three years for biomass and over the preceding six years for recruitment.

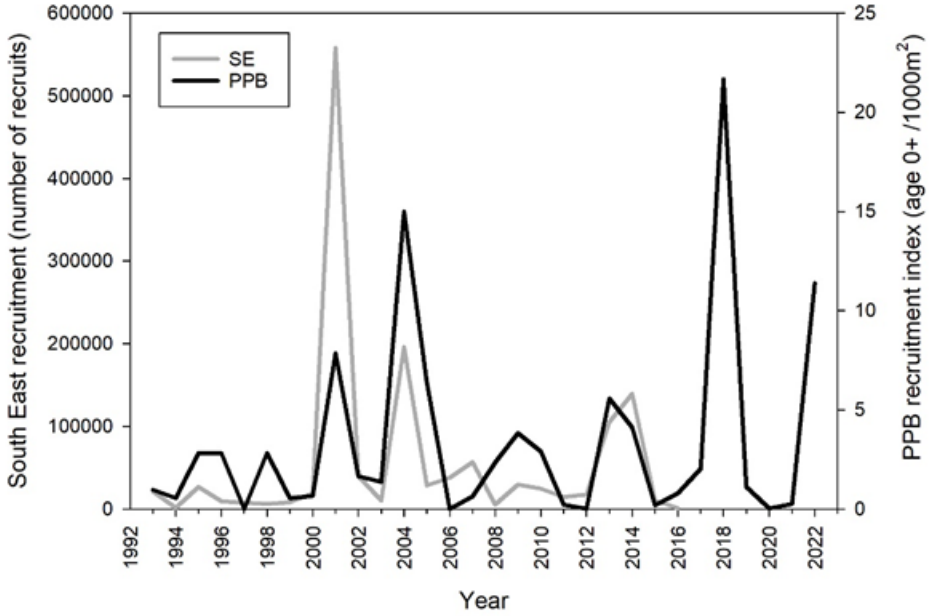

Figure 5-4. Comparison of recruitment estimates for the South-East (SE) Region between the SnapEst model (grey line) developed from the annual population age structures and the relative abundance of age 0+ Snapper sampled from Port Phillip Bay (PPB), Victoria (dark line).

5.4. Discussion

The estimates of fishable biomass from SnapEst for both the SG/WCS and GSVS inferred an end to the continued declining trends up to 2020, but showed no evidence of measurable stock recovery, or recruitment, since the fishery closure. This Discussion addresses two key aspects of the SnapEst model outputs: (1) the sensitivity of SnapEst outputs under a range of inputs or assumptions for the SG/WCS and GSVS to investigate model robustness; and (2) for the SE Region, the change in CPUE timeseries used as the index of abundance, i.e., from HL to a modified LL CPUE.

The sensitivity analyses indicate that SnapEst is generally robust to a range of model inputs and assumptions. For the GSVS, even exclusion of estimates of spawning biomass from the DEPM yielded similar model outcomes to when they were included. The close agreement between estimates of fishable biomass from SnapEst and estimates of spawning biomass from the DEPM, both in absolute biomass and in the trend from 2014 to 2019 (Figure 9-5.5) validates this large biomass decline. These results add confidence in the outputs derived from both SnapEst and the DEPM in that gulf. In SG/WCS close agreement is retained if only the last DEPM biomass is removed, but when all DEPM data inputs are removed, the model estimates diverge from plausible values (Figure 9-5.2). The SG/WCS model has identified no significant recruitment over the past 20 years, since the large stock decline. Whereas, for the GSVS, over the last 10 years some small recruitment events have occurred. Strong year classes generally remain important in these two stocks for around 20 years. This may explain the differing performances of the model by gulf when DEPM biomass inputs are (entirely) excluded. In addition, prior to the implementation of DEPM fishery-independent surveys, model estimates based on CPUE were problematic, most notably for the SG/WCS. Accordingly, estimates of spawning biomass using the DEPM have become a vital component of the stock assessment for Snapper in SA, as a fishery-independent data source and remain an essential key input into the SnapEst model.

For the SG/WCS and GSVS, a second sensitivity analysis was undertaken to compare the effect of three geostatistical interpolation methods used to estimate total daily egg production ($P_0 \times A$) in the DEPM. Fitting SnapEst to the estimates of spawning biomass using the geostatistical methods produced negligible variation in the four biological performance indicators for both stocks (Figure 9-5.1; 9-5.4). This implies that the SnapEst model outputs are insensitive to the narrow range of spawning biomass estimates generated using geostatistical methods for the surveys in 2019 and 2021 (Appendix 9-5).

Catch rates are an important input as an index of abundance for SnapEst when estimates of spawning biomass are unavailable. For the SE Region, SnapEst was previously fitted to HL

CPUE (kg.fisherday^{-1}) as the index of abundance. However, in recent years, HL gear has become effectively obsolete in the SE Region, with zero HL catch in two of the last four half-years, so there are no longer sufficient data to inform a reliable index of CPUE from HL. This necessitated a switch to an abundance index from the now-dominant gear type of LL. An improved effort measure is now employed for years since 2003 when the number of LL hooks set began to be recorded on catch logs. This LL CPUE of kg.hook^{-1} was used instead of kg.fisherday^{-1}, as it is known that kg.fisherday^{-1} for LL is subject to higher variability, particularly since the number of sets of hooks has been constrained since 2016 by the management measure of trip limit. Sensitivity of model outcomes to changes in the fitted index of abundance was tested, and showed large divergence among model estimates between CPUE indices (Figure 9-5.6). Compared to the new baseline measure of CPUE (kg.hook^{-1}), the previous HL index (kg.fisherday^{-1}) predicts much lower biomass in recent years, whereas the raw LL index (in kg.fisher-days^{-1}) predicts much higher biomasses (Figure 9-5.6). The higher model sensitivity to this index of abundance is consistent with the much wider confidence intervals of model baseline estimates found for the last years of the model time frame (error bars in Figure 5-3). This higher model uncertainty for the SE Region in recent years reflects modest numbers of length and age samples from 2014 to 2019 (though large samples were collected since 2020; Figure 3-9). It is also consistent with the relatively low confidence in CPUE as an index of Snapper stock biomass (compared to using estimates of spawning biomass from DEPM).

6. FISHERY PERFORMANCE INDICATORS

6.1. Introduction

The MSF Management Plan includes a harvest strategy for Snapper that outlines the process for monitoring the effectiveness of the management arrangements using two sets of fishery performance indicators, 'general' and 'biological' (PIRSA 2013). General fishery performance indicators are based on commercial fishery statistics (Chapter 2), and biological fishery indicators are based on the outputs from the SnapEst assessment model (Chapter 5) and population age structures (Chapter 3). The performance indicators are divided into primary and secondary, with the former identified as the key determinants of fishery performance, and the latter providing supporting information for the weight-of-evidence approach to assign stock status (Chapter 7).

The MSF Management Plans harvest strategy outlines the breakdown of allocation of the Snapper resource between the commercial, recreational, and Aboriginal/Traditional sectors, based on commercial catch data and estimates of recreational catch, aligning with the 2007/08 recreational fishing survey (Jones 2009). Commercial State-wide sector allocations were distributed between the MSF, SZRLF, NZRLF, and LCF. As part of the MSF reform in July 2021, a regional distribution of the State-wide sector allocations was applied to allocate quota units to sectors in the various fishing zones for each species (Smart et al. 2022). The performance of the SG/WCS, GSVS and SE Region were determined based on the assessment of the fishery performance indicators against reference points defined in the Management Plan (PIRSA 2013).

6.2. Methods

6.2.1. Performance Indicators

A series of general and biological fishery performance indicators and associated reference points were used to assess stock status, which primarily relate to fishery dependent data (PIRSA 2013). As a result of the fishery closures for the SG/WCS and GSVS from November 2019, fishery-dependent statistics for 2020 and 2021 were only available for the SE Region. These were considered for each of the two main gear types of HL and LL at the scale of the post-reform management zone for the SE Region.

The general performance indicators were considered in each case (PIRSA 2013). 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 2021 was compared against those calculated for the reference period of 1984 to 2021 and assessed using several trigger reference points (Table 6-1). The estimates of Prop200kgTarHL and Prop200kgTarLL for 2021 were compared against those from the reference period of 2004 to 2021, using the same trigger reference points that are used for the general performance indicators (Table 6-1).

There are five biological performance indicators: fishable biomass; model-estimated egg production; harvest fraction; recruitment; and age structures (Table 6-1). The first four are yearly time-series of output parameters from the SnapEst model (Chapter 5), whilst age composition is the catch proportions by age, derived from commercial fish market sampling and targeted adult sampling. The SnapEst model performance indicators for each of SG/WCS, GSVS and the SE Region were compared against their respective trigger reference points indicated in Table 6-1. For age composition, 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.

Table 6-1. 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 Indicator	Type	P or S	Trigger Reference Point
Total catch	G	S	3rd lowest/3rd highest
			Greatest interannual change (±)
			Greatest 3-year trend (±)
			Decrease over 5 consecutive years?
Targeted handline effort	G	P	3rd lowest/3rd highest
			Greatest interannual change (±)
			Greatest 3-year trend (±)
			Decrease over 5 consecutive years?
Targeted handline CPUE	G	P	3rd lowest/3rd highest
			Greatest interannual change (±)
			Greatest 3-year trend (±)
			Decrease over 5 consecutive years?
Targeted longline effort	G	P	3rd lowest/3rd highest
			Greatest interannual change (±)
			Greatest 3-year trend (±)
			Decrease over 5 consecutive years?
Targeted longline CPUE	G	S	3rd lowest/3rd highest
			Greatest interannual change (±)
			Greatest 3-year trend (±)
			Decrease over 5 consecutive years?
Prop200kgTarHL		P	3rd lowest/3rd highest
			Greatest interannual change (±)
			Greatest 3-year trend (±)
			Decrease over 5 consecutive years?
Prop200kgTarLL		S	3rd lowest/3rd highest
			Greatest interannual change (±)
			Greatest 3-year trend (±)
			Decrease over 5 consecutive years?
Fishable biomass	B	P	3-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

6.2.2. Allocation

The *Fisheries Management Act 2007* states that the Management Plan must specify the allocation of the resource amongst the various sectors within the fishery. Allocated shares at the state-wide level for the MSF were informed by the 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 6-2). 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). 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 6-2). The trigger limits have been set at levels that are commensurate with the 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.

Since 1 February 2020, catches in the SE Region have been managed by a Total Allowable Catch (TACC). Commercial sector allocations of the TACC between 1 February 2020 and 30 June 2021 were based on the State-wide allocations prescribed in the Management Plan (PIRSA 2013). As part of the MSF reform, a regional distribution of the State-wide sectoral allocations was applied from 1 July 2021. The regional allocations were recommended by the SMAC using the same methodology that established the original State-wide allocations in the Management Plan (PIRSA 2013) (Table 1-2). Total catch for the commercial and Charter Boat sectors was determined from daily logbooks and catch returns. Total catch for the recreational sector was estimated by multiplying the number of fish reported by the average weight of Snapper in the SE Region (i.e., 1.98 kg).

Table 6-2. Allocation of Snapper catch among the commercial fisheries as prescribed in the MSF Management Plan (PIRSA 2013).

	MSF	SZRLF	NZRLF	LCF
Commercial allocation	97.50	1.78	0.68	0.04
Trigger 2 (%)	n/a	2.68	1.30	0.75
Trigger 3 (%)	n/a	3.58	2.00	1.00

6.3. Results

6.3.1. Fishery Performance Indicators

Across the SG/WCS, GSVS and the SE Region, there was one trigger reference point breached for the general fishery performance indicators and eighteen breaches for the biological performance indicators (Table 6-3).

For SG/WCS there were four negative and two positive breaches of the biological performance indicators based on the outputs of SnapEst. The negative breaches were average fishable biomass was 40% lower for the past three years compared to the average of the previous three years. Model-estimated egg production was estimated to be at 2% of level expected of a pristine biomass, which is well below the <20% threshold. Average recruitment for the past three years is 81% lower than the historical mean, and 28% lower than the previous six-year average. The two positive breaches in biological performance indicators were harvest fraction was 0%, as this stock is closed to fishing, and the percent of fish older than 10 years for the combined SG/WCS was 30% which is above the reference point of 20%.

For the GSVS, there were four negative and two positive breaches of the biological performance indicators derived from the SnapEst outputs. The negative breaches were average fishable biomass for the past three years was estimated as 69% lower than the previous three-year average. Model-estimated egg production was estimated to be at 2% of the level expected of a pristine biomass, which is well below the <20% threshold. Average recruitment for the past three years is 90% lower than the historical mean, and 78% lower than the previous six-year average. The two positive breaches were harvest fraction was 0%, as this stock is closed to fishing, and the percent of fish older than 10 years was 35%, which is above the reference point of 20%.

For the SE Region, there was one positive breach in the general performance indicators and all six of the biological performance indicators. Handline fishery statistics were confidential for 2021 as there were less than five fishers targeting Snapper with this gear type. Targeted longline CPUE, which is a secondary general performance indicator, was the highest on record at 101.7 kg.fisher-day^{-1}. For the biological performance indicators, the four negative breaches were model-estimated egg production was <20% of pristine population at 6.5% of the level expected for an unfished stock. Average recruitment of the last three years was 98% lower than for the previous six years, and 97% below the historical mean and 18.5% of fish in the age structure were >10 years old, which was below the reference point of 20%. The two positive breaches in biological performance indicators were fishable biomass was, on average, 27% higher in the last three years compared with the previous three-year average and harvest fraction was low at 11%, which is below the 32% trigger reference point.

Table 6-3. 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, N = no breach.

Performance Indicator	Type	Trigger Reference Point	SG/WC	GSV	SE
Total catch	G	3rd lowest/3rd highest	NA	NA	N
		Greatest interannual change (±)	NA	NA	N
		Greatest 5-year trend (±)	NA	NA	N
		Decrease over 5 consecutive years?	NA	NA	N
Targeted handline effort	G	3rd lowest/3rd highest	NA	NA	CONF
		Greatest interannual change (±)	NA	NA	CONF
		Greatest 5-year trend (±)	NA	NA	CONF
		Decrease over 5 consecutive years?	NA	NA	CONF
Targeted longline effort	G	3rd lowest/3rd highest	NA	NA	N
		Greatest interannual change (±)	NA	NA	N
		Greatest 5-year trend (±)	NA	NA	N
		Decrease over 5 consecutive years?	NA	NA	N
Targeted handline CPUE	G	3rd lowest/3rd highest	NA	NA	CONF
		Greatest interannual change (±)	NA	NA	CONF
		Greatest 5-year trend (±)	NA	NA	CONF
		Decrease over 5 consecutive years?	NA	NA	CONF
Targeted longline CPUE	G	3rd lowest/3rd highest	NA	NA	Highest
		Greatest interannual change (±)	NA	NA	N
		Greatest 5-year trend (±)	NA	NA	N
		Decrease over 5 consecutive years?	NA	NA	N
Prop200kgTarHL		3rd lowest/3rd highest	NA	NA	CONF
		Greatest interannual change (±)	NA	NA	CONF
		Greatest 5-year trend (±)	NA	NA	CONF
		Decrease over 5 consecutive years?	NA	NA	CONF
Prop200kgTarLL		3rd lowest/3rd highest	NA	NA	N
		Greatest interannual change (±)	NA	NA	N
		Greatest 5-year trend (±)	NA	NA	N
		Decrease over 5 consecutive years?	NA	NA	N
Fishable biomass	B	3-yr ave is +/- 10% of previous 3-yr ave	-40%	-69%	27%
Harvest fraction	B	above 32% (int. standard)	0%	0%	11%
Egg production	B	<20% of pristine population	2%	2%	6.5%
Recruitment	B	3-yr ave is +/- 10% of historical mean	-81%	-90%	-97%
	B	3-yr ave is +/- 10% of previous 6-yr ave	-28%	-78%	-98%
Age composition	B	Prop >10yrs <20% of fished population	30%	35%	18.50%

6.3.2. Allocation

Sectoral catches varied over the three fishing seasons. For the 2020 fishing season, the commercial sector caught 95% of their allocation, Charter Boat caught 4.7%, and the recreational sector caught 14.3% (Table 6-4). For the 2021 fishing season, i.e., from 1 February 2021 to 30 June 2021, the commercial sector exceeded their allocation by 5.1%, Charter Boat caught 13.9% of their allocation, and the recreational sector exceeded their allocation by 20% (Table 6-4). Data for the 2021/22 fishing season are not finalised. Up to 31 December 2021, the commercial sector had caught a quarter of their allocation, Charter Boat had caught 8%, and the recreational sector had caught 16% (Table 6-4).

Table 6-4. Percentage of reported catch compared to total allowable catch (TAC) for the commercial, Charter Boat, and recreational sectors for the SE Region from 1 February 2020 and 31 December 2021. *Includes data up to 31 December 2021 and not the full TAC period. Values are percent (%).

Time period	TAC (kg)	Total catch %	Commercial %	Charter %	Recreational %
1 Feb 2020 – 31 Oct 2020	75,000	78.8	95.3	4.7	14.3
1 Feb 2021 – 30 June 2021	26,667	96.1	105.1	13.9	119.7
*1 July 2021 – 30 June 2022**	*48,000*	*21.3*	*24.7*	*7.7*	*16.4*

For the assessment of the catches of the different commercial fisheries in 2021 against their allocations, their percentage contributions to annual total catch were compared using Triggers 2 and Trigger 3 reference points (Table 6-5). For 2021, one trigger reference point was activated by the SZRLF exceeding primary trigger limit 3 according to the Management Plan (PIRSA 2013). However, during the MSF Reform, an Independent Allocation Advisory Panel (IAAP) recommended that regional allocations of quota units among sectors for each Tier 1 stock. As a result, for the SE Region 20.8% of commercial Snapper quota units were allocated to the SZRLF (Smart et al. 2022), and this share of the commercial quota was not exceeded in 2020 and 2021.

Table 6-5. Snapper Commercial Fishery Allocation from 2015 to 2021 according to the Management Plan 2013 including reference trigger limits for each commercial fishery. Values are percent (%).

	MSF	SZRLF	NZRLF	LCF
Allocation	*97.50*	*1.78*	*0.68*	*0.04*
Trigger 2 (%)	*n/a*	*2.68*	*1.30*	*0.75*
Trigger 3 (%)	*n/a*	*3.58*	*2.00*	*1.00*
% 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
% total 2020	82.21	17.78	0	0
% total 2021	80.67	19.33	0	0

6.4. Discussion

There have been major changes to the dynamics of the Snapper fishery in SA since the previous assessment in 2020 (Fowler et al. 2020). As a result of the fishery closures for the SG/WCS and GSVS, there were no fishery statistics available to evaluate the general performance indicators for 2020 and 2021. For the SE Region, the general performance indicator for targeted LL CPUE was the highest on record and was the only performance indicator to exceed its trigger reference point. This increase in CPUE is likely to reflect an increase in the relative abundance of Snapper in this region and/or an increase in fishing efficiency. However, targeted LL CPUE by fisher-day is a coarse metric and does not account for differences in fishing activity on that day, i.e., the number of hooks or lines set. Consequently, an alternative LL CPUE metric of fish weight by hook was considered in this assessment, which accounts for variation in fisher behaviour. If this measure of CPUE is to be continued in future assessments, a relative general performance indicator should be revised to assess and measure its trends. The trends in HL catch were not reported as the data were confidential (i.e., <5 fishers). The use of HLs in the SE Region has become almost obsolete as the fishery has largely transitioned to LL.

There have also been changes to the sectoral share of TACC in the SE Region following the MSF reform (Smart et al. 2022). Since 1 February 2020, the fishery has been managed by a TAC that was divided among sectors based on their prescribed allocations in the Management Plan (PIRSA 2013). For the 2020 and 2021 fishing seasons (i.e., up to 30 June 2021), the TAC was divided based on State-wide sectoral allocations. During this time, commercial fishers with MSF endorsements could target Snapper until the TAC was fulfilled, which resulted in a significant increase in catch and effort in 2020 by both MSF and SZRLF fishers. The increase in catch and effort by SZRLF fishers resulted in this commercial sector exceeding its State-wide allocation. Following the implementation of the MSF reform on 1 July 2021, access to the commercial fishery has been further limited through the introduction of regional individual transferable quota (ITQ) units for all Tier 1 species, including Snapper in the South-East. The initial allocation of ITQ units for Snapper in the SE Region meant that the allocations between commercial sectors no longer conformed to the State-wide allocations prescribed in the Management Plan (PIRSA 2013). Furthermore, the proportional allocations for the commercial sector will continue to change due to the transfer of ITQ units between license holders.

7. GENERAL DISCUSSION AND STOCK STATUS

7.1. Context of this Assessment

State-wide commercial catches of Snapper fluctuated during the late 1990s and 2000s, reached a peak of 1,035 t in 2010, and then substantially declined to 252 t in 2019. The SG/WCS provided the highest annual catches up to 2009, after which they declined 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 exponential increase between 2007 and 2010. However, catches from the GSVS then declined considerably from 2016 to 2019. The catches from the SE Region also increased rapidly between 2007 and 2010, before they declined to much lower levels in 2016. Catches have since increased but remain at comparatively low levels.

For both the SG/WCS and the GSVS, the trends in fishery statistics identified ongoing reductions in the fishable biomass of Snapper over numerous years. In each case, the declines in fishable biomass reflected prolonged periods of poor recruitment coupled with, prior to the closure, continued exploitation of a depleting stock. The declining trends in fishery performance combined with the lack of stock recovery despite a suite of management interventions resulted in the SG/WCS being classified as depleted in 2018 (Fowler et al. 2019), followed by the GSVS in 2020 (Fowler et al. 2020). In response to the low stock levels and depleted stock status classifications, fishery closures were implemented from 1 November 2019 for the SG/WCS and GSVS.

7.2. Stock Status

7.2.1. Spencer Gulf / West Coast Stock

No fishery statistics were available for the SG/WCS from November 2019. Historic trends have shown substantial declines in most fishery statistics from the mid-2000s. 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. These patterns indicated a rapid decline and persistent low biomass levels.

Age structures sampled in 2019, 2020 and 2021 show the population in NSG is dominated by small, young fish up to seven years of age, and a low proportion of older fish. These contemporary age structures contrast with those from the 1990s and 2000s that included many fish of >20 years of age and some >30 years old (McGlennon et al. 2000, Fowler et al. 2010, 2016a). Recent age structures indicate the presence of 2014 and 2016-year classes. The age structures for SSG contained a broader range of year classes but were still predominantly

composed of fish up to seven years of age. These data demonstrate that the age composition of the SG/WCS remains truncated and that recent recruitment has been comparatively low.

Applications of the DEPM in NSG in 2013, 2018, 2019 and 2021 indicated a continued decline in spawning biomass over this period. The estimate of spawning biomass in 2021 was 108 t (± SE; 65); which was a 39% reduction from the estimate in 2019 (177 t ± SE; 34). The reduction in spawning biomass between surveys largely resulted from a 49.5% reduction in spawning area and an 18% increase in spawning fraction. The results from four applications of the DEPM since 2013 support the continued low level of spawning biomass in NSG.

The SnapEst model estimates of fishable biomass declined by 90% from a peak of 5,244 t (± SE; 104) in 2005 to 543 t (± SE; 65) in 2022, which is the third lowest estimated biomass. Fishable biomass from SnapEst has remained largely unchanged since the lowest estimate of 469 t (± SE; 53) in 2020. Model outputs indicate that the decline in fishable biomass relates to a prolonged period of poor recruitment throughout the 2000s and from 2010, and increasing harvest fractions, caused by the continued fishing of a depleting stock prior to the closure. The model outputs show that egg production in 2022 was estimated at 2% of that expected for an unfished stock and that average recruitment over the last three years was estimated at 28% lower than the previous six years, and 81% lower than the historical mean. Consistent with low recent biomass, extended trends in poor recruitment and low levels of egg production, the four reference points for the biological performance indicators were negatively triggered.

Several independent datasets demonstrate that the fishable biomass and recruitment for the SG/WCS indicate no signs of measurable improvements and have continued to persist at historically low levels. These include: (i) truncated age structures – the very low proportion of large, old fish in the population; (ii) continued lack of recruitment of any new strong year classes; and (iii) continuing declines in spawning biomass. Integration of all data in SnapEst confirms this. The model-estimated decline in biomass of the SG/WCS has occurred since the mid-2000s and has been apparent at the regional and biological stock levels since 2013 (Fowler et al. 2013, 2016a, 2019, 2020). The primary causes of the decline are 23 consecutive years of poor recruitment since 1999, evident in the lack of strong year classes in annual age structures throughout the 2000s and into the 2010s (Fowler et al. 2016a, 2019), coupled with ongoing exploitation of a depleting stock prior to the fishery closure.

The SG/WCS has been classified as ‘depleted’ since 2018. It is evident that the biomass and recruitment of the SG/WCS remains at low levels with no evidence of measurable stock recovery following the closure of the fishery. Biomass is depleted, recruitment is impaired and the SG/WCS remains classified as ‘**depleted**’.

7.2.2. Gulf St Vincent Stock

Fishery statistics were not available for the GSVS in 2020 and 2021 following the closure of the fishery from November 2019. Trends in commercial fishery statistics for the GSVS, particularly for the LL sector, increased between 2007 and 2010 to its highest levels, which were maintained until 2015. Thereafter, declines were observed 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.

Age structures developed for 2020, 2021 and 2022 were broad for both the NGSV and SGSV. A small number of fish from the previous strong year classes of 2007 and 2009 remained in the recent age structures in NGSV. A 2014-year class has emerged and persisted and this age class dominated the recent age structures for SGSV. Nevertheless, recruitment over recent years has been comparatively low.

Fishery-independent estimates of spawning biomass from the DEPM show declines in spawning biomass, from 2,780 (± SE; 1,444) in 2014 to 404 t (± SE; 124) in 2021. There was a 50% decline in estimated spawning biomass between the 2019 and 2021 surveys, which was directly related to a 57% reduction in spawning area.

Modelled fishable biomass from SnapEst peaked at 4,300 t (± SE; 104) in 2011, before declining by 92% to 343 t (± SE; 67) in 2020, which was the lowest on record. Fishable biomass has since remained largely unchanged. The increase in biomass through the 2000s reflected the recruitment of numerous strong year classes to the population during the 1990s and 2000s. The subsequent reduction in biomass related to relatively poor recruitment from 2009 to 2019, coupled with unprecedented catches. Egg production in 2022 was estimated at 2% of that expected for an unfished stock. Average recruitment over the last three years was estimated at 78% lower than for the previous six years and 90% lower than the historical level. Consistent with low recent biomass, poor recent recruitment, and low egg production, the four reference points for the biological performance indicators were negatively triggered.

In 2020, the status of the GSVS was changed from ‘depleting’ to ‘depleted’ (Fowler et al. 2020). This reflected the decline in estimated spawning biomass from DEPM since 2014, poor recruitment since 2009, persistent high targeted catch and effort until 2019, and decreasing fishable biomass from SnapEst. Multiple lines of evidence demonstrate that management has not yet resulted in measurable improvements, and the stock has continued to persist at low levels. These are: (i) poor recruitment between 2010 and 2019, despite the appearance of the 2014-year class; (ii) continued low estimates of spawning biomass; and (iii) continued low SnapEst estimated fishable biomass and egg production. Biomass is depleted, recruitment is impaired and the GSVS remains classified as ‘**depleted**’.

7.2.3. South-East Region

The Snapper population in the SE Region of SA is the western extremity of the cross-jurisdictional Western Victorian Stock (WVS). This population is primarily sustained through the emigration of fish from the main nursery area, which is located in Port Phillip Bay (PPB), Victoria (Fowler 2016, Fowler et al. 2017a).

Substantial increases in annual fishery catches, effort, and catch rates occurred between 2008 and 2012, which then declined through to 2015 and remained at low levels to 2019. Longline catch and effort moderately increased in 2020 and then moderated in 2021, consistent with changes in total allowable catches between fishing seasons. As a result of recent increases in catch and effort, targeted LL CPUE (kg.fisher-day^{-1}) reached its highest level in 2021, triggering the general performance indicator.

Age structures in 2020, 2021 and 2022 were dominated by the 2013- and 2014-year classes and there were comparatively few fish remaining from the above average 2009- and 2010-year classes. The age structures for the SE Region continue to demonstrate strong correlation with the timeseries of 0+ recruitment in PPB. As such, it is expected that the strong 2018-year class from PPB will recruit to the fishable biomass of the SE Region in the near future.

Outputs from the SnapEst model indicate a substantial increase in fishable biomass between 2005 and 2008 following recruitment of two strong year classes to PPB in 2001 and 2004, and the subsequent emigration of Snapper from PPB to the SE Region. Fishable biomass then decreased until 2015 as a result of exploitation and low recruitment since 2004. Transitioning from HL to LL CPUE (kg/hooks) in SnapEst has resulted in a doubling of model-estimated fishable biomass compared to the previous assessment. Model-estimated fishable biomass has steadily increased from 176 t (± SE; 45) in 2016 to 349 t (± SE; 70) in 2022, which reflects recruitment of the 2013- and 2014-year classes to the fishery. All six biological performance indicators were triggered, four negative (trends in recruitment, egg production, age composition) and two positive (harvest fraction and trends in fishable biomass).

While the TACs for the 2021 and 2021/22 fishing seasons were set based on the estimated fishable biomass for the SE Region (which is largely influenced by CPUE), there are other considerations 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 is dependent on recruitment success within PPB. Fish from this area move to the SE Region of SA, but relatively few return (Fowler et al. 2017). Secondly, recent strong recruitment to PPB in 2013, 2014, 2018 and 2022 suggests future replenishment of the SE Region population.

In 2016 (Hamer and Conron 2016), 2018 (Stewardson et al. 2018) and 2021 (Piddocke et al. 2021), the WVS was classified as '**sustainable**'. The annual 0+ recruitment survey showed that over the 30 years to 2022, there had been eight years for which recruitment was above the long-term average. Furthermore, the 2018-year class was the largest yet recorded and the 2022- year class the third highest on record (Table 5-4). These lines of evidence suggest that the adult biomass is at a level sufficient to ensure that 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.

7.3. Assessment Uncertainties

There was a high level of consistency from the differing datasets available to infer stock status in this assessment. However, there remain several sources of uncertainty associated with the assessment that require consideration. The estimates of spawning area for SG and GSV in 2021, used in DEPM spawning biomass calculations, were considerably lower than in 2019. This resulted in a proportional decrease in estimates of spawning biomass. The reduced spawning area observed in 2021 may reflect the natural aggregating behaviour of Snapper throughout the spawning period (Wakefield 2006, 2010, Saunders 2009), and the concurrent lack of disturbance in the two years prior to the surveys due to the fishery closures. As a result, the reduction in spawning area may relate to the shift in the spatial distribution and abundance of adults, rather than the decrease in spawning biomass.

This hypothesis is supported by the localised distribution of moderate to high egg densities in SG and GSV that were strongly correlated with the distribution of spawning adults sampled simultaneously (Figure 4-10). Furthermore, for SG, the same number of Snapper eggs were sampled in 2021 as 2019, albeit from fewer stations. This suggests that the current method used to estimate total daily egg production, where P_0 and A are considered independent parameters, had difficulty accounting for the aggregating behaviour of spawning fish. To account for this, several geostatistical approaches were investigated to reduce the cumulative error associated with the interaction of numerous parameters and account for the correlation between P_0 and A. Estimates of total daily egg production obtained from these geostatistical methods were used to estimate spawning biomass (Appendix 9.3.3) and incorporated into SnapEst to estimate total fishable biomass (Appendix 9.5). However, the geostatistical methods were not adopted in this assessment because they are likely to require simulation testing and refinement to validate their suitability.

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. Therefore, DEPM spawning biomass estimates were required for SnapEst to generate outputs which aligned 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 9-5.2), 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 the model inputs (Figure 9-5.5). Furthermore, as current CPUE timeseries are no longer available for the SG/WCS and GSVS SnapEst is primarily reliant on contemporary estimates of spawning biomass as an estimate of absolute abundance.

The reduction in targeted HL effort of Snapper in the SE Region has necessitated a shift to an alternate CPUE of LL, in kg per hook, as a more informative index of relative abundance for SnapEst. The switch to an alternate CPUE timeseries has seen a near doubling of the model-estimated fishable biomass that was previously estimated from HL CPUE (Figure 9-5.6). Consistent with all SnapEst outputs, large errors around the biomass estimates are expected if there are uncertainties and errors associated with the input data sources. The large error estimates around the last few years of the SE Region fishable biomass outputs are the result of the lack of historic population structure information.

7.4. Conclusions

Since the previous assessment (Fowler et al. 2020), there have been negligible changes in the population structures and estimates of fishable biomass of Snapper for the SG/WCS and GSVS, despite the closure of the two stocks to fishing since 1 November 2019. This is not unexpected given the life history of the species. Snapper is a long-lived, slow-growing species, and its population dynamics and fishery productivity are fundamentally driven by highly variable inter-annual recruitment, i.e., the numbers of 0+ juveniles that enter the population each year (McGlennon et al. 2000, Fowler and Jennings 2003, Saunders 2009, Fowler et al. 2010). The occasional strong year class sustains the population through periods of time that are characterised by poor to average recruitment (McGlennon et al. 2000, Fowler et al. 2017a). As such, the differences in fishable biomass and fishery productivity for the SG/WCS and GSVS over the past 30 years reflect the same demographic processes, that occurred independently of each other, but at different times. That is, the increase in fishable biomass, and subsequent fishery productivity, is a consequence of the recruitment of multiple strong year classes to the population in preceding years. Thereafter, the decline in fishable biomass was associated with a prolonged period of poor recruitment, concurrent with the continued exploitation of the population and increasing exploitation rates prior to the fishery closures

(Fowler 2016, Fowler et al. 2017a). These periods of poor recruitment have now extended for more than one decade for GSV and two decades for SG. Furthermore, the prolonged periods of high exploitation rates, which often exceeded reference trigger limits that were likely too high for the productivity of these stocks.

For SG, the high fishable biomass and fishery productivity during the 2000s resulted from several exceptionally strong year classes in the 1990s (specifically 1991, 1997, and 1999). The subsequent decline in fishable biomass since 2008, to record low levels in recent years, reflected a prolonged period of poor recruitment since 1999 (Figure 5-1) (Fowler et al. 2017a). Moreover, there have been just five strong year classes in SG since 1970 (i.e., 1973, 1979, 1991, 1997, and 1999) (McGlennon et al. 2000, Fowler et al. 2017a). For GSV, the increase in fishable biomass during the 2000s to a peak in 2011 reflected the accumulation of fish from multiple moderate to strong year classes during the 1990s and 2000s (specifically 1991, 1997, 1999, 2001, 2004, 2006, 2007, and 2009) (Figure 5-2) (Fowler et al. 2017a). Similarly, the severe decline in fishable biomass from 2011 to 2020 resulted from a decade of poor recruitment since 2009 and the continued exploitation of the population.

The recovery of the SG/WCS and GSVS is dependent on the recruitment of juveniles to each population and the subsequent accumulation of biomass. There is no evidence from the regional population age structures to suggest that a strong year class has recruited to either stock in recent years (i.e., since 1999 for NSG and 2009 for NGSV). Furthermore, because the two populations are heavily depleted and egg production is at a historically low level, the probability of a strong recruitment event is, accordingly, very low. As any recruitment events in the past few years and into the near future will likely be comparatively low compared to the strong year classes of the 1990s (SG) and 2000s (GSV), it is expected that multiple recruitment events will be required to replenish the two stocks before there is a substantial increase in fishable biomass. The expected timeframes for recovery are unknown, but may take several years or possibly much longer. As the Snapper biomass is likely to be low for a long period with prolonged low and/or no catches of this species, the resultant impact from redirection of targeted effort to other MSF species will need careful monitoring.

7.5. Directions for Future Research

There are nine priorities for monitoring and assessing the status of Snapper in SA. These are: (i) continuation of the adult sampling program to access biological data and monitor regional age structures; (ii) to better understand and monitor inter-annual recruitment variability of 0+ juveniles; (iii) continuation and refinement of DEPM surveys to estimate spawning biomass; (iv) developing forecasting capability in the SnapEst model; (v) improve the understanding of the Snapper population on the WC; (vi) develop a new harvest strategy (including

Management Strategy Evaluation (MSE) for Snapper and stock recovery/re-opening strategy; (vii) to improve the information on recreational catches; (viii) to understand post-release mortality; and (ix) continue stock enhancement.

For Snapper in SA, recruitment variability is assessed through the interpretation of a time series of annual population age structures (Chapter 3), which are a key input into the SnapEst model. Recruitment is retrospectively inferred from the relative strength of individual year classes in the population, and therefore there is a delay of 4–5 years from when the fish undergoes settlement as an age 0+ juvenile to when fish approach or exceed the MLL (38 cm TL) and enter the fishable biomass. Age structure information is integrated with trends in overall stock abundance to infer recruitment variation over longer time frames. As a result of the cessation of fishing for the SG/WCS and GSVS, and the consequent lack of fishery-dependent data, the development of age structures for these stocks has heavily relied on the targeted sampling of Snapper by contracted commercial MSF fishers. The fish are processed for biological information by SARDI researchers, filleted by accredited fish processors, and the fillets donated to support community needs through Foodbank. The continuation of the adult sampling program is fundamental to develop annual length and age structures for the six regional populations and to detect recruitment to the fishable biomass.

There is a need to monitor the relative abundance of age 0+ Snapper to provide an early indication of recruitment for the SG/WCS and GSVS. This approach is used in PPB, Victoria, to predict future trends in population demographics and fishable biomass for the WVS and has demonstrated a strong correlation with the population age structures over the past 25 years (Figure 5-4) (Hamer and Jenkins 2004, Hamer and Conron 2016). Furthermore, the development of a fishery-independent recruitment index was identified as the highest research priority at the most recent National Snapper Workshop (Cartwright et al. 2021). There is a current research project addressing this need (FRDC 2019-046), which is focussed on recruitment variability of Snapper in SA and has two primary objectives: (i) to develop a sampling method and protocol for monitoring the inter-annual variation in recruitment of 0+ Snapper; and (ii) enhance our understanding of the variability in recruitment of Snapper in SA. This project is expected to be completed in December 2023. The sampling protocol developed through the project will need to be continued to develop an ongoing annual timeseries of relative recruitment abundance for both SG/WC and GSV stocks.

The estimates of spawning biomass using the DEPM have become a vital component of the stock assessment for Snapper in SA, as a fishery-independent data source and a key input into the SnapEst model. Furthermore, the estimates of spawning biomass become essential in the absence of fishery-dependent data (i.e., CPUE), as SnapEst requires an index of

abundance to reliably produce the four biological performance indicators. There have been several areas for method development identified through this assessment to refine the estimates of spawning biomass and reduce uncertainty associated with individual parameters. These include: (i) the application of geostatistical methods to determine total daily egg production; (ii) evaluating the effectiveness of different sampling methodologies to collect Snapper eggs; and (iii) understand and account for the aggregating behaviour of Snapper in the estimate of spawning biomass. These areas for method development could be investigated through a targeted study, particularly by incorporating a hydroacoustic survey to better understand the aggregating behaviour of Snapper during the spawning period.

There is also a need to develop forecasting capability in the SnapEst model to predict how the fishable biomass of the SG/WCS and GSVS will respond under various recruitment scenarios. Such projections would provide an indication of the expected recovery time for the two stocks under different scenarios and assist the development of appropriate management and harvest strategies. Furthermore, the predictive model would be heavily dependent on a reliable index of recruitment (i.e., relative abundance of 0+ juveniles) to inform changes in the population age structures and subsequent fishable biomass. A similar model was developed for Southern Rock Lobster (*Jasus edwardsii*) in SA and is used to predict changes in fishable biomass and inform harvest strategy development (McGarvey et al. 2014, 2016). Developing forecasting ability for Snapper would require a new modelling framework and numerous additional components, which could be addressed through a targeted research project. The time frame for incorporating the 0+ index of recruitment into future forecasting extends forward about 10 years because it takes about 4-5 years for 0+ Snapper to reach fishable size, and then sufficient years of overlap thereafter are needed to derive the forecasting relationship that uses the 0+ index to produce future simulations of yearly recruit numbers.

The Snapper population on the WC of Eyre Peninsula is a regional component of the SG/WCS (Fowler 2016, Fowler et al. 2017a). Episodically, in years of exceptionally strong recruitment in NSG (e.g., 1991, 1997, and 1999), the WC population is replenished through the density dependent emigration of fish of a few years of age as they disperse from NSG and through SSG (Fowler et al. 2017a). This hypothesis of stock structure is supported by the similarity of regional age structures between NSG and the WC during the 2000s (Fowler unpublished) and the otolith chemistry of Snapper from the exceptionally strong year classes in NSG (Fowler et al. 2005, 2017a). As a result of the prolonged period of poor recruitment in NSG since 1999 and the subsequent depletion of the population in SG, it is unlikely that this density dependent movement has occurred for a number of years or will occur until the SG population is replenished. Regional age structures for the WC developed in 2020 and 2021 showed that a very small number of Snapper from the strong 1997- and 1999-year classes in NSG remained

in the WC population, but there were several other year classes in the age structures for the WC that were not apparent in NSG (Figure 3-13). This information, coupled with the presence of spawning females and small, young fish (age 1+), suggests there is some local recruitment and replenishment of the WC population. Consequently, there is a need to understand the relative contributions of local population processes and emigration from NSG to the WC population. This is particularly important following the regionalisation of the fishery through the MSF reform.

The current MSF Management Plan, which included a harvest strategy for Snapper, was developed in 2013 and was scheduled for review in 2018. The review of the Management Plan was postponed while the MSF underwent a large reform process (Smart et al. 2022). Since the implementation of the MSF reform (July 2021), there is a need to develop a new harvest strategy which will integrate updated performance measures and control rules relevant to the contemporary population dynamics information and recent changes in fishery management. Incorporating MSE capabilities will enable scientists and managers to test outcome probabilities, by simulation, of various harvest decisions or management procedures. Development of the harvest strategy should also include a stock rebuilding / recovery strategy with clear timelines and targets aligned with performance measures and integrated for the SG/WCS and GSVS.

Infrequent and imprecise estimation of recreational catch is a primary limitation on assessment of all Marine Scalefish species. Recreational surveys occur infrequently and commonly result in estimates with large standard errors, particularly in regional areas where sample sizes are typically low. Between survey years, recreational catches are estimated by extrapolation using SA population statistics, which provides no yearly information about the intervening change over time. Estimates of recreational catch of Snapper in 2020 and 2021 were available for the SE Region as a result of a TARC and mandatory catch reporting, which provided highly resolved information for this assessment. Updated catch and effort data from the current recreational survey (due for completion in 2022; FRDC 2020-056) and the continuation of catch reporting (and potential extension across the State when re-opened) will improve the accuracy of future assessments. There is also a need to determine the sensitivity of the SnapEst model to the uncertainty of recreational estimates.

Snapper is highly susceptible to barotrauma through fishing activities, which results from the rapid change in pressure when the fish is brought from the seabed to the surface. The severity of barotrauma has a significant effect of the survival of released fish, and therefore there is a need to quantify the discard rates of Snapper in the MSF and determine post-release survival to estimate total fishing mortality. The estimates of mortality across sectors can then be

incorporated into the SnapEst model. This need is currently being addressed through a targeted research project (FRDC 2019-044) which is due to be completed in June 2023. The project is developing a 'Code of Practice' through consultation with each sector of the MSF to promote responsible fish handling practices, humane treatment, and harm minimisation to optimise the survival of Snapper after capture and release.

The recovery of the SG/WCS and GSVS could be supported by the continued restocking of juvenile Snapper. Stock enhancement is used to supplement natural recruitment in depleted populations where the potential for spawning and recruitment has been significantly reduced. In 2021, a total of ~200,000 juveniles were released in NSG and NGSV, with a further 85,000 released in NSG in 2022. The released fish were marked using a fluorescent compound on their otoliths to enable researchers to differentiate between wild and reared fish when they are recaptured.

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9. APPENDICES

9.1. Estimating length-at-age and weight-length relationships

Growth, as increasing mean and standard deviation of observed body lengths for every half-yearly age was estimated from catch length-at-age samples. A normal distribution of length-at-age for each cohort age is assumed. Most 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 probability density function (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

$$L_i = \frac{1}{\sqrt{2\pi}\,\sigma(a_i)} \exp\left[-\frac{1}{2}\left\{\frac{l_i - \bar{l}(a_i)}{\sigma(a_i)}\right\}^2\right] \tag{9.1.1}$$

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:

$$\bar{l}(a_i) = L_\infty \left\{1 - \exp\left[-K\left(\frac{a_i - t_0}{2}\right)\right]\right\} \tag{9.1.2}$$

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 ($\sigma(a_i)$) 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:

$$\sigma(a_i) = s_0 \cdot \left(\bar{l}(a_i)\right)^{s_1}. \tag{9.1.3}$$

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 ($\bar{l}(a_i)$).

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

$$L_i = \begin{cases} \dfrac{1}{\sigma(a_i)} \exp\left[-\dfrac{1}{2}\left\{\dfrac{l_i - \bar{l}(a_i)}{\sigma(a_i)}\right\}^2\right] \Bigg/ \left\{ \displaystyle\int_{LML}^{+\infty} \dfrac{1}{\sigma(a_i)} \exp\left[-\dfrac{1}{2}\left\{\dfrac{l - \bar{l}(a_i)}{\sigma(a_i)}\right\}^2\right] dl \right\}, if\ l_i \geq LML \\ 0,\ if\ l_i < LML \end{cases} \tag{9.1.4}$$

postulates a probability of zero for landed samples less than LML and a re-normalised 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:

$$O = -\sum_{i=1}^{n} \ln(L_i). \tag{9.1.5}$$

The estimated length-at-age curves with associated 95% confidence intervals obtained from Eq. 9.1.3 (Figure 3-13) 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 2004). Because the fishing mortality rate on SA Snapper is generally lower, 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-at-

length relationship is accurately applicable to all SA regions. Mean weight versus total length was modeled by an allometric relationship:

$$\overline{w}(l_i) = \alpha\, l_i^{\beta}\,. \tag{9.1.6}$$

A normal likelihood was again used. The standard deviation $\sigma_w(l_i)$ of the likelihood (i.e. of the fitted spread of observed weights about the mean $\overline{w}(l_i)$) 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. 9.1.3:

$$\sigma_w(l_i) \;=\; \sigma_{w0}\left(\overline{w}(l_i)\right)^{\sigma_{w1}}\,. \tag{9.1.7}$$

The resulting weight-length curve (not shown) was obtained by minimising the negative log-likelihood function. The weight-length exponent β was set equal to 2.8, with preliminary least squares estimates by region all close to 2.8. The maximum likelihood estimate of α was obtained using the TMB parameter estimation package in R. The final weight-at-length formula used for SnapEst in all SA regions is $\overline{w}(l) = 4.00\times10^{-8}\, l^{2.8}$.

9.2. Supplementary Material for DEPM

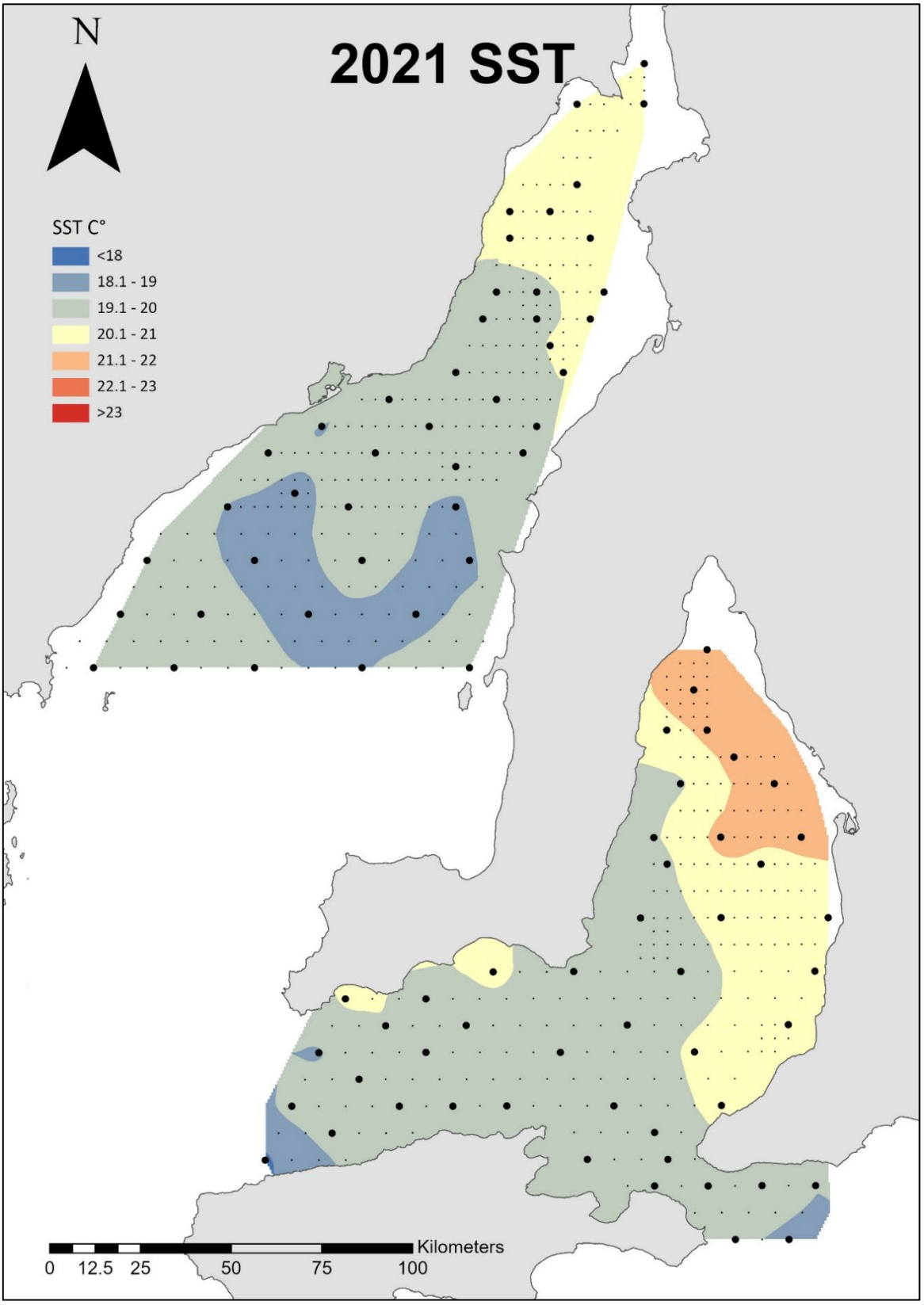

Figure 9-2.1. Mean water temperature (°C) to a depth of 5 m from the surface during the ichthyoplankton surveys in Spencer Gulf and Gulf St Vincent in 2021. Water temperature was determined from CTD casts at a subset of stations (●) in each gulf (n = 43 in SG and n = 47 in GSV / IS) and interpolated throughout the survey area using ArcGIS.

Table 9-2.1. Criteria assessed to determine spawning fraction based on the macro-analysis of the ovaries of Snapper (Saunders et al. 2012, Fowler unpublished). Also shown are the microscopic characteristics of ovaries at each stage, based on the analysis of histological slides, considering the stages of oocyte development presented in Table 9.2.2.

Stage	Mature	Spawning	Macroscopic appearance of ovary	Microscopic characteristics from histological preparations
1 – immature	No	No	Ovary small, undeveloped, clear.	Only unyolked and non-atretic oocytes.
2 – developing	Yes	No	Ovary small, opaque, light yellow in colour, individual oocytes not discernible.	Mainly unyolked and some partially yolked oocytes.
3 – developed	Yes	?	Ovary relatively large, yellow to orange in colour, individual oocytes discernible.	Dominant oocyte stage is advanced yolked oocyte. Some oocytes may be atretic.
4 – hydrated	Yes	Yes	Ovary large, yellow to orange. Clear, large translucent hydrated oocytes visible amongst smaller opaque ones. Oocytes may be ovulated, i.e., located in the oviduct.	Oocytes at all stages from unyolked to hydrated. Also, atretic advanced yolked oocytes and post ovulatory follicles may be present.
5 – regressing	Yes	No	Ovary small, flaccid, yellow to pink to dark brown.	Atretic vitellogenic oocytes outnumber vitellogenic ones. Pre-vitellogenic oocytes also present.

Table 9.2.2. Descriptions of each stage of development of oocytes, α-atretic oocytes and post-ovulatory follicles (modified from Hunter and Macewicz 1985).

Stage	Microscopic characteristics from histological preparations
1 - Unyolked	Oogonia small, cytoplasm basophilic, nuclei large, centrally located, several nucleoli occur at the periphery of the nucleus.
2 – Partially yolked	Similar to Stage 1, but larger (mean = 211 µm, range 156-312 µm, n = 392, 10 ovaries). Clear lipid granules throughout cytoplasm. Follicular layer comprises two cell layers. Zona radiata present but thin.
3 – Advanced yolked	Oocytes large (mean = 351 µm, 204 - 555 µm, n = 756, 10 ovaries). Lipid granules and eosinophilic yolk protein granules throughout the cytoplasm. Nucleus still centrally located. Zona radiata thick and highly eosinophilic.
4 – Migratory nucleus	Similar to Stage 3 oocytes except nucleus is migrating or has migrated to the peripheral cytoplasm. This represents the initiation of the hydration process.
5 - Hydrated	Oocytes much larger (mean = 822 µm, 720 – 974 µm, n = 352, 10 ovaries) with uptake of fluid. Nucleus absent. Yolk plates occupy entire volume of cytoplasm, then fuse to form a homogeneous mass. Zona radiata and follicular layers become greatly stretched.
α-atretic oocyte	Zona radiata dissolves, oocyte shape loses integrity. Yolk globules begin to disintegrate and are less regular in shape.
Post-ovulatory follicle (new)	Remaining follicle soon after ovulation is large, highly convoluted with an obvious lumen, and may contain fine granular material. The layered nature of both cell types (thecal and granulosa) remains intact.
Post-ovulatory follicle (old)	Convoluted nature much less apparent, lumen much reduced, even closed, and the thecal and granulosa cells no longer retain their orderly arrangement.

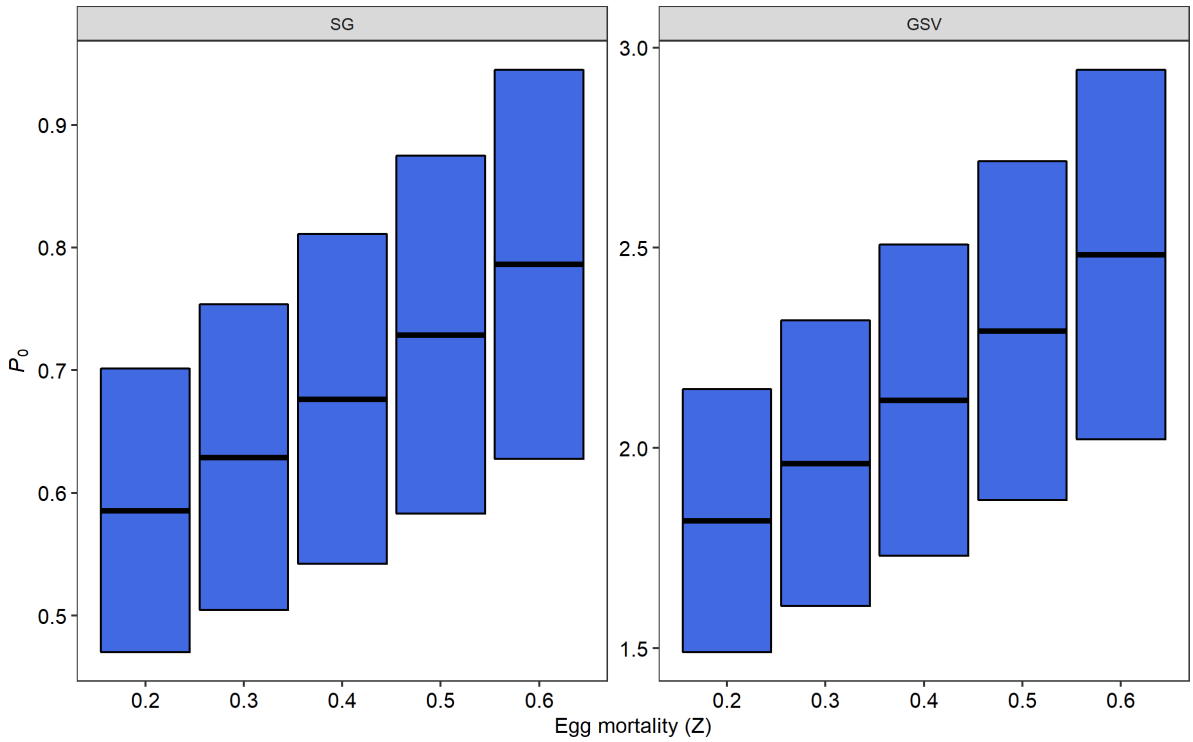

Figure 9-2.2. Estimated mean daily egg production (P_0) for Spencer Gulf (SG) and Gulf St Vincent (GSV) in 2021 using various rates of egg mortality (Z). Black lines represent P_0 and coloured bars show ± SE. Five values of Z are presented (0.2 to 0.6 day^{-1}) and Z = 0.4 day^{-1} was used in all analyses. Note the different y-axis scales between panels.

Table 9-2.3. Summary of reproductive information used to estimate spawning fraction (*S*) for each sample of Snapper from Spencer Gulf and Gulf St Vincent. Map code corresponds to Figure 4-4.

Map code	Sample code	Total no. fish	No. males	No. females	No. mature females	No. spawning females	Spawning fraction (*S*)
	Spencer Gulf	**505**	**239**	**266**	**265**	**118**	**0.44**
1	PP12/2101	92	40	52	52	0	0.00
2	PP12/2102	37	21	16	16	5	0.31
3	PP12/2103	8	1	7	7	0	0.00
5	PP12/2104	40	23	17	17	0	0.00
6	PP12/2105	24	11	13	13	0	0.00
22	MB12/2101	22	14	8	8	7	0.88
13	PB12/2101	29	13	16	16	11	0.69
14	PB12/2102	12	5	7	7	6	0.86
15	PB12/2103	25	11	14	14	9	0.64
7	WA12/2101	28	16	12	12	11	0.92
8	WA12/2102	23	9	14	14	12	0.86
9	WA12/2103	7	2	5	5	5	1.00
10	WA12/2104	9	4	5	5	5	1.00
12	ES12/2101	18	8	10	10	0	0.00
20	ES12/2102	24	15	9	9	1	0.11
4	JP12/2101	13	5	8	8	7	0.88
21	JP12/2102	18	7	11	11	11	1.00
19	MT12/2101	21	9	12	12	7	0.58
23	WA12/2105	2	2	0	0	0	n.a.
11	WA12/2106	12	4	8	8	6	0.75
17	WA12/2107	24	11	13	12	9	0.75
16	WA12/2108	12	7	5	5	5	1.00
18	WA12/2109	5	1	4	4	1	0.25
	Gulf St Vincent	**479**	**276**	**203**	**202**	**168**	**0.83**
24	AR01/2201	42	21	21	21	17	0.81
25	AR01/2202	34	20	14	14	11	0.79
26	AR01/2203	56	34	22	22	17	0.77
27	AR01/2204	11	8	3	3	1	0.33
32	NH01/2201	30	16	14	14	13	0.93
33	NH01/2202	20	13	7	7	4	0.57
35	NH01/2203	16	8	8	8	7	0.88
28	ST01/2201	9	6	3	3	3	1.00
36	ST01/2202	25	18	7	7	7	1.00
37	ST01/2203	20	7	13	12	12	1.00
34	CJ01/2201	53	29	24	24	21	0.88
38	CJ01/2202	52	33	19	19	19	1.00
39	CJ01/2203	18	12	6	6	5	0.83
42	CJ01/2204	3	1	2	2	2	1.00
43	FO01/2201	1	0	1	1	1	1.00
40	FO01/2202	2	0	2	2	0	0.00
41	FO01/2203	20	13	7	7	0	0.00
29	NH01/2204	40	23	17	17	16	0.94
30	NH01/2205	10	5	5	5	4	0.80
31	NH01/2206	17	9	8	8	8	1.00

9.3. DEPM Sensitivity Analyses

9.3.1. Sensitivity analyses for individual DEPM parameters

Introduction

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 capable of detecting changes in spawning biomass, this imprecision requires sensitivity analyses to determine which parameters could influence estimates of biomass if determined inaccurately. A sensitivity analysis was undertaken for the 2021 surveys for: spawning fraction (S), mean daily egg production (P_0), spawning area (A), and sex ratio (R).

Methods

Sensitivity analyses of individual parameters were done independently for each gulf. The sensitivity analysis for spawning fraction was performed by maintaining all other parameters (i.e., P_0, A, R, W, and F) at their mean values and altering the value of S in the DEPM equation (Equation 4.9). This same process was followed for P_0, A and R. The values included in the sensitivity analysis were determined differently for each parameter. Spawning fraction was analysed using the estimate in 2019, S of 100%, and S of 0.72 (Saunders 2009). Mean daily egg production was analysed by halving and doubling the estimated value and using the estimate from 2019. Spawning area was analysed using the estimate in 2019, an upper limit where A was the entire survey area, and a 50% decrease in A. Sex ratio (R) was analysed using the estimate in 2019, and the upper and lower bounds of previous surveys (i.e., 0.4 and 0.6).

Results

Spencer Gulf

One of the most influential parameters to estimated spawning biomass for SG was spawning fraction. This was because *S* was relatively low (0.44 ± 0.11) and therefore variance around this value has a greater influence on estimated spawning biomass than when *S* is high, i.e., spawning fraction has an inverse exponential relationship with estimated spawning biomass because it is a proportional measure (Figure 9-3.1). The high variance around the population spawning fraction reflected significant spatial variation in the spawning fraction of adult samples throughout SG (range 0.00 to 1.00). The estimated spawning fraction in 2021 was higher than in 2019 (i.e., 0.37 ± 0.05), and using the 2019 value produced an estimated spawning biomass of 129 t. The spawning fraction in 2021 was considerably lower than the value from Saunders (2009) which was used in 2013 and 2018 (i.e., 0.72 ± 0.05). Using this value, the spawning biomass would be 66 t.

Mean daily egg production has a positive linear relationship with estimated spawning biomass (Figure 9-3.1). Estimated P_0 in 2021 was 0.68 (± 0.13) which produced a spawning biomass of 108 t. The P_0 in 2021 was higher than in 2019 (i.e., 0.47 ± 0.06), and using the 2019 value produced an estimated spawning biomass of 75 t. The higher value of P_0 in 2021 resulted from sampling the same number of eggs over a smaller area. Mean daily egg production would have to be considerably underestimated to exceed the upper error margin of estimated spawning biomass (i.e., 173 t).

The significant decrease in spawning area between the 2019 and 2021 surveys was primarily responsible for the proportional decrease in estimated spawning biomass. As for P_0, spawning area has a positive linear relationship with spawning biomass (Figure 9-3.1). The spawning area in 2021 was 2,411 km^2, which was 49.5% lower than the spawning area in 2019 (4,772 km^2). Using the 2019 value resulted in an estimated spawning biomass of 213 t.

The mean sex ratio by female weight in 2021 was 0.56 (± 0.02), which was almost identical to the value in 2019 (0.55 ± 0.04). Reducing the sex ratio to the 2018 value of 0.40 (± 0.06) resulted in an increased estimated spawning biomass of 151 t (Figure 9-3.1).

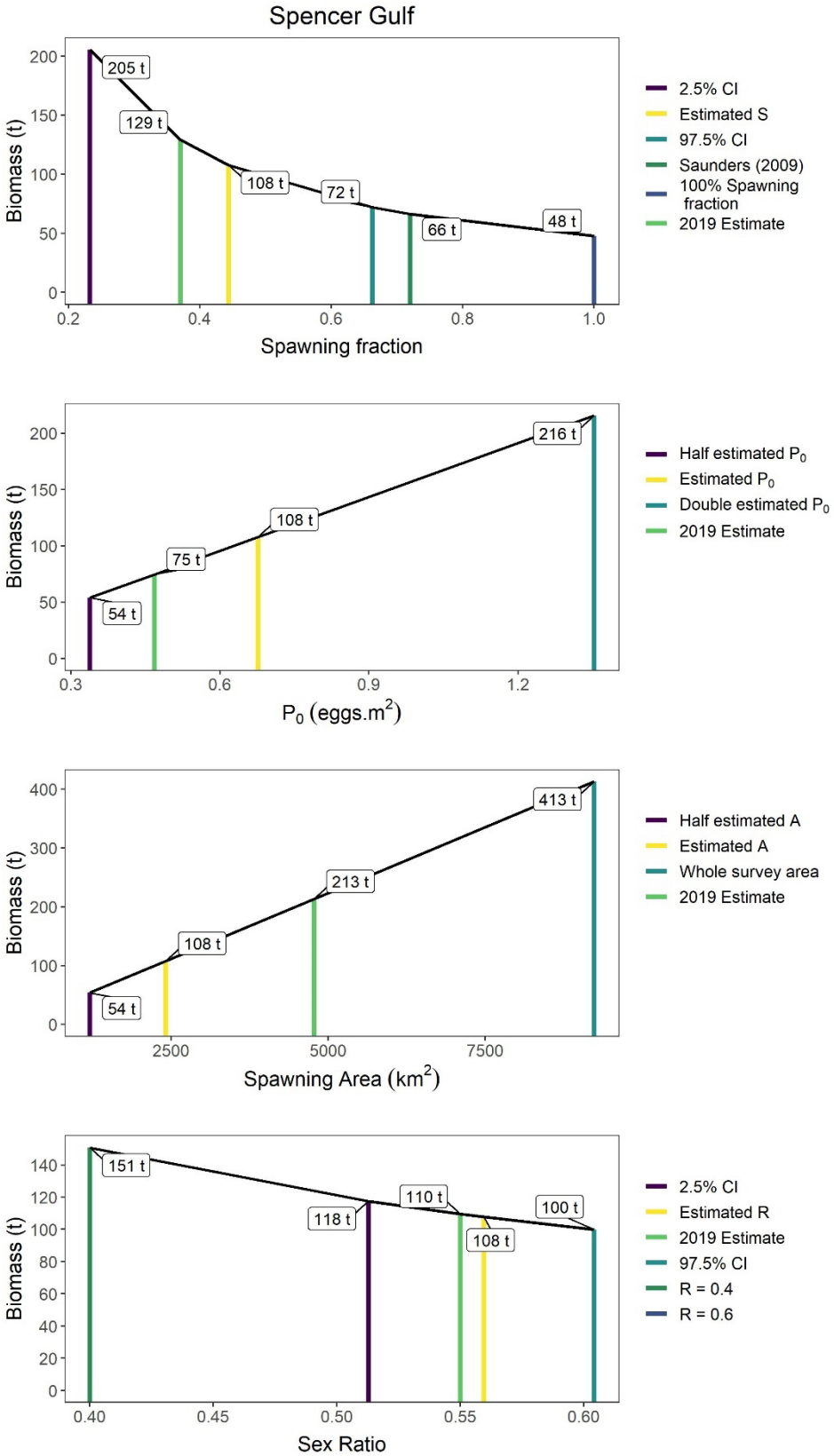

Figure 9-3.1. Sensitivity analyses for individual DEPM parameters used to estimate spawning biomass of Snapper for Spencer Gulf in 2021. The parameters are spawning fraction (*S*), mean daily egg production (P_0), spawning area (*A*), and sex ratio (*R*). Yellow lines identify the values used in 2021, green lines are the values from 2019, and the other coloured lines represent the values identified in the respective figure legends. The estimated spawning biomass produced by each value is shown where it intersects the black horizontal line.

Gulf St Vincent

The estimated spawning fraction for GSV in 2021 of 0.83 (± 0.04) was almost identical to the value in 2019 (i.e., 0.85 ± 0.10). In contrast to SG, the low variance around the population spawning fraction for GSV reflected consistently high values of *S* among adult samples (Table 9-2.1). Furthermore, the relatively high spawning fraction meant that variance in *S* had a much lower effect on estimated spawning biomass than if *S* was low (Figure 9-3.2). The spawning fraction in 2021 was higher than the value from Saunders (2009) that was used in 2013 and 2018 (i.e., 0.72 ± 0.05). Using this value, the spawning biomass would be 467 t.

Mean daily egg production in 2021 (2.12 ± 0.39) was very similar to P_0 in 2019 (2.24 ± 0.21), and therefore there was only a marginal difference in estimated spawning biomass between the two values (Figure 9-3.2). The similarity in P_0 between years resulted from a proportional decrease in the number of eggs sampled and the spawning area. Mean daily egg production would have to be considerably underestimated to surpass the upper error margin of estimated spawning biomass (i.e., 528 t).

As for SG, the significant decrease in spawning area for GSV between the 2019 and 2021 surveys was primarily responsible for the proportional decrease in spawning biomass. The spawning area in 2021 was 4,293 km^2, which was 57.5% lower than the spawning area in 2019 (10,112 km^2). Using the 2019 value, which covered >98% of the survey area, resulted in an estimated spawning biomass of 951 t (Figure 9-3.2).

The female sex ratio for GSV in 2021 (0.45 ± 0.03) was lower than in 2019 (0.54 ± 0.03). Using the 2019 value reduced the estimate spawning biomass to 334 t (Figure 9-3.2). The lowest value of *R* was recorded in 2018 (0.40 ± 0.03), which would result in an estimated spawning biomass of 450 t if used in 2021.

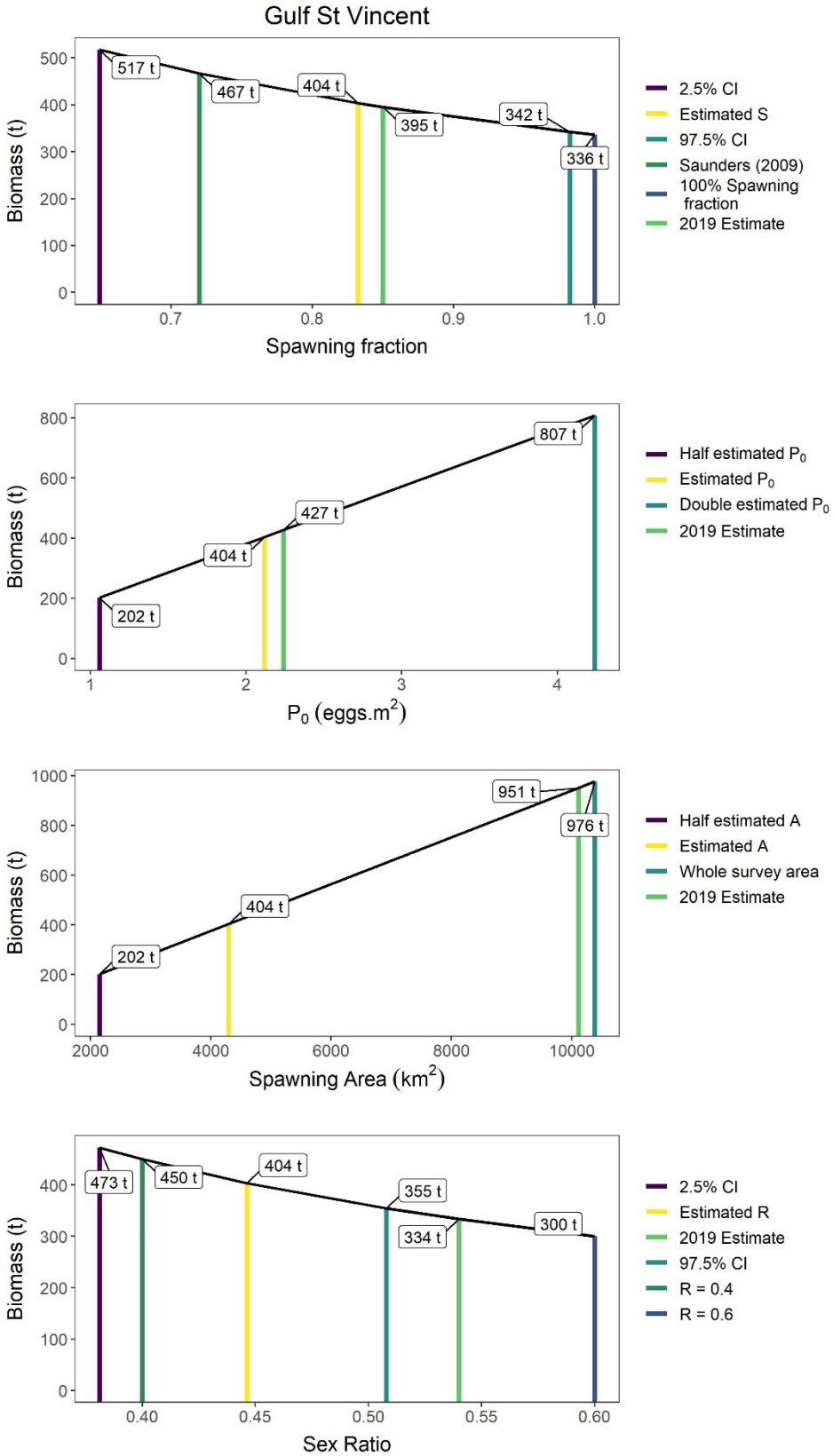

Figure 9-3.2. Sensitivity analyses for individual DEPM parameters used to estimate spawning biomass of Snapper for Gulf St Vincent in 2021. The parameters are spawning fraction (*S*), mean daily egg production (P_0), spawning area (*A*), and sex ratio (*R*). Yellow lines identify the values used in 2021, green lines are the values from 2019, and the other coloured lines represent the values identified in the respective figure legends. The estimated spawning biomass produced by each value is shown where it intersects the black horizontal line.

9.3.2. Exclude northern Spencer Gulf from DEPM estimate

There was considerable spatial variation in the distribution and abundance of Snapper eggs and spawning adults in Spencer Gulf (SG). In particular, there were no eggs sampled in northern SG (i.e., north of a line from Murninnie to Port Davis), and the adult fish sampled from this region had a very low spawning fraction of 0.05. Therefore, it could be argued that northern SG should be removed from the estimate of spawning biomass because it did not contribute to the spawning component of the population at the time of the survey. Subsequently, there were two approaches that could be used to estimate the biomass of Snapper in northern SG.

1. Include the samples of adults from northern SG into the DEPM estimate of spawning biomass for the total survey area. In which case, the biomass of Snapper in northern SG is accounted for by the spawning fraction of the population throughout the survey area.
2. Exclude the samples from northern SG and calculate adult parameters that directly relate to the region where Snapper eggs were sampled. In which case, the biomass of Snapper in northern SG is estimated by SnapEst based on the DEPM spawning biomass of the reduced survey area and scaled using proportional commercial catch.

The first approach was presented in the main body of the report. This appendix describes the second approach and compares the results.

Given there were no Snapper eggs sampled in northern SG, the removal of samples only affected adult parameters (i.e., mean female weight (*W*), female sex ratio by weight (*R*), batch fecundity (*F*), and spawning fraction (*S*)). These parameters were recalculated for the reduced survey area after five samples of adult fish were removed from the dataset (i.e., PP12/2101 to PP12/2105) (Table 4-4, Figure 4-4). Spawning biomass was then estimated for the reduced survey area following Eq. 4.9.

There were marginal differences in the estimates of *W*, *R*, and *F* for the reduced survey area. Mean female weight (*W*) increased from 3.71 kg to 4.05 kg, which also resulted in a proportional increase in batch fecundity (*F*) (Table 9-3.1). The female sex ratio by weight (*R*) was largely unchanged. There was a considerable difference in spawning fraction (*S*) which increased from 0.44 to 0.71. This difference was driven by the removal of 100 mature, but not spawning females. The estimated spawning biomass (± SE) for the total survey area in SG was 108 ± 65 t (Table 9-3.1). When the adult samples from northern SG were removed, the estimated spawning biomass decreased to 67 ± 37 t. The 38% decrease in spawning biomass was primarily associated with the considerable increase in spawning fraction.

Table 9-3.1. Comparison of DEPM parameters and the estimated spawning biomass (*SB*) (t) of Snapper for Spencer Gulf in 2021 including and excluding northern Spencer Gulf (NSG). P_0 – mean daily egg production (eggs.m^{-2}); *A* – spawning area (km^2); *W* – mean female weight (kg); *R* – sex ratio; *F* – batch fecundity of *W*; *S* – spawning fraction; SE – standard error.

DEPM survey area	P_0	*A*	*W*	*R*	*F*	*S*	*SB* (t)	SE
Total survey area	0.68	2,411	3.71	0.56	223,501	0.44	108	65
NSG excluded	0.68	2,411	4.05	0.57	244,159	0.71	67	37

The proportional commercial catch of Snapper for the DEPM survey area relative to the entire model region (i.e., Prop_DEPM) was used to scale the estimate of spawning biomass in SnapEst. The survey area that was excluded in northern SG corresponded to the total area of MFA 21 (i.e., sub-blocks 21A, 21B, and 21C). Prop_DEPM for the reduced survey area was calculated from the half-yearly summer catches for the most recent uninterrupted fishing year (i.e., October 2018 to March 2019). The decrease in Prop_DEPM from 0.5445 to 0.3301 reflected the large proportion of catch that was taken from MFA 21 (Table 9-3.2).

There was a negligible (<1%) difference in the estimates of fishable biomass for the SG/WC Stock from SnapEst between the two methods (Table 9.3.2).

Table 9-3.2. Comparison of DEPM inputs for SnapEst including and excluding northern Spencer Gulf (NSG). SB – estimated spawning biomass from DEPM (t); CV_DEPM – coefficient of variation for DEPM estimate; Prop_DEPM – proportional commercial catch from the DEPM survey area relative to the entire SnapEst model region. Values in parentheses are standard error.

DEPM survey area	*SB* (t)	CV_DEPM	Prop_DEPM	SnapEst fishable biomass (t)
Whole survey area	108 (65)	0.6008	0.5445	544 (65)
NSG excluded	67 (37)	0.5541	0.3301	549 (66)

9.3.3. Geostatistical methods to estimate total daily egg production.

Traditionally, mean daily egg production (P_0) and spawning area (A) are considered independent parameters in the DEPM equation (Eq. 4.1), despite the fact they are inherently correlated, calculated using the same dataset, and are therefore not truly independent. Therefore, we consider the application of geostatistical methods to estimate total daily egg production (i.e., $P_0 \times A$) as a logical approach to overcome this issue and to reduce uncertainty in the estimate of total egg production. Subsequently, several geostatistical methods were explored during the DEPM analysis and incorporated into SnapEst to understand their effect on subsequent estimates of fishable biomass.

Four methods to estimate total daily egg production were compared:

- Independent $P_0 \times A$: The two parameters were calculated independently for the entire survey area (i.e., each gulf) and multiplied together. This is the approach that has been used in all applications of DEPM for Snapper and was described in Section 4.2.1. Mean daily egg production (P_0) was calculated in R following McGarvey et al. (2018) and A was calculated in ArcGIS.
- Ordinary kriging: An interpolation method that makes predictions at unsampled locations using a linear combination of observations at nearby sampled locations. Calculated in R.
- Natural neighbour: Finds the closest subset of input samples to a query point and applies weights to them based on proportionate areas to interpolate a value. Calculated in ArcGIS.
- Direct sum: Mean daily egg production at each station (P_t) was multiplied by the area of each station (A_t), and total daily egg production for each gulf was calculated as the sum of $P_t \times A_t$ for all stations. Mean daily egg production at each station was calculated in R following McGarvey et al. (2018) and A_t was the area of the polygon around each station.

The estimates of total daily egg production using geostatistical methods were consistently lower than when P_0 and A were considered independent parameters (Table 9-3.3). There was considerable similarity in the estimates of $P_0 \times A$ using the Natural Neighbour and Direct Sum methods. The estimates of $P_0 \times A$ from Ordinary Kriging were comparatively variable and did not follow a consistent trend between regions or years. The differences in total daily egg production between methods were reflected in the subsequent estimates of spawning biomass (Table 9-3.3).

Table 9-3.3. Comparison of geostatistical methods to estimate total daily egg production ($P_0 \times A$) and the associated DEPM estimates of spawning biomass (*SB*) (t) for Snapper surveys in Spencer Gulf and Gulf St Vincent in 2019 and 2021. P_0 – mean daily egg production (eggs.m^{-2}); *A* – spawning area (km^2).

	2019				2021			
Method	P_0	*A*	$P_0 \times A$	*SB* (t)	P_0	*A*	$P_0 \times A$	*SB* (t)
Spencer Gulf								
Independent $P_0 \times A$	0.47	4,773	2,243	239	0.68	2,411	1,640	108
Ordinary kriging			1,259	135			1,273	84
Natural neighbour			1,508	162			1,207	80
Direct sum			1,508	162			1,208	80
Gulf St Vincent								
Independent $P_0 \times A$	2.24	10,111	22,649	812	2.12	4,293	9,101	404
Ordinary kriging			15,665	562			8,216	364
Natural neighbour			15,619	560			6,362	282
Direct sum			15,802	567			6,391	283

9.4. Supplementary Material for 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 pseudo-code, is described in Appendix C of the 2015 Garfish stock assessment report (Steer et al. 2016).

9.4.1. 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: $\sigma(a) = \sigma_0 \cdot \left(\bar{l}(a)\right)^{\sigma_1}$.

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 σ_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 $(n_L)^2$ 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.

9.4.2. Recruitment

Recruitment is defined as the creation of the (normal) length-at-age cohort at age $a_b = 5$ half-years (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 number of Snapper above LML 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 sub-model (Appendix C of Steer et al. 2016).

9.4.3. 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 77 = summer 2022) "t", (2) spatial region (1 = SG/WC, 2 = GSV, 3 = SE) "r", (3) cohort (i.e., year-class, given by year of spawning) "c", and (4) slice "s". For the SE Region, the last model time step is 77.

Variable subscripts for winter or summer half-year (t_{season}), 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.

The initial state of the population structure at time step 1 was created as follows. Prior to model recruitment cohort 1982, most cohort years back to 1960 are estimated by a single common estimated parameter that is shared among all years, except for the gulfs where a separate parameter is applied for cohorts 1972 and 1978. A fixed level of initial fishing mortality is assumed overall years prior to 1983, of value 0.06. The initial population state is then calculated by applying natural and fishing mortality for each of the constituent recruitment cohorts composing the population, from cohort 1981 back to 1960 inclusive, employing the slice growth mechanism at each time-step.

9.4.4. 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 cut-off 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 yr^{-1} (Gilbert et al. 2006; Gilbert, pers. comm.).

Length selectivity, s_{len}, by region is applied for the longline gear only ($g(i_E = 2)$) and follows a logistic function of fish length, the latter specified by the midpoint of each slice (*s*) ($\bar{l}(s)$), as:

$$s_{len}(r, g(i_E), s) = 1 \Big/ \left(1 + \exp\left[-r_{sel}\left(r, g(i_E)\right) \cdot \left(\bar{l}(s) - l_{50}\left(r, g(i_E)\right)\right)\right]\right) \qquad (9.4.1)$$

where $r_{sel}\left(r, g(i_E)\right)$ is the logistic slope parameter (fixed at 1), $l_{50}\left(r, g(i_E)\right)$ is the logistic 50% level parameter. A decreasing logistic selectivity function of fish half-yearly age (a), s_{age}, was applied for handline only ($g(i_E = 1)$), by region (SG/WC and GSV), as:

$$s_{age}(r, g(i_E), a) = 1 - 1 \Big/ \left(1 + \exp\left[-r_{agesel}\left(r, g(i_E)\right) \cdot \left(a - l_{50agesel}\left(r, g(i_E)\right)\right)\right]\right) \qquad (9.4.2)$$

where r_{agesel} is the logistic slope parameter and $l_{50agesel}$ is the logistic 50% level parameter.

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.

$\tilde{C}w(t,r,i_E)$ = commercial catch data that is reported in weight (kg) for time step, *t*, region, *r*, and effort type $i_E < n_E - 1$.

$\tilde{C}n(t,r,i_E)$ = charter boat ($i_E = n_E - 1$) 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 non-survey periods, catch data input into SnapEst are obtained by linearly interpolation between the surveys. Catches from the 2013/14 survey are carried forward for SG/WC 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 $\hat{C}n(t,r,i_E)$ and $\hat{C}w(t,r,i_E)$ for each $i_E \le n_E$, which are computed as sums over cohorts (*c*) and legal sized slices (*s*) of $\hat{C}n(t,r,c,s,i_E)$ and $\hat{C}w(t,r,c,s,i_E)$, defined below.

$$\hat{C}n(t,r,c,s,i_E) = \exp\left[-0.5 \cdot M\right] \cdot N(t,r,c,s) \cdot s_{len}(r,g(i_E),s) \cdot s_{age}(r,g(i_E),a(t,c)) \cdot H\exp(t,r,i_E) \tag{9.4.3}$$

and

$$\hat{C}w(t,r,c,s,i_E) = \hat{C}n(t,r,c,s,i_E) \cdot w(a(t,c),s) \tag{9.4.4}$$

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 $H\exp(t,r,i_E)$ is the predicted harvest fraction of the exploitable population or biomass (i.e., portion of fishable biomass accessible by the gear).

$H\exp(t,r,i_E)$ 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 $i_E < n_E - 1$,

$$H\exp(t,r,i_E) = \frac{\tilde{C}w(t,r,i_E)}{\exp\left[-0.5\cdot M\right]\cdot \sum_{c=cohort1}^{cohort23plus}\sum_{s=1}^{nlegs} s_{len}(r,g(i_E),s)\cdot s_{age}(r,g(i_E),a(t,c))\cdot w(a(t,c),s)\cdot N(t,r,c,s)}$$

(9.4.5)

and for $i_E = n_E - 1 \; or \; n_E$

$$H\exp(t,r,i_E) = \frac{\tilde{C}n(t,r,i_E)}{\exp\left[-0.5\cdot M\right]\cdot \sum_{c=cohort1}^{cohort23plus}\sum_{s=1}^{nlegs} s_{len}(r,g(i_E),s)\cdot s_{age}(r,g(i_E),a(t,c))\cdot N(t,r,c,s)}$$

(9.4.6).

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

$$N(t+1,r,c,s) = \left(\exp\left[-0.5\cdot M\right]\cdot N(t,r,c,s) - \hat{C}n(t,r,c,s,i_E)\right)\cdot \exp\left[-0.5\cdot M\right] \quad (9.4.7).$$

9.4.5. Estimation: Parameters and model likelihood

The model likelihood (Fournier and Archibald 1982) was fitted, for each region, to half-yearly (1) handline (SG/WCS and GSV) and longline (SE Region) catch rates (catch totals by weight divided by effort totals), (2) absolute biomass from DEPM surveys (SG/WCS and GSVS), (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.

<u>Parameters</u>

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.

<u>Likelihood for catch rates</u>

Predicted catch rates were fitted to data for the handline (HL, $i_E = 1$) effort type for the gulfs regions (r = 1 and 2) and to longline (LL, $i_E = 2$) effort type for the SE Region (r = 3). CPUE was fitted using a lognormal likelihood. The model time steps (t) over which catch rates were fitted varied by region. In the SE, LL catch rate was fitted over all model time steps. In SGWC, HL catch rate was fitted for $t \le 61$ (up to and including summer 2013/14 when DEPM surveys

commenced), and in GSV HL was fitted for $t \le 63$ (summer 2014/15, when DEPM commenced in that gulf). The catch rate data, $\tilde{C}pue(t,r,i_E)$, was calculated in each time step as the ratio of total catch by weight (kg) divided by total effort. Fishing effort was calculated as fisher-days in all years, except the SE Region for which used number of hooks from 2003 onwards. Yearly CPUE values were rescaled so that LL CPUE using fisher-days joins with the CPUE series using hooks in 2003. Predicted catch rate was calculated as

$$\hat{C}pue(t,r,i_E) = q(t_{season},r,i_E) \cdot B\exp_{\text{mid}}(t,r,i_E) \tag{9.4.8}$$

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

$$B\exp_{mid}(t,r,i_E) = \sum_{c=cohort1}^{cohort23plus} \sum_{s=1}^{nlegs} s_{len}(r,g(i_E),s) \cdot s_{age}(r,g(i_E),a(t,c)) \cdot w(a(t,c),s) \cdot \left(\exp\left[-0.5 \cdot M\right] \cdot N(t,r,c,s) - 0.5 \cdot \hat{C}n(t,r,c,s,i_E)\right) \tag{9.4.9}$$

The likelihood for fitting to catch rates was written:

$$L_{Cpue} = \prod_{t=1}^{n_t} \prod_{r=1}^{n_r} \frac{1}{\sqrt{2\pi} \cdot \sigma_{Cpue}(t_{season},r,i_E) \cdot \tilde{C}pue(t,r,i_E)} \exp\left[-\frac{1}{2}\left(\frac{\ln[\hat{C}pue(t,r,i_E)] - \ln[\tilde{C}pue(t,r,i_E)]}{\sigma_{Cpue}(t_{season},r,i_E)}\right)^2\right] \tag{9.4.10}$$

Where: $\sigma_{Cpue}(t_{season},r,i_E)$ = estimated standard deviation parameter, one per season, region, and effort type (HL/LL).

<u>Likelihood for DEPM biomass</u>

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: SG/WC time steps (t) = {61, 71, 73, 77} and GSV (t) = {63, 71, 73, 77}. Legal size ($\ge$380 mm) DEPM spawning biomass was assumed to approximate 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 5.2.

The likelihood for fitting to the DEPM biomass was written:

$$L_{DEPM} = \prod_{t=1}^{n_t}\prod_{r=1}^{n_r} \frac{1}{\sqrt{2\pi}\cdot\sigma_{DEPM}(surv(t),r)} \exp\left[-\frac{1}{2}\left(\frac{P(surv(t),r)\cdot B_{\text{mid}}(t,r) - Bdepm(surv(t),r)}{\sigma_{DEPM}(surv(t),r)}\right)^2\right]$$

(9.4.11)

Where:

$B_{\text{mid}}(t,r)$ = predicted total fishable biomass mid-way into time step *t* (after exactly half of the catch is taken) = $\sum_{c=cohort1}^{cohort23plus}\sum_{s=1}^{nlegs} w(a(t,c),s)\cdot\left(\exp\left[-0.5\cdot M\right]\cdot N(t,r,c,s) - 0.5\cdot\hat{C}n(t,r,c,s,i_E)\right)$,

$Bdepm(surv(t),r)$ = DEPM biomass estimate from the survey at time step *t,*

$P(surv(t),r)$ = 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

$\sigma_{DEPM}(surv(t),r)$ = fixed standard deviation parameter, one per survey and region.

<u>Likelihood for catch samples by age</u>

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, $\tilde{n}(a;\ i_A)$. 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

$$L_{Ages} = \prod_{i_A=1}^{n_A}\prod_{a=a_b}^{48+} \hat{p}(a;\ i_A)^{\tilde{n}_{cor}(a;\ i_A)} \qquad (9.4.12)$$

where:

i_A = index over the set of n_A catch samples of fish ages over half-year, region, and gear;

$\hat{p}(a;\ i_A)$ = an array of model-predicted fish proportions captured by age for each sample indexed by i_A;

$\tilde{n}_{cor}(a;\ i_A)$ = scaled and corrected observed fish numbers for each age in the catch-at-age sample i_A.

<u>Likelihood for catch samples by length</u>

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:

$$L_{Lengths} = \prod_{i_A=1}^{n_A(i_{mp})} \prod_{i_{mp}=1}^{4} \prod_{a=a_b}^{48+} \left\{ \frac{\exp\left[-\frac{1}{2} \left(\frac{\left\{ \tilde{b}(i_{mp}, a; i_A) - \hat{b}(i_{mp}, a; i_A) \right\}}{\sigma_{mp}} \right)^2 \right]}{\sqrt{2\pi} \cdot \sigma_{mp}} \right\}^{\tilde{n}(a;\, i_A)} \qquad (9.4.13)$$

where

σ_{mp} = is the estimated moment-likelihood standard deviation parameter, separately per season, region, and gear, but not differing among all four moments.

$\tilde{b}(i_{mp}, a; i_A)$ = observed moment, indexed by i_{mp}, per sample and half-yearly age.

$\hat{b}(i_{mp}, a; i_A)$ = model-predicted counterpart to $\tilde{b}(i_{mp}, a; i_A)$.

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 ($\tilde{n}(a;\ i_A)$), 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 $L_{Lengths}$ likelihood, depending on the moment property fitted. Thus, the number of qualifying data sets, $n_A(i_{mp})$, decreased with increasing moment property i_{mp}. 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

$$-\left(\lambda_{Cpue}\cdot\ln[L_{Cpue}]+\lambda_{DEPM}\cdot\ln[L_{DEPM}]+\lambda_{Ages}\cdot\ln[L_{Ages}]+\lambda_{Lengths}\cdot\ln[L_{Lengths}]\right) \quad (9.4.14)$$

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

9.4.6. 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 some recruitment years. These were set to a minimum value of 1000 for the SE Region for 1984-1990, 2019-2020, and to 1281 for 2016-2018. For SG and GSV, only 2019-2020 were set to 1000.

Annual fishable biomass

The fishable biomass indicator reported in Section 5.3 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

$$B(y,r)=\frac{1}{2}\sum_{t=t_{season}(1,y)}^{t_{season}(2,y)}B(t,r) \quad (9.4.15)$$

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

$$B(t,r)=\sum_{c=cohort1}^{cohort23plus}\sum_{s=1}^{nlegs}N(t,r,c,s)\cdot w(a(t,c),s). \quad (9.4.16)$$

For each time step t, ranging from 1 (October 1983 to March 1984) to 77 (October 2021 to March 2022), 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 9.4.15 span the two seasonal half-yearly times steps in each full year ($t_{season}(1,y)$ = October-March, $t_{season}(2,y)$ = 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

$$H(y,r) = \frac{\sum_{i_E=1}^{n_E} \sum_{t=t_{season}(1,y)}^{t_{season}(2,y)} \hat{C}w(t,r,i_E)}{B(y,r)} \tag{9.4.17}$$

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 $fm(t,c,s) = 61398 \cdot \left[w(a(t,c),s)\right]^{0.9942}$ 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 139-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

$$Eggs(y,r) = \sum_{c=cohort1}^{cohort23plus} \sum_{s=0}^{nlegs} 0.5 \cdot \left(\exp\left[-0.5 \cdot M\right] \cdot N(t_{season}(1,y),r,c,s) - 0.5 \cdot \hat{C}n(t,r,c,s,i_E)\right) \cdot fm(t,c,s) \tag{9.4.18}$$

9.5. SnapEst Sensitivity Analyses

Spawning biomass from the DEPM represents an important data source for SnapEst model estimates in the two gulfs, particularly since those regions were closed to fishing from November 2019 leaving only DEPM to inform stock abundance. Consequently, several scenarios were tested for how the DEPM biomass estimates were computed and used in SnapEst.

Due to historically high uncertainty in the DEPM parameter of spawning area (A), and because the two DEPM parameters of spawning area and daily spawning egg density (P_0) are not independent, but are treated as independent in traditional DEPM analysis, we tested a number of alternative methods to combine A and P_0 in DEPM biomass estimates. These new A times P_0 interpolation analysis methods compute total egg production within the DEPM survey area by either geostatistical interpolation and summing under the interpolation surface or by directly computing the sum of $\sum_{t}^{nstations} P_t \cdot A_t$ over all DEPM survey stations subscripted by t. A_t is the polygon area surrounding each survey sample location in m^2, and P_t is the computed daily spawning egg density in eggs/m^2/day estimated using the stage-based method (McGarvey et al. 2018) that gives a single estimate of spawning egg density P_t at each station. A range of geostatistical interpolation methods were tested, and two are presented here, natural neighbour and ordinary kriging. These alternative methods for estimating $P_0 \cdot A$ (direct sum $\sum_{t}^{nstations} P_t \cdot A_t$, natural neighbour, and kriging) do not treat these parameters as independent. For each alternative A* P_0 product sensitivity run tested, DEPM biomass was computed, and taken as input to the model in the last two DEPM summer half-years of 2019 and 2021. These sensitivity model-estimated biological performance indicators are plotted alongside the baseline in Figures 9-5.2 and 9-5.4 below.

For the SE Region, the index of abundance fitted by SnapEst is commercial catch rate. Two alternative choices for the fitted CPUE index were tested, one being the index used in previous assessments, namely that based on HL fisher-days for all years, the other being LL fisher-days for all years, in contrast to the 'baseline' which uses LL fisher-days only until 2003 but thereafter number of hooks.

For all sensitivity test figures presented in this section, the tested model sensitivity run outputs are presented alongside the 'baseline' SnapEst run presented in the main text of Chapter 5 and further described in section 9-4 above.

9.5.1. Spencer Gulf / West Coast Stock

For SG/WCS, three sensitivity analyses were undertaken. First, we tested for different ways to compute the combination of A and P_0. These three alternative methods for estimating $P_0 \cdot A$ all gave nearly identical results to the baseline (Figure 9-5.1). Therefore, the SnapEst model estimates are not sensitive to the input DEPM biomass estimates for the last two surveys, summer spawning seasons of 2019 and 2021.

Second, we examined the impact of omitting either the most recent DEPM biomass estimates, or all of them. When all DEPM estimates for SG were removed from the model, a greatly different outcome was observed (green lines of Figure 9-5.2). The resulting estimates of biomass were an order of magnitude higher than the estimates of biomass from the DEPM and from all other sensitivity scenarios (Figure 9-5.2). 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 needed for producing reliable levels of biomass in model estimates, i.e., to anchor absolute population levels and to infer the large stock decline. However, when only the 2021 DEPM biomass was omitted as a data input, the model and baseline estimates were very similar (Figure 9-5.2).

Thirdly, in SG only, we tested for the effect of a second way to account for the NSG sampling where no eggs were counted in the DEPM survey but some Snapper were present (Figure 9-5.3). The result, again, showed negligible impact on final model outcomes.

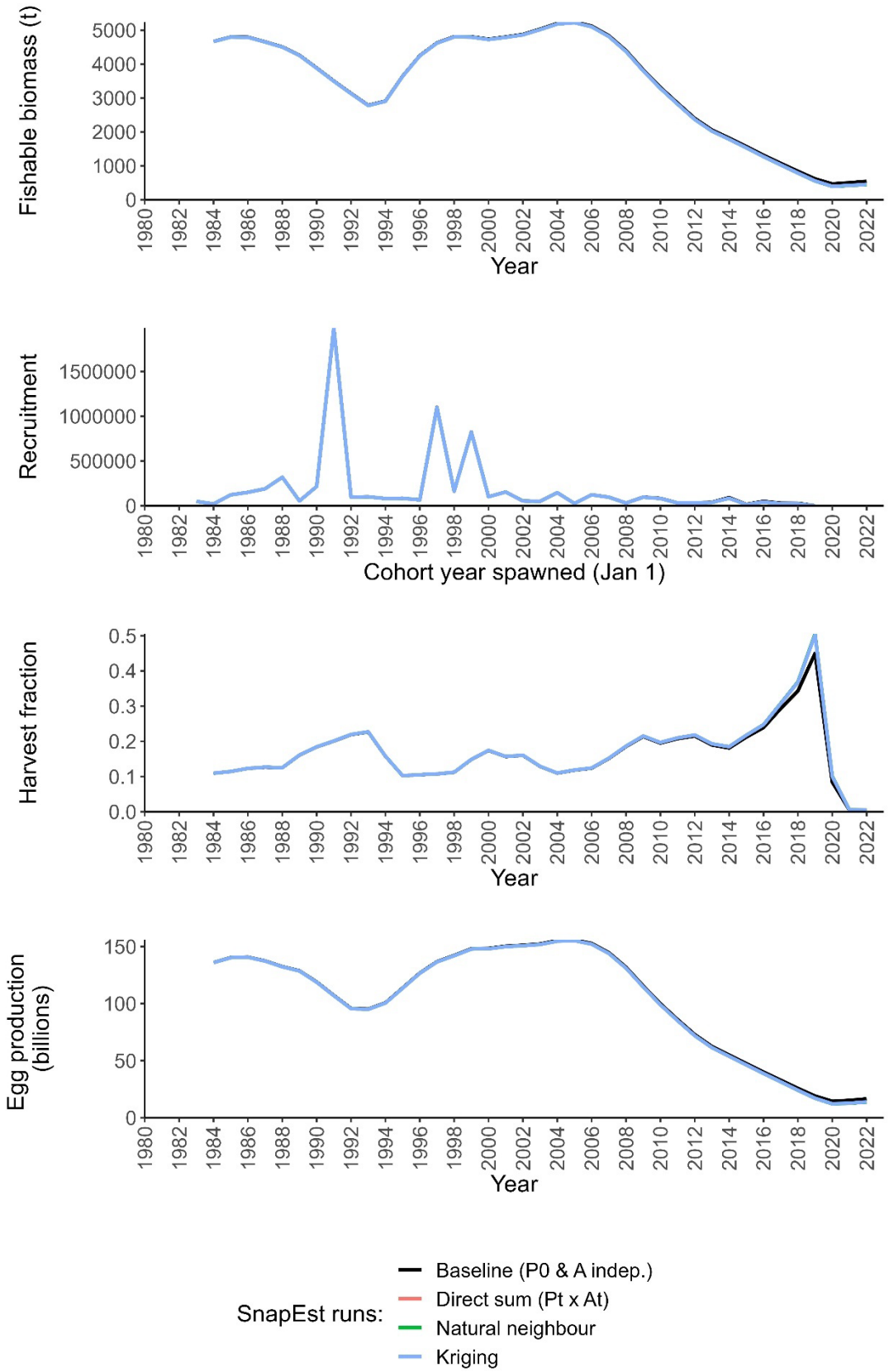

Figure 9-5.1. Sensitivity test for SG/WCS, comparing four methods to compute the product of daily spawning egg density (P_0) and spawning area (A).

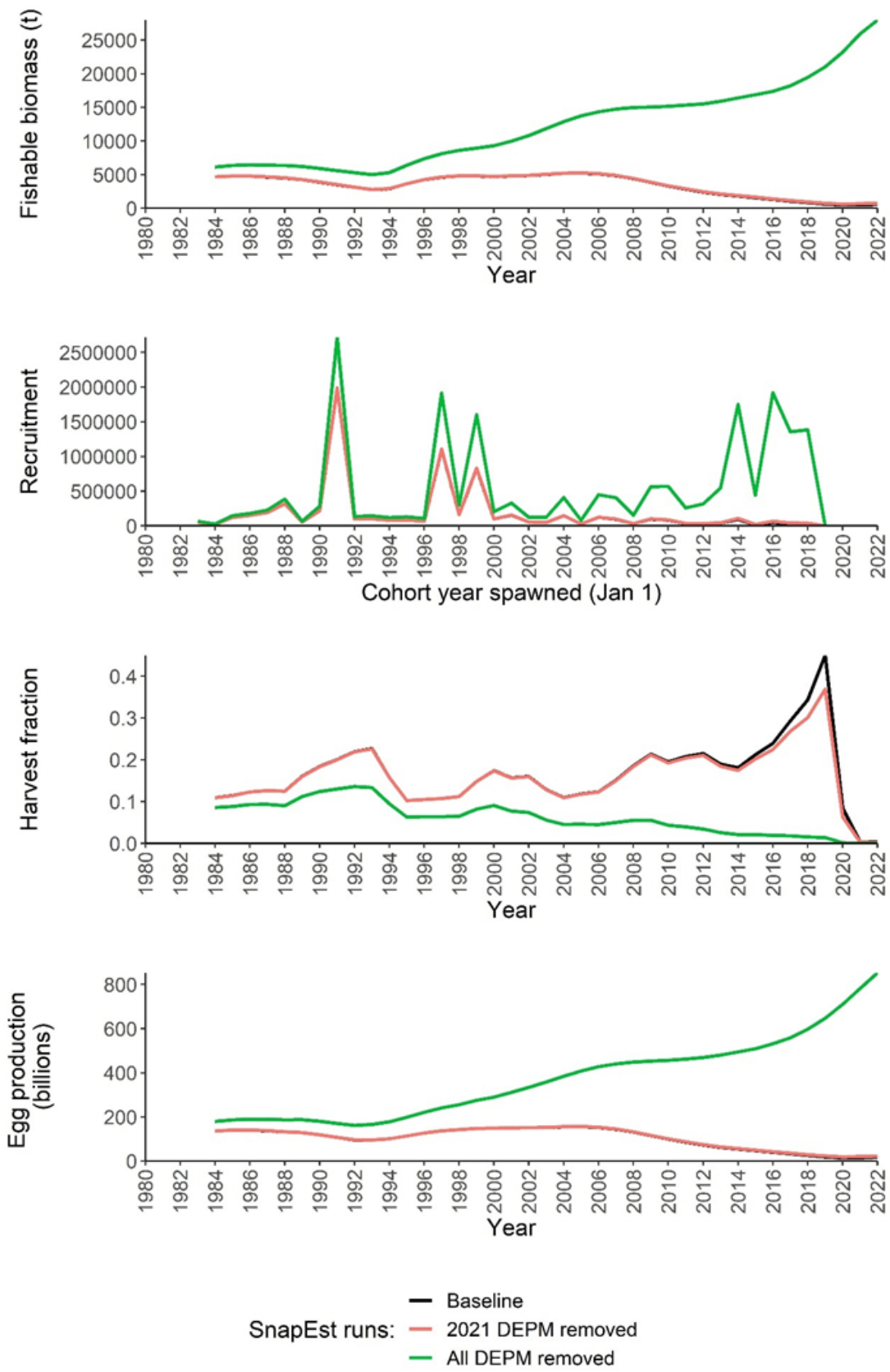

Figure 9-5.2. Sensitivity test for SG/WCS, comparing baseline to runs where either the most recent DEPM biomass was omitted as model input, or all DEPM biomass estimates were omitted.

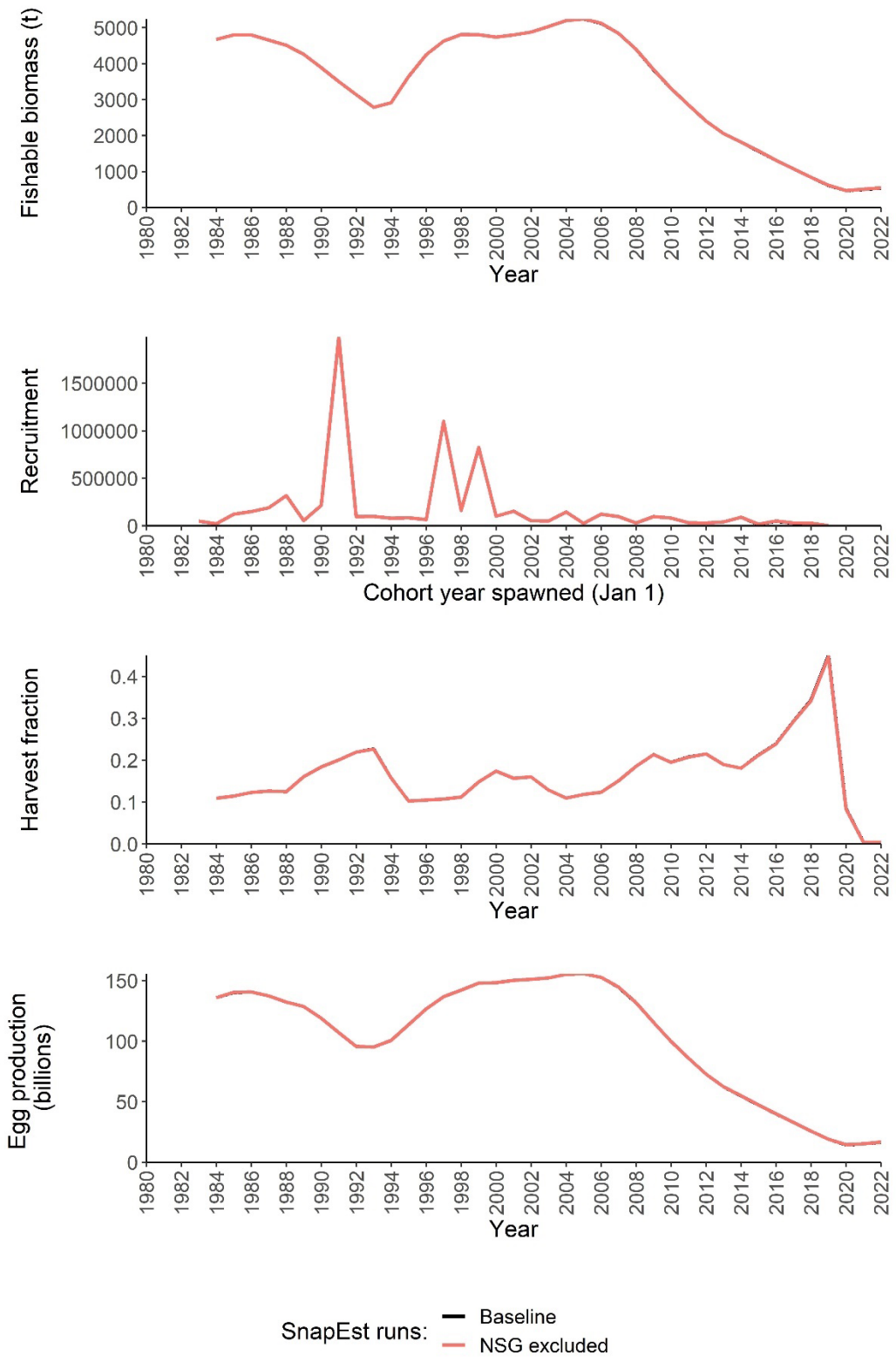

Figure 9-5.3. Sensitivity test in SG/WCS, comparing two methods to account for NSG Snapper in the SnapEst model. In the baseline, the standard DEPM approach was used, where the DEPM survey study region is treated as a single whole region, and the egg counts and female spawning fraction from the entire study region are estimated and used. In the alternative shown here, the egg counts and spawning fraction from NSG Snapper were removed from the DEPM spawning biomass estimate, and these Snapper were accounted for by computing their biomass based on the proportion of commercial catch taken in NSG as a proportion of the total catch from the SG/WCS.

9.5.2. Gulf St Vincent Stock

In the GSVS, the first two sensitivity analyses undertaken for SG/WCS were repeated. The three alternative methods for estimating $P_0 \cdot A$ again gave nearly identical estimates to the baseline (Figure 9-5.4). The SnapEst model estimates in both gulfs are insensitive to the input DEPM biomass estimates for the last two surveys, 2019 and 2021.

Second, we again examined the impact of omitting either the most recent DEPM biomass estimates, or of omitting all years of DEPM biomass as a model input. Unlike in the SG/WCS, the model sensitivity run that omitted all years of DEPM biomass gave good agreement with baseline, only exhibiting a decrease in biomass over 2018-2022 (by 4%-19%). This implies, in Gulf St. Vincent (but not Spencer Gulf) that the standard data set available, with strong and long-term age samples and potentially more informative catch rates in years prior to DEPM survey, can produce usable model estimates without a fishery-independent measure of stock biomass.

Because the extent to which these GSV runs differ is unexpectedly small, we added two further runs where the initial parameters were variously modified in both the baseline version of SnapEst and in the run with DEPM entirely omitted (Figure 9-5.5). Most types of estimated parameters (i.e. growth, recruitment, absolute catchability, length selectivity, likelihood sigma) had their initial values either increased or decreased by 20% relative to baseline estimates, except for the L50 parameter for the age selectivity curve which had values increased by 10%. These induced only small variation (increase of 1.4% in 2022 biomass), supporting further robustness of this close agreement of SnapEst outputs for these two model fits.

Other tests of this robustness of runs with and without DEPM biomass added were undertaken (figure plots not shown). These include increasing the weighting on DEPM by a factor of 10 (equivalent to a reduction by a factor ~3.16 in the CV of those fitted DEPM biomass estimates), or alternatively increasing by 20% the biomass datum input from DEPM survey 2021, which again showed only modest variation in the final model assessment outcomes (increases in 2022 GSV biomass of 5% and 8% respectively).

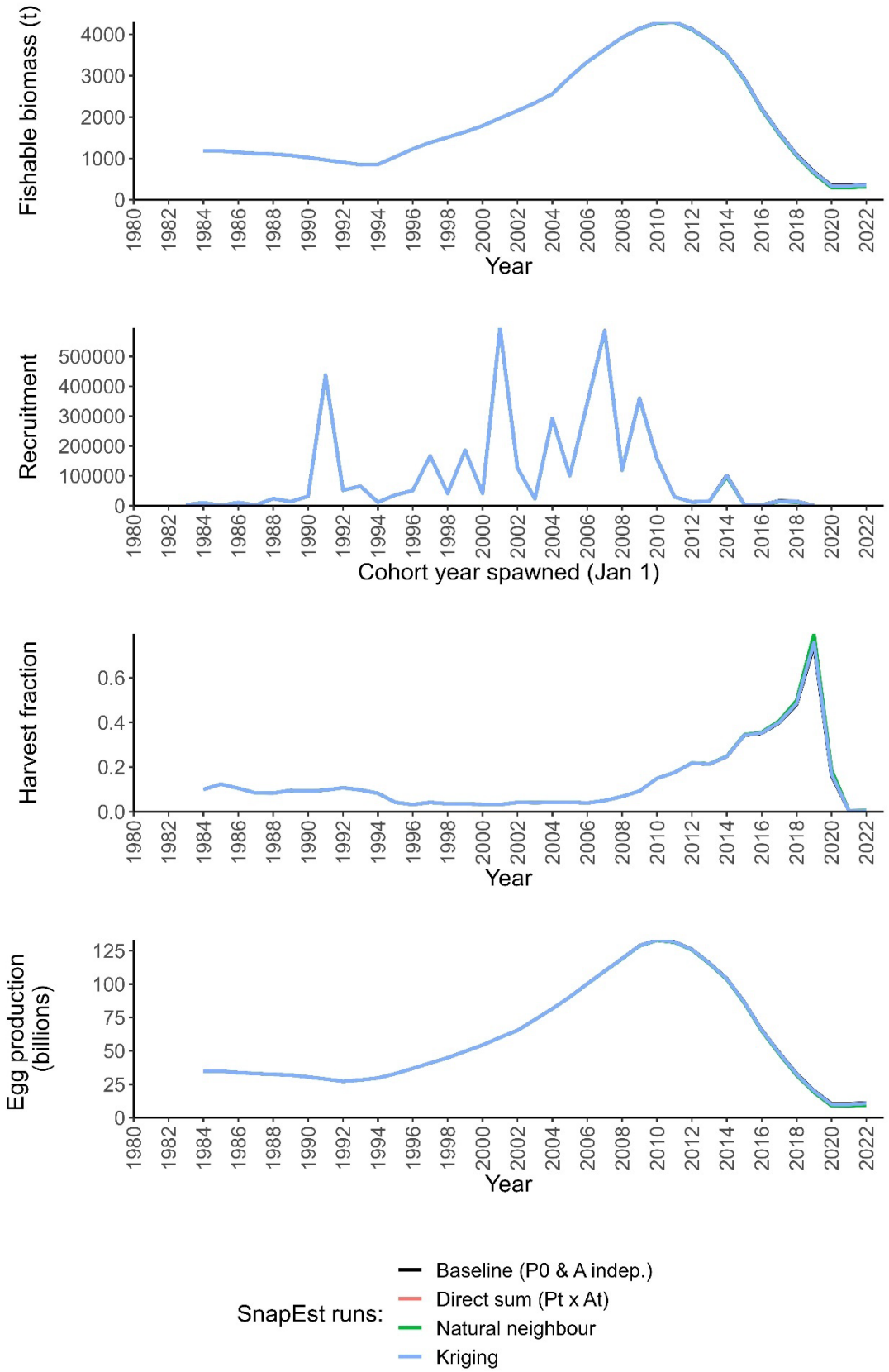

Figure 9-5.4. Sensitivity test for GSVS, comparing four methods to compute the product of daily spawning egg density (P_0) and spawning area (A).

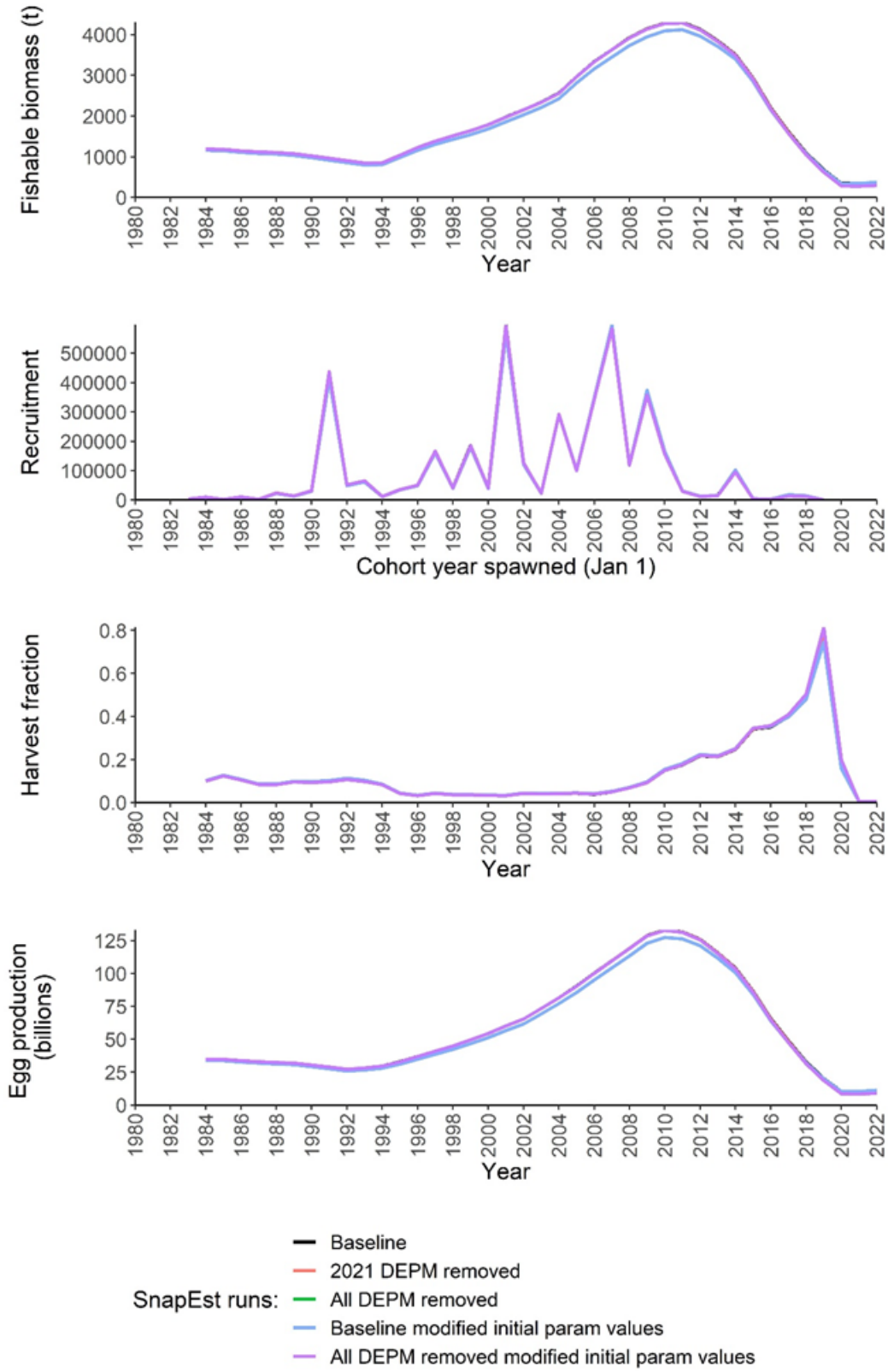

Figure 9-5.5. Sensitivity test for GSVS, comparing baseline to runs where either the most recent DEPM biomass was omitted, or all DEPM biomass estimates.

9.5.3. South-East Region

In the previous assessment, the index of abundance used was commercial HL CPUE. However, commercial fishing with HL has largely ceased in the SE Region, and in the last two winter half-yearly model time steps, zero HL effort was reported meaning no information on HL catch rate is available. In this 2022 assessment, the baseline model was fitted to LL CPUE, which comprised nearly all of the commercial catch (Figure 2-5). Two versions of LL CPUE are employed in the baseline. For years since 2003 when number of hooks employed in each day's LL fishing has been reported on commercial catch logs, CPUE was computed using total hooks set as the measure of effort in the CPUE denominator. Prior to 2003, LL CPUE with fisher-days as the measure of effort was used as the baseline model index of abundance. In the construction of the predicted catch rate we employed a common catchability parameter applied in all years.

The sensitivity analysis for the SE Region (Figure 9-5.6) compares this baseline with two alternative model runs using LL/fisher-days for all model years, or the previous CPUE measure of HL/fisher-days again fitted for all years.

This SE Region sensitivity comparison among indices of abundance (here CPUE) shows very large differences in model outcomes, mainly for years since the biomass peak around 2009 (Figure 9-5.6). The CPUE index using LL/fisher-days (for all years) gives estimates of very high biomass, above even the 2009 peak in the most recent years, with recruitment and egg production tracking this trend. In strong contrast, the CPUE index using HL/fisher-days implied on-going lower stock abundance. The baseline run provides a middle ground between these two sensitivity outcomes.

This baseline uses what is believed to be a better CPUE index using hooks set, since 2003 when this measure of LL fishing effort became available. On a given fisher-day fishers can set out from one to about seven runs of LL hooks. As a cause of potential bias or high variation in the effort unit of fisher-days, a daily trip limit in 2016 in the SE Region has meant that the number of sets per day sometimes fell well under the previous (or daily practical) maximum, meaning half or fewer LL sets were run in some fisher-days. Using hooks in the LL CPUE index as the measure of fishing effort accounts for that variation.

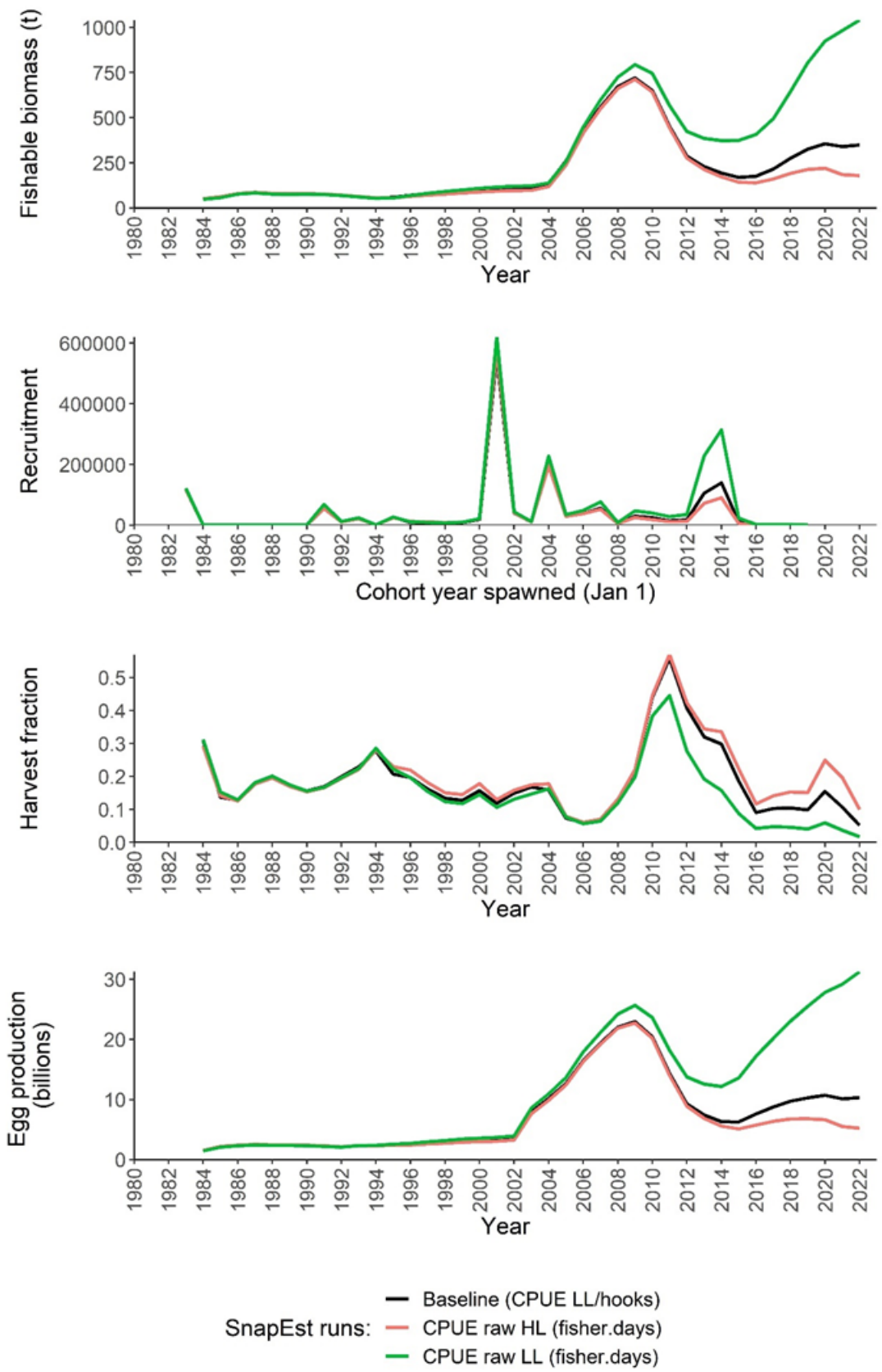

Figure 9-5.6. Sensitivity test for the SE Region, comparing baseline to sensitivity runs testing different choices for commercial CPUE as the fitted index of abundance. The baseline used LL kg.hooks^{-1} for 2003+.

9.6. SnapEst model fits to data

9.6.1. Fits to age proportions

The following plots show the SnapEst model fits (bold line) of age proportions to sample data (thin line) for the half-yearly model timesteps (Sum – summer, October to March; Win – winter, April to September) for each stock. Data were separated by gear type.

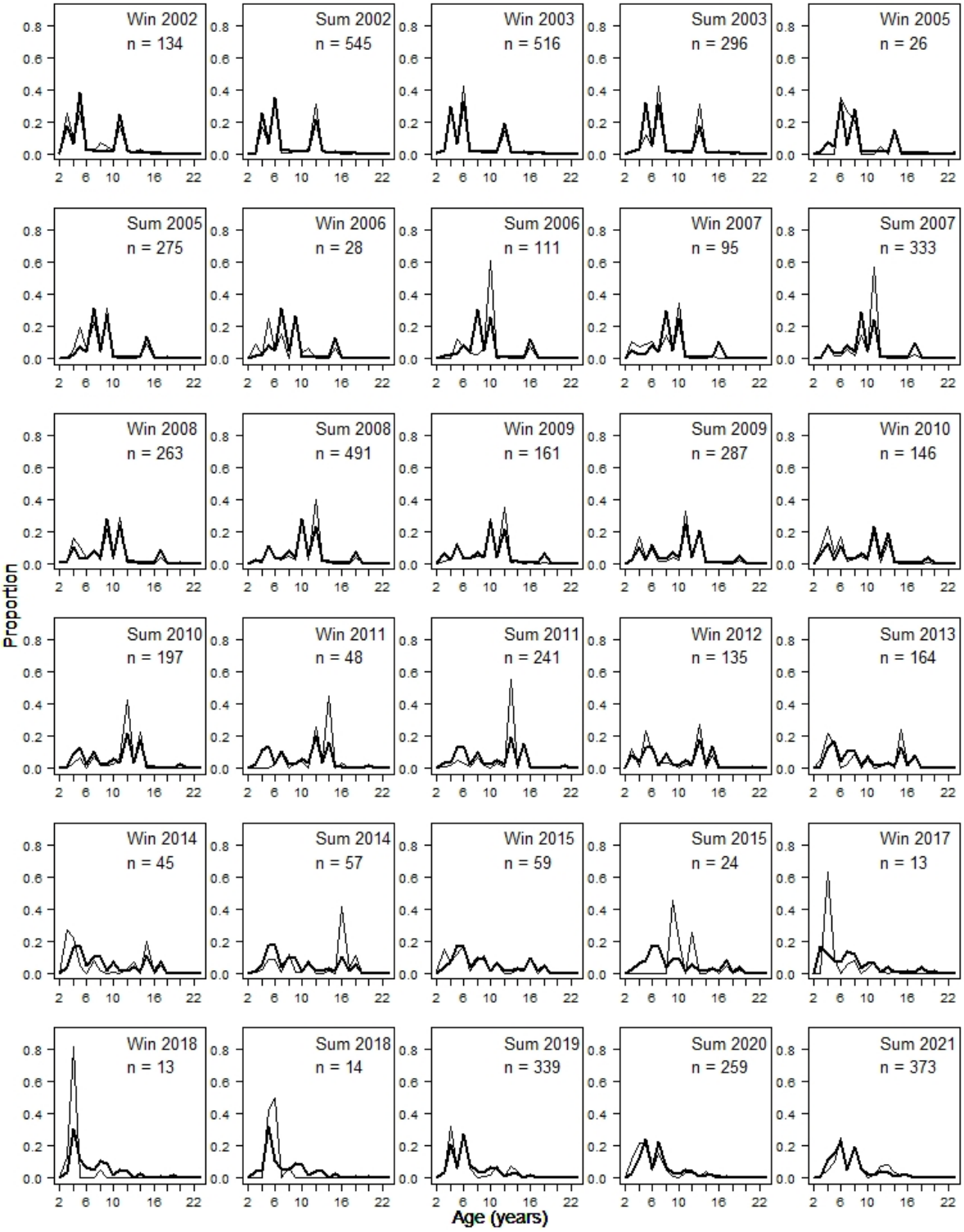

Figure 9-6.1. Model fits to age proportions for the Spencer Gulf/West Coast stock for handline.

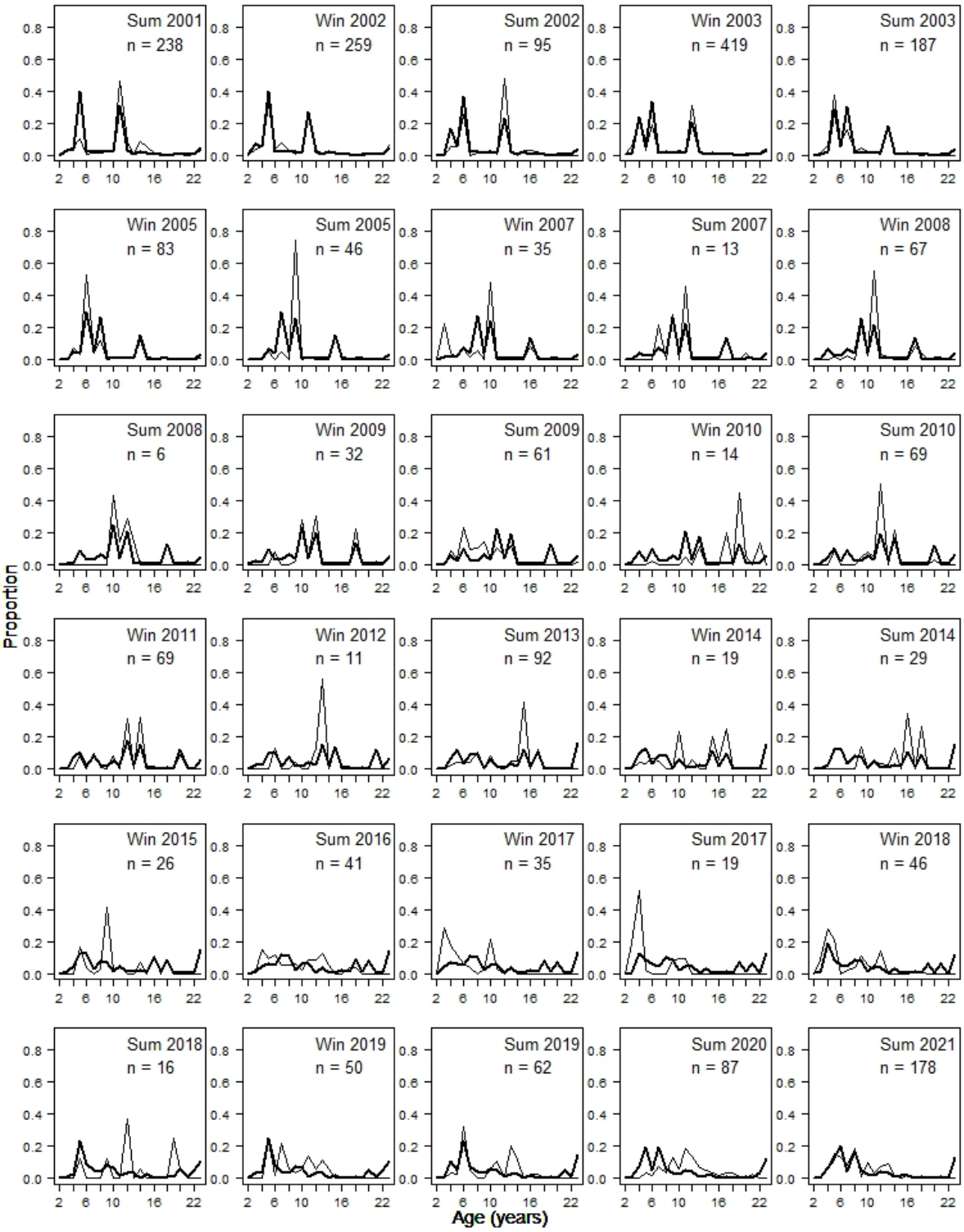

Figure 9-6.2. Model fits to age proportions for the Spencer Gulf/West Coast stock for longline. Sum – summer, Win – winter. Bold line – SnapEst model prediction, thin line – data.

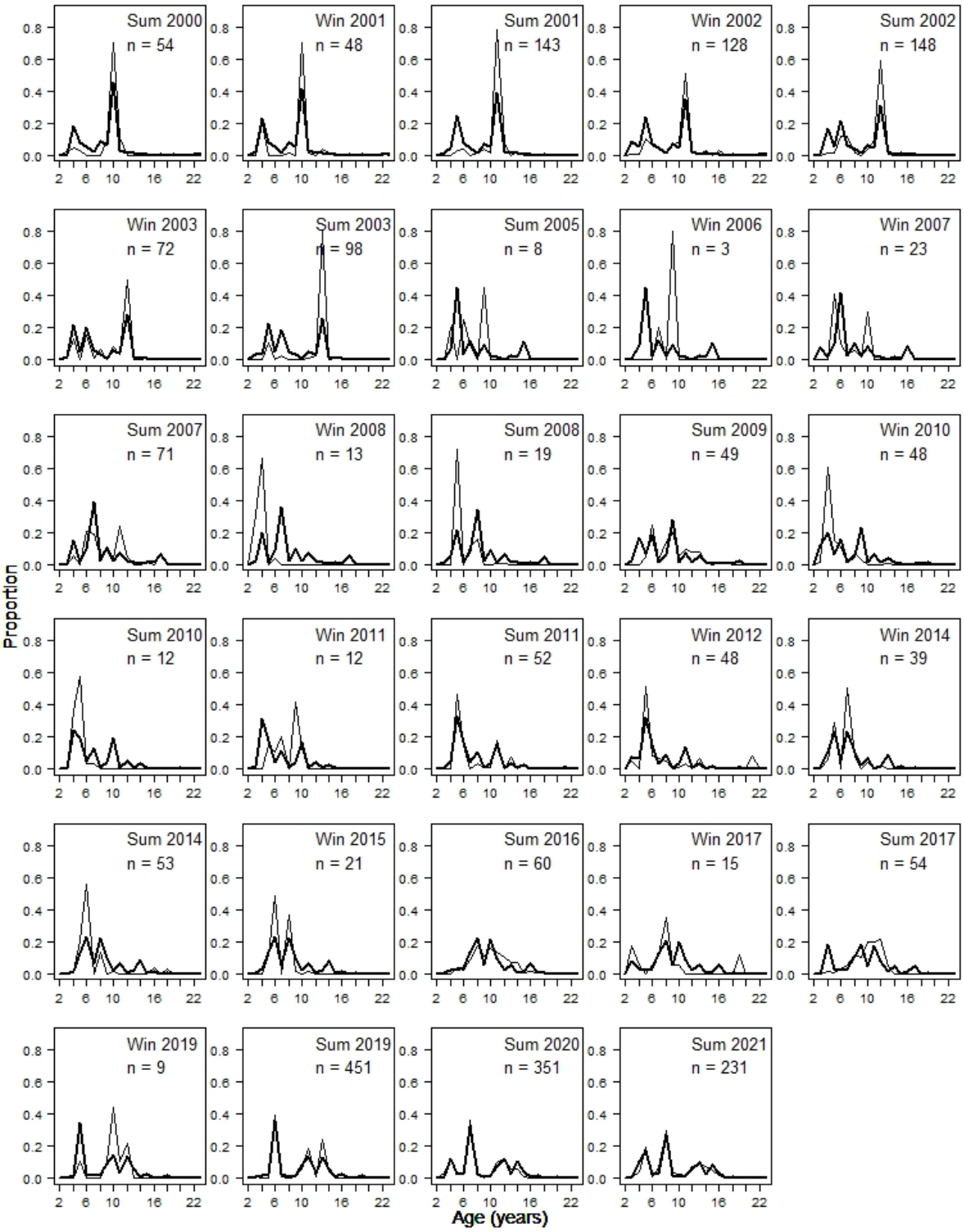

Figure 9-6.3. Model fits to age proportions for the Gulf St Vincent stock for handline. Sum – summer, Win – winter. Bold line – SnapEst model prediction, thin line – data.

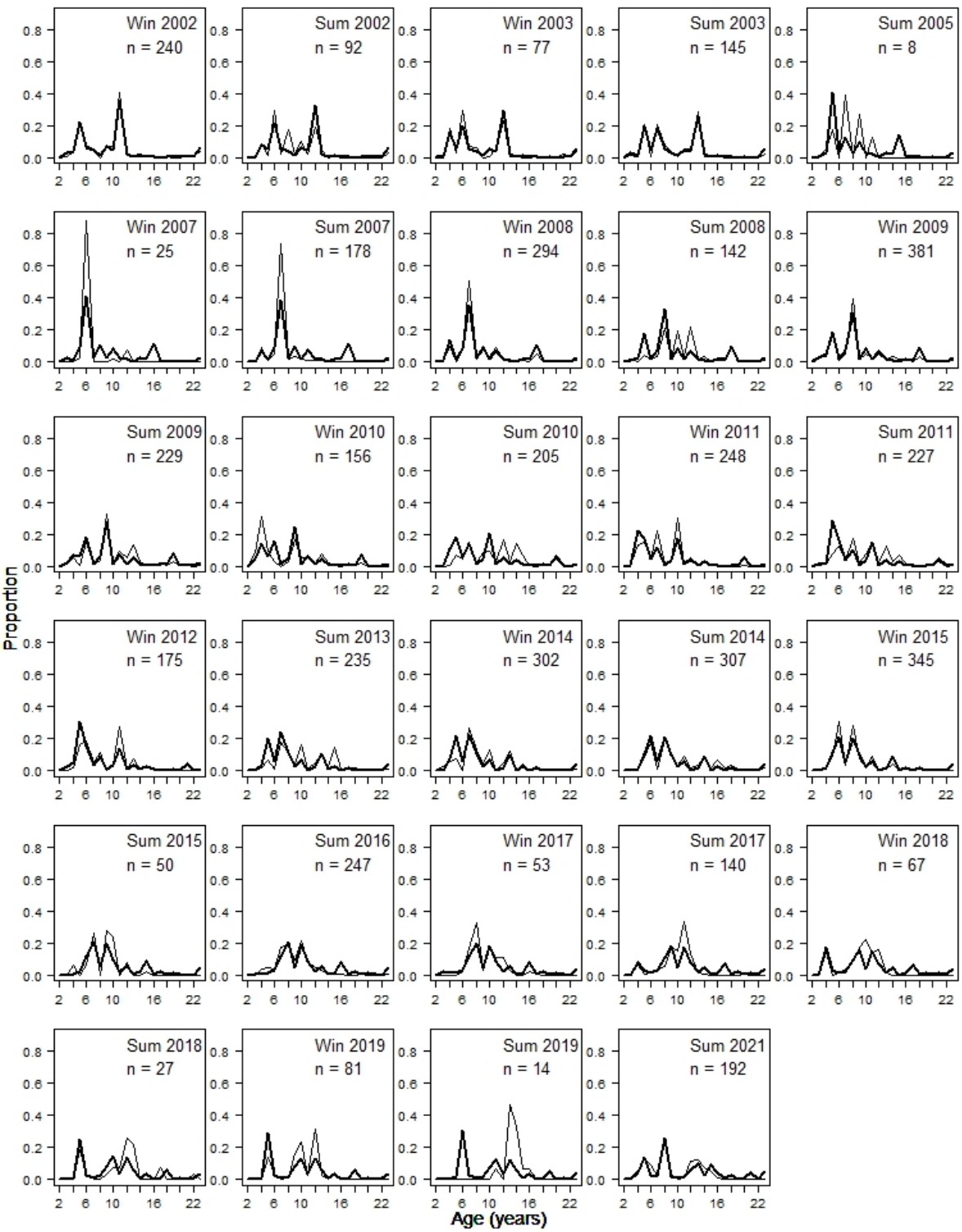

Figure 9-6.4. Model fits to age proportions for the Gulf St Vincent stock for longline. Sum – summer, Win – winter. Bold line – SnapEst model prediction, thin line – data.

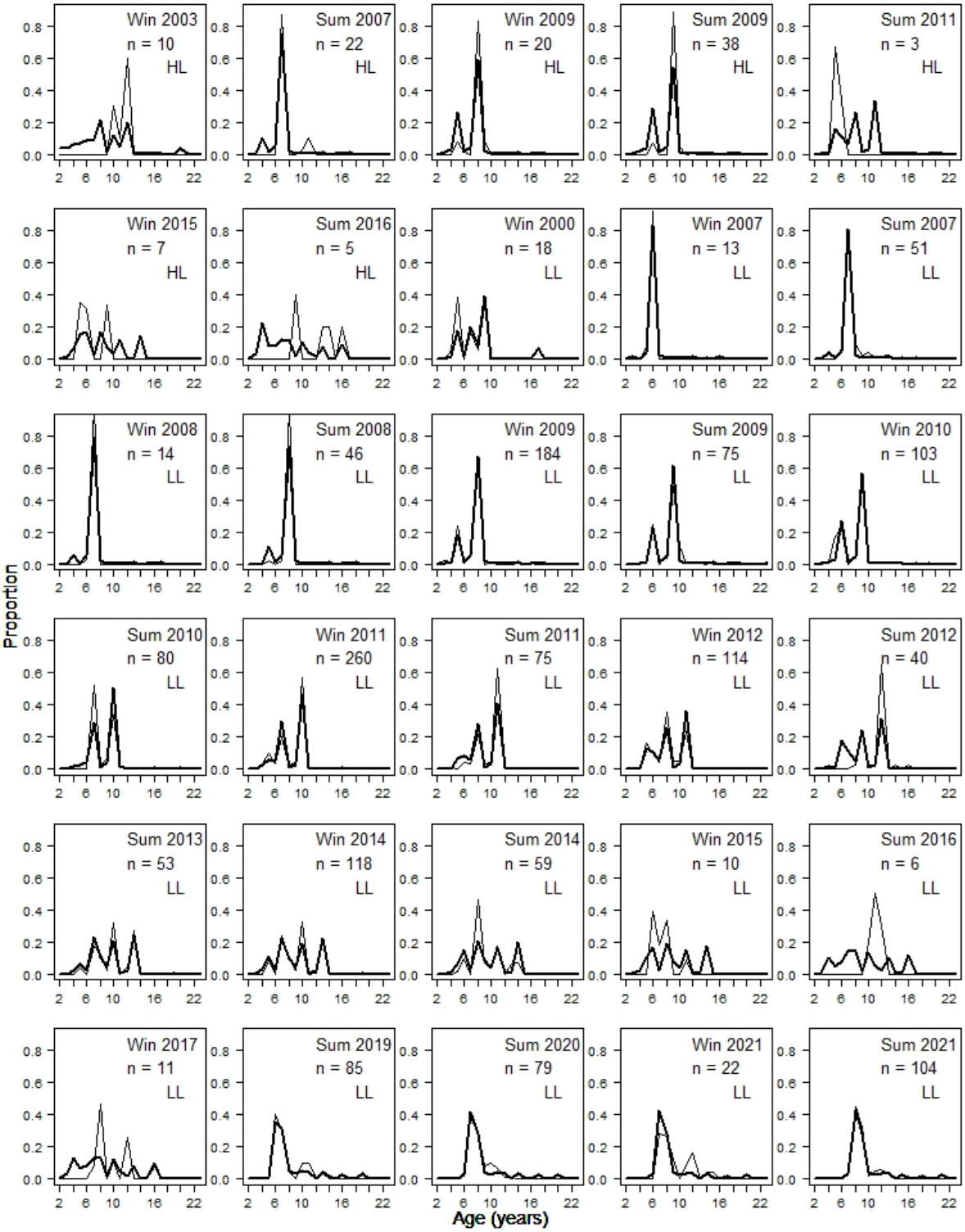

Figure 9-6.5. Model fits to age proportions for the SE Region for handline (HL) and longline (LL). Sum – summer, Win – winter. Bold line – SnapEst model prediction, thin line – data.

9.6.2. Fits to mean length-at-age

The following plots show the SnapEst model fits of model-predicted mean lengths of Snapper versus age for the half-yearly model timesteps (Sum – summer, October to March; Win – winter, April to September) for each stock. Data were separated by gear type. Solid markers are model-predicted values and open markers are data.

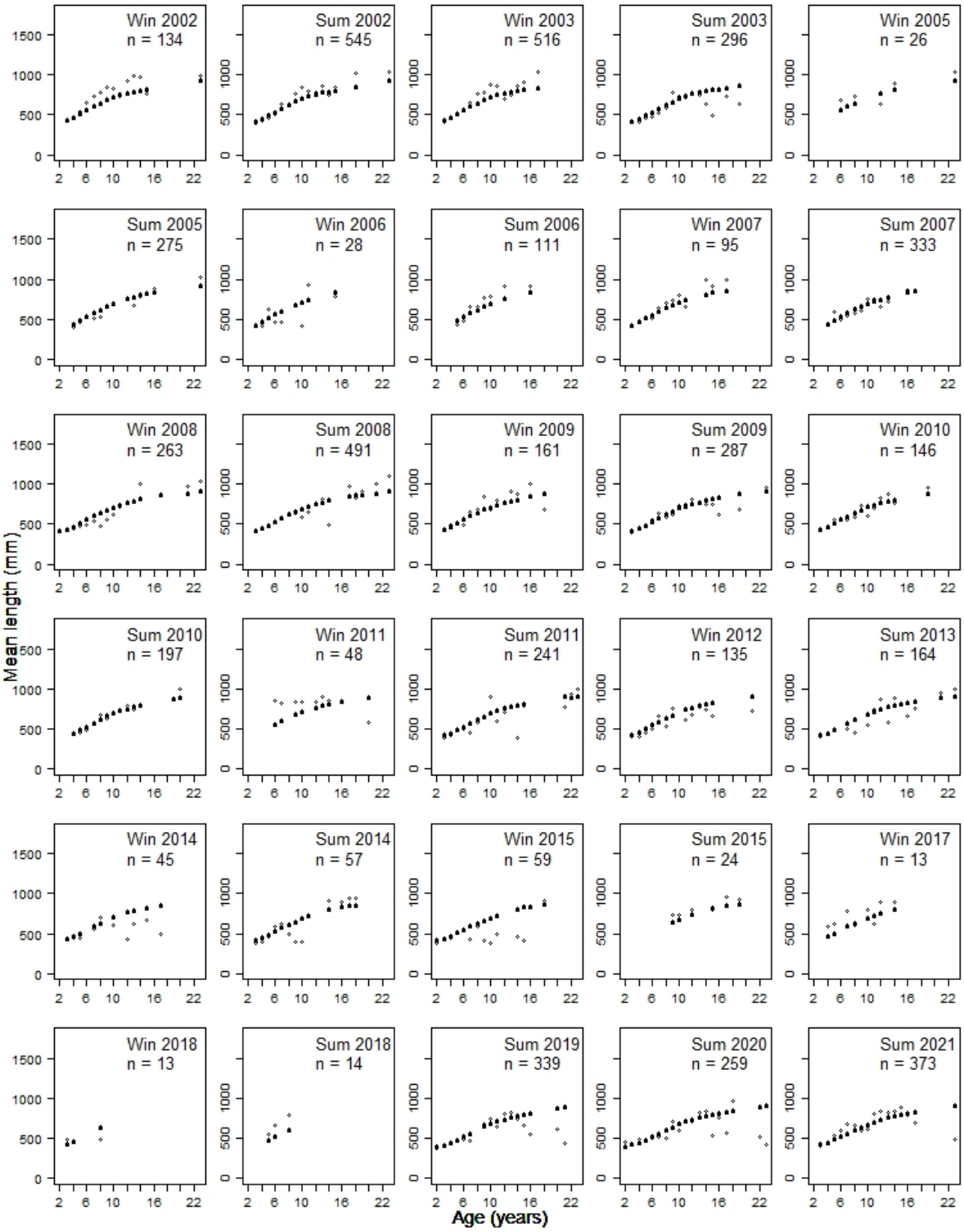

Figure 9-6.6. Model fits to mean length-at-age for the Spencer Gulf/West Coast stock for handline.

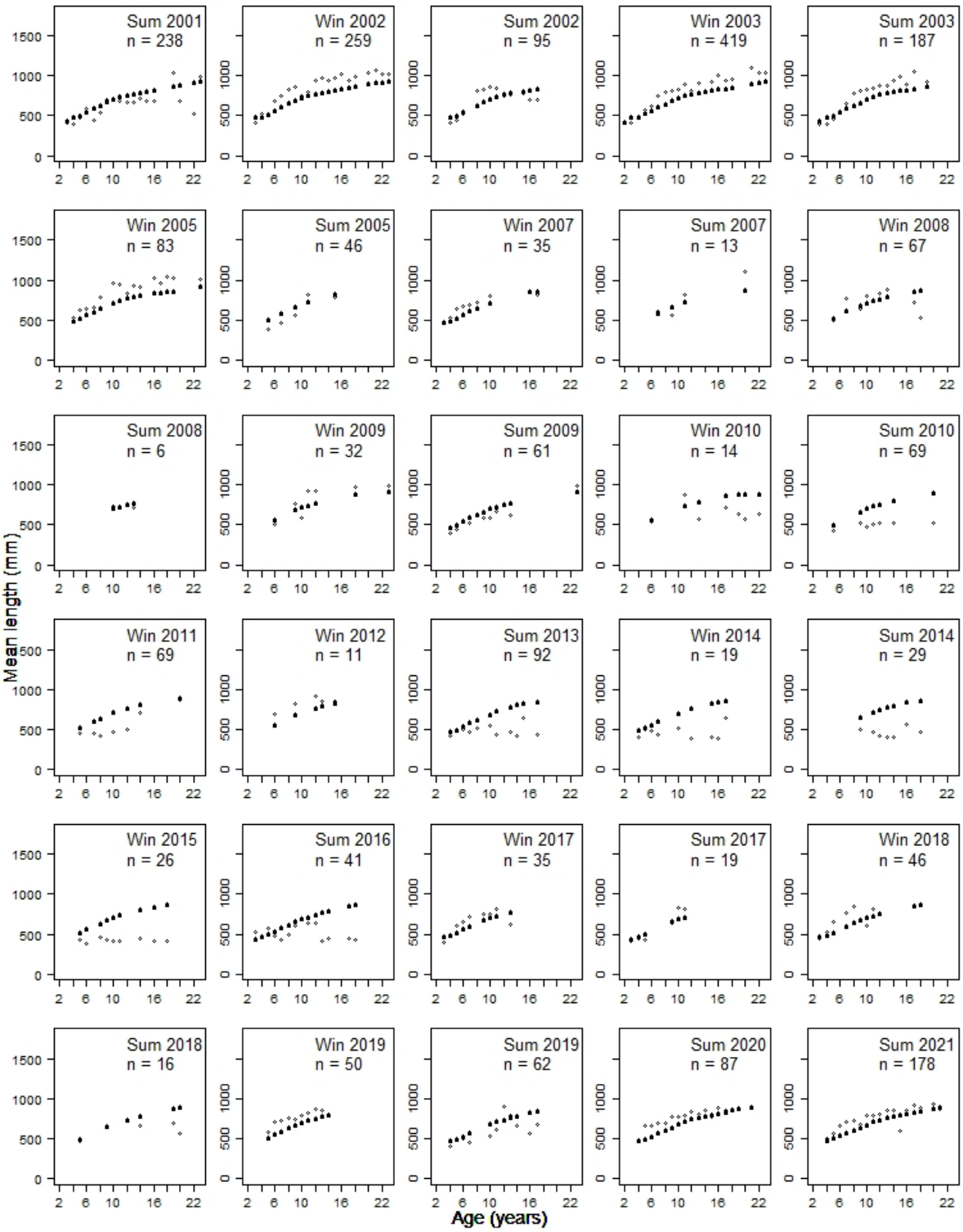

Figure 9-6.7. Model fits to mean length-at-age for the Spencer Gulf/West Coast stock for longline. Sum – summer, Win – winter. Closed markers – SnapEst model prediction, open markers – data.

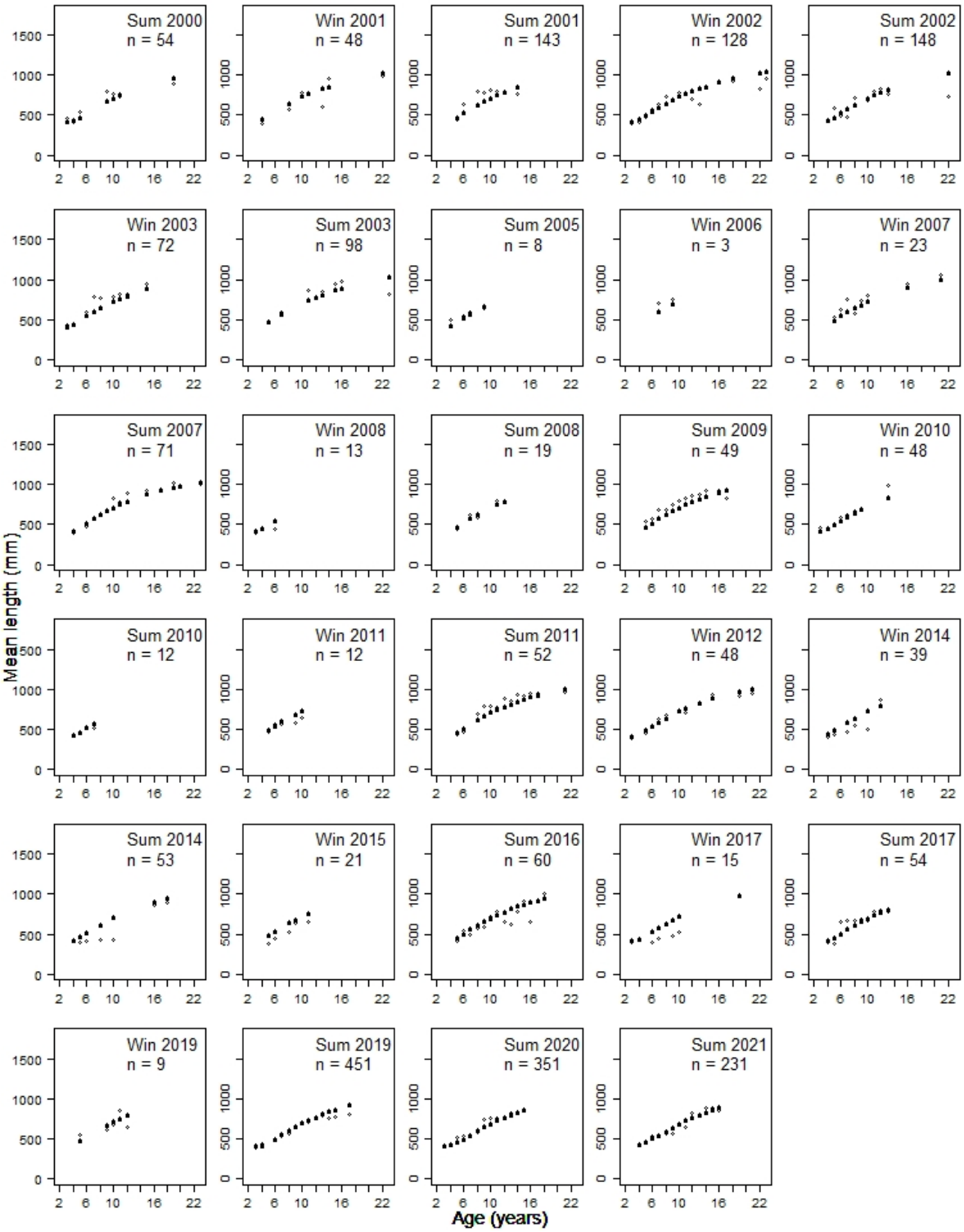

Figure 9-6.8. Model fits to mean length-at-age for the Gulf St Vincent stock for handline. Sum – summer, Win – winter. Closed markers – SnapEst model prediction, open markers – data.

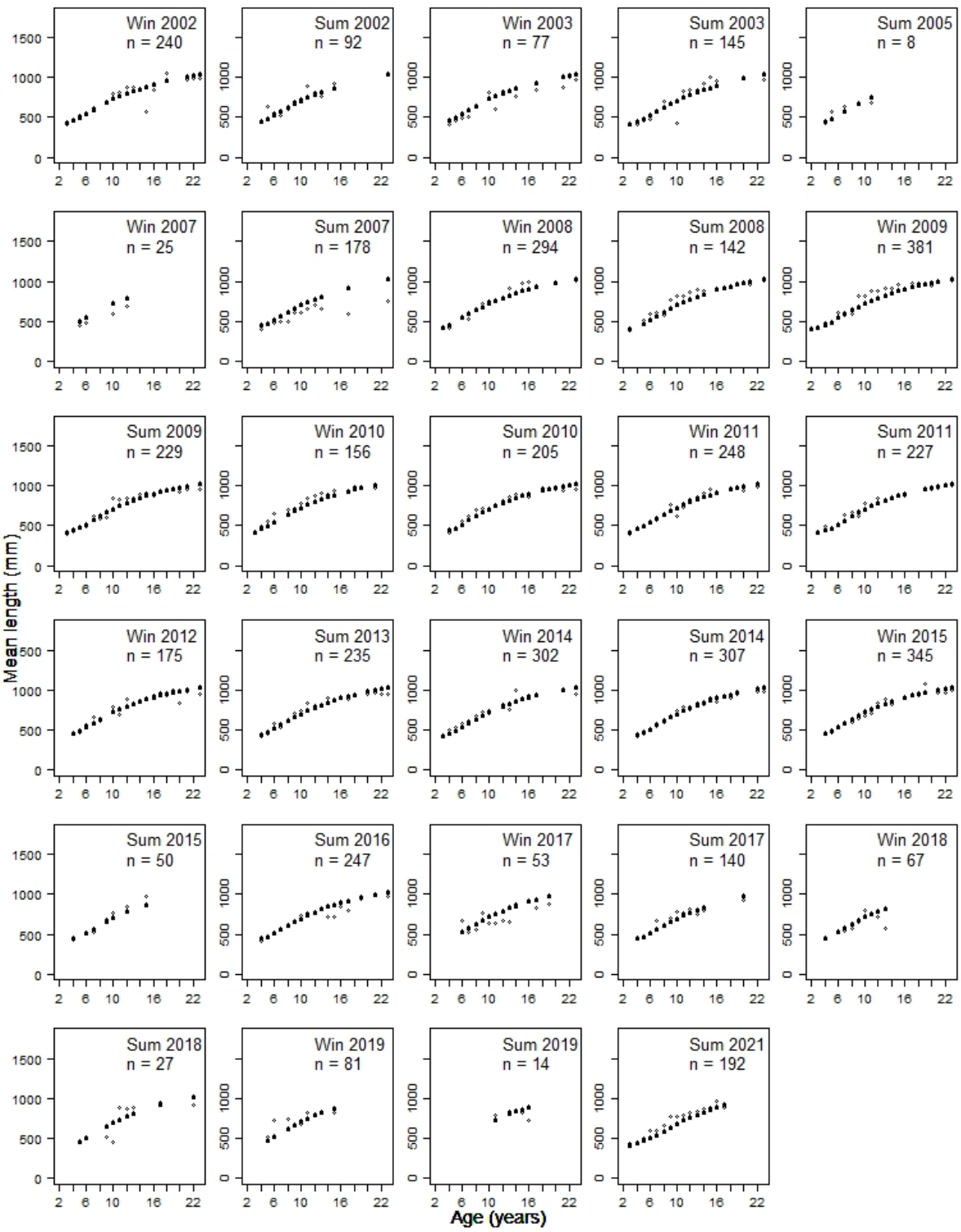

Figure 9-6.9. Model fits to mean length-at-age for the Gulf St Vincent stock for longline. Sum – summer, Win – winter. Closed markers – SnapEst model prediction, open markers – data.

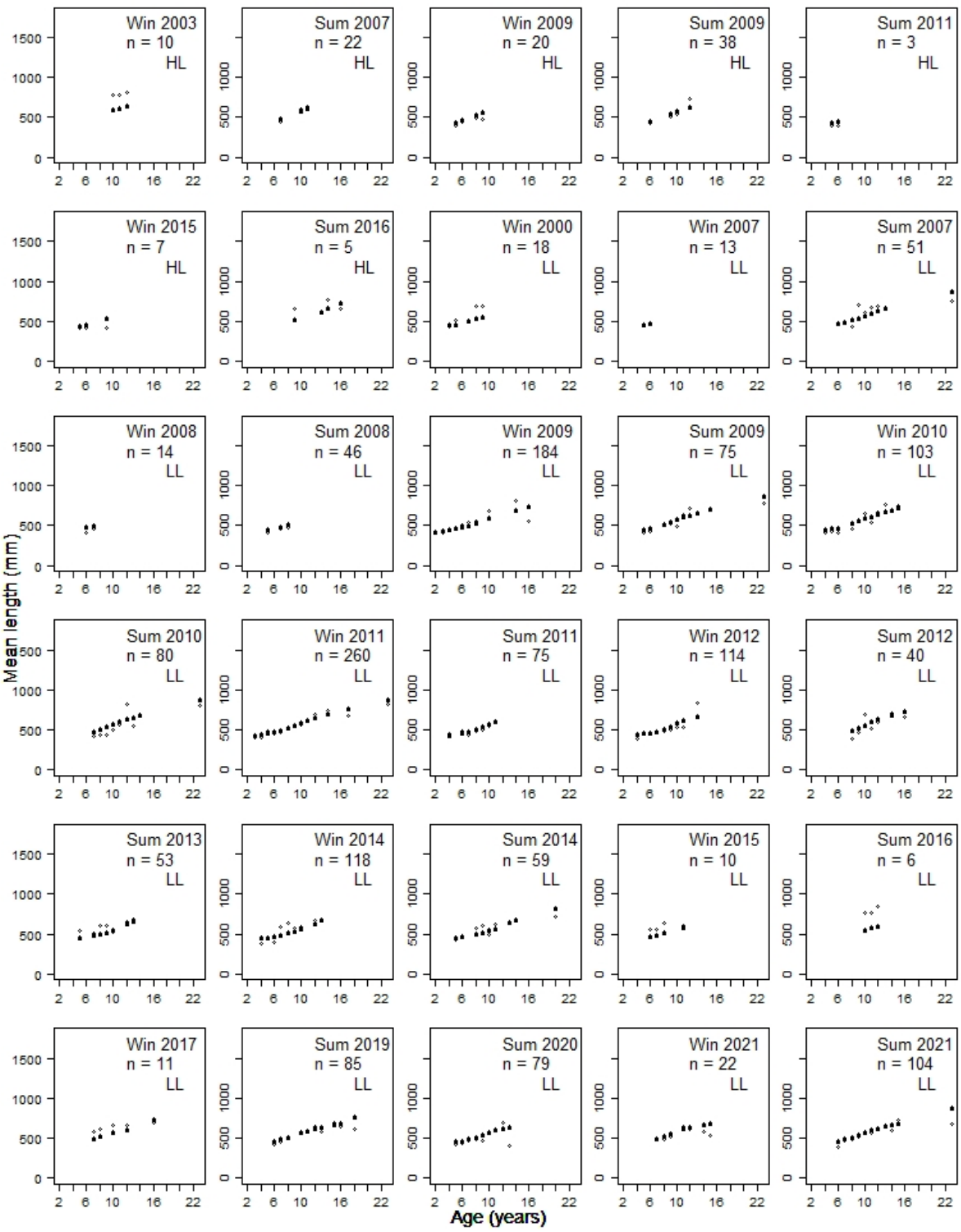

Figure 9-6.10. Model fits to mean length-at-age for the SE Region for handline (HL) and longline (LL). Sum – summer, Win – winter. Closed markers – SnapEst model prediction, open markers – data.

9.6.3. Fit to catch rate

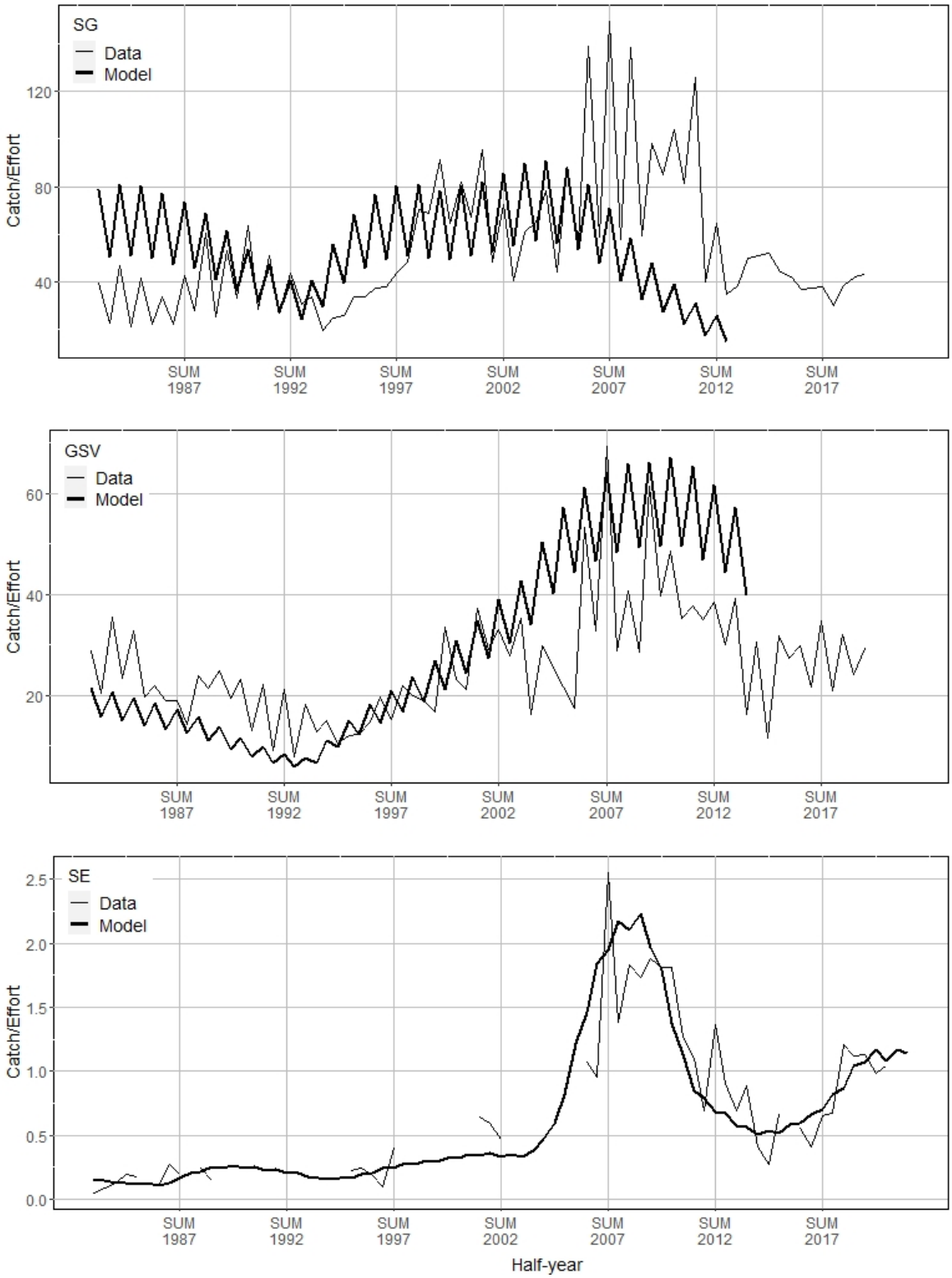

Figure 9-6.11. Model fits to catch rate (CPUE) data. For the Spencer Gulf/West Coast stock (top) and Gulf St Vincent stock (middle), the later years of CPUE (handline, kgs/fisherdays) have been omitted since they are not used in the model. For SE Region (bottom), missing data (longline, kgs/fisherdays from 1984 to 2003. and kg/hook from 2003 to 2022, rescaled values) are for years where the confidentiality requirement (<5 licences) was not met.

9.6.4. Fit to DEPM

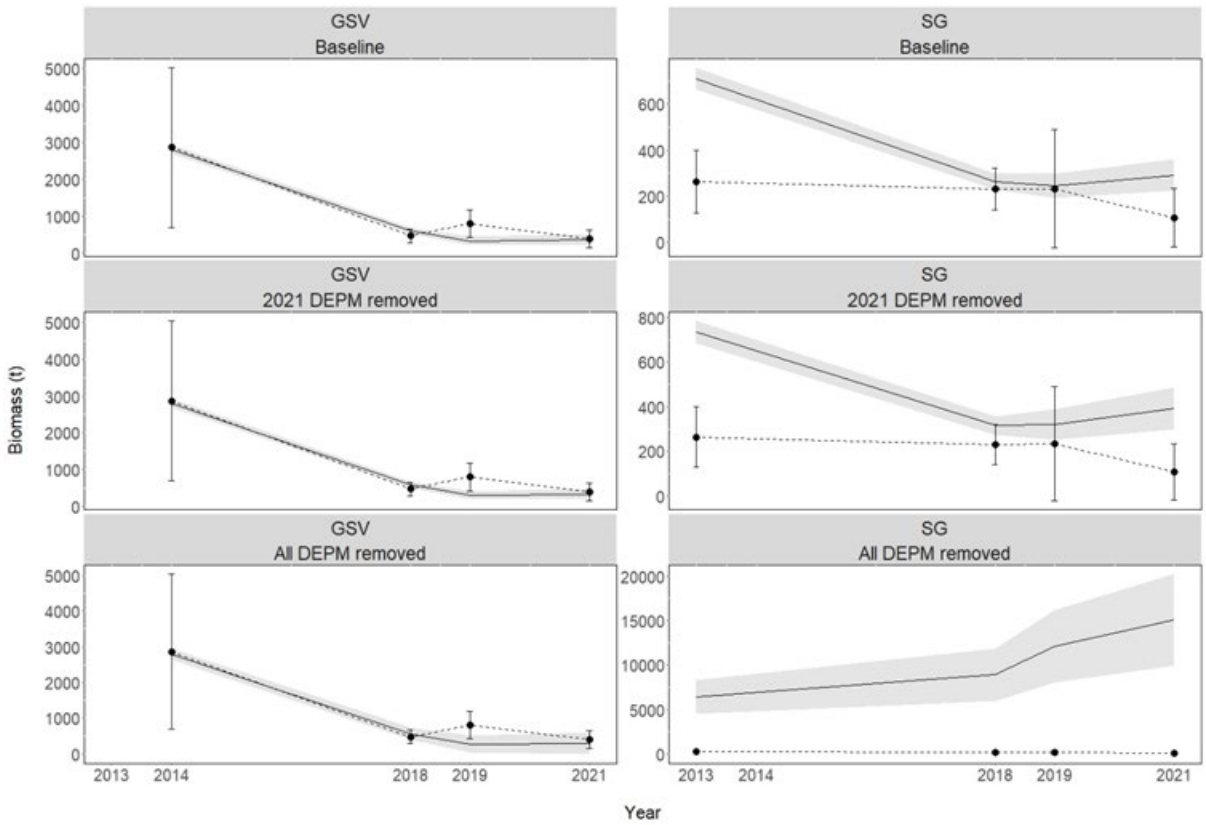

Figure 9-6.12. Model fits to the estimates of spawning biomass using the DEPM for Spencer Gulf (SG) (right) and Gulf St Vincent (GSV) (left). The model biomass estimates (solid lines; grey shading indicates 95% confidence intervals, CI) have been rescaled to include only the areas of SG and GSV covered by DEPM surveys (dotted lines; error bars are 95% CI). The fits of model to DEPM are shown for three sensitivity scenarios: (1) baseline model (all years of DEPM fitted, top), (2) model with 2021 DEPM omitted (middle), and (3) model with all estimates of spawning biomass omitted from model fitting (bottom). The same estimates of spawning biomass for GSV and for SG and their errors are shown for all three graphs.

9.7. Reports by external reviewers

9.7.1. Reviewer 1

Review of the South Australian Snapper Assessment 2022

Tony Smith

Independent fishery scientist

18 October 2022

Terms of reference

No terms of reference for the review were provided but I was asked by Dr Stephen Mayfield from SARDI to provide an independent scientific review of the 2022 SARDI stock assessment report for Snapper.

Process for the review

The draft SARDI report (Snapper (Chrysophrys auratus) Stock assessment Report 2022, by M.J. Drew et al) was provided on 14 October 2022, with the review to be provided by 18 October. I have previously reviewed SARDI Snapper stock assessment reports in 2019 and 2020. Those previous reviews involved providing SARDI with a draft report to which they could respond, with their responses included and considered in my final report. The short time frame for this 2022 review has not allowed sufficient time for this two-stage process.

Main findings of this review

1. The 2022 Snapper assessment finds the Spencer Gulf/West Coast (SG/WC) and Gulf St Vincent (GSV) stocks to be (highly) depleted. This conclusion is very well supported by the data and analyses presented, notwithstanding any of the more detailed comments and issues raised in this review.
2. The status of the SE region is harder to determine given that it is part of a larger Victorian stock and there has been no analysis of fishery data from Victoria. However, the analysis supports the conclusion that its status is considerably healthier than the other two SA stocks.
3. My main concern with the SnapEst analysis is the lack of any figures showing fits of the model to the data. This is a standard requirement for any good quantitative assessment. Its absence should be rectified in this 2022 assessment report.
4. The assessment seems to assume that the “total Snapper fishing closure” for the two main stocks has resulted in zero fishing mortality. This seems unlikely but if it is the case, this assumption needs clear justification. For such highly depleted stocks, even low levels of incidental mortality could be consequential.
5. The report sets great store by the similarities in model output for the GSV stock with or without use of the DEPM survey estimates. Although factually correct, I find its emphasis

and repetition misleading and potentially open to misinterpretation (Why do we need the DEPM surveys then?).

6. The report is somewhat inconsistent concerning the relationship between stock and recruitment. Clearly, recruitment is highly variable and episodic for these stocks and not obviously related to stock abundance, but the current extremely low levels of egg production should clearly be of great concern for the prospects of stock recovery.
7. There are multiple problems with the current suite of performance indicators for the fishery. Not least among them is the trigger reference level for harvest fraction.

Further explanation for these findings may be found in the next section covering specific comments on the report.

Specific comments on the report

These more or less follow the order found in the report and are not in order of significance. Where relevant, I cite page numbers and some text from the Draft Report.

1. Page 10. "previous management regulations were superseded by a total Snapper fishing closure that was imposed for the waters of the WC, SG, and GSV from 1 November 2019 to 31 January 2023". I know that these regulations apply to commercial fishing by the MSF. How have they affected other fisheries such as for Rock Lobster? And what impact have they had on recreational and charter fishing? Even for the MSF, have they affected targeted fishing for other stocks such as whiting and garfish? (This question goes to later questions about residual levels of Snapper catch despite closures).
2. Page 14. "This assessment is the first in the series of Snapper assessments to have data relating to recreational Snapper catches from mandatory reporting, which have been included in this Chapter and integrated into the SnapEst model". Related to point 1, this suggests some ongoing recreational catch of Snapper. The SnapEst chapter does not actually list estimated catch levels since the closures and should do so.

   Page 15. "Data from the Charter Boat sector were obtained from the MSFIS. Daily records of the number and estimated weight of retained Snapper were analysed for the calendar years 2020 and 2021". Does this only refer to SE catches?
3. Page 16. "In 2020 and 2021, catch declined to record low levels with all landings coming from the SE Region". This implies a zero catch of Snapper for SG/WC and GSV. Is this really the case? No bycatch at all from other target fishing? No recreational take? No discard mortality? Even very low levels of incidental fishing mortality can be significant for highly depleted stocks. See also comment 29d.
4. Page 27. "The recent introduction of a Total Allowable Recreational Catch (TARC) and compulsory catch reporting in the SE Region has provided the most comprehensive understanding of recreational Snapper fishing to date". It is heartening to see this improvement in recreational catch reporting. Does this apply in principle to the other stocks? And to any other species? Uncertainty about recreational catches has been an important source of uncertainty for the overall assessment of stock status.
5. Page 29. It is good to see this targeted adult sampling program in place during the closures. Table 3.1 is useful, but it would also be very helpful to extend it backwards in time to have a complete record of sample sizes for length for all the data used in the assessment (I realise that these are listed in some figures but having them all together in one table would help

address some later questions). Sample sizes for age over time should also be listed in a Table.

6. Chapter 4. I am not an expert on DEPM methods, so I have not really reviewed this section in any detail. I understand and acknowledge the importance of the data and estimates from these surveys to the overall assessment of stock status for the two gulf stocks, and generally commend the thought and effort that is going into improving these methods. I do have some later comments about interpreting the uncertainty in the DEPM estimates, in the context of fitting them in the SnapEst model.
7. Page 74. "The estimated spawning biomass of Snapper in each gulf has differed between surveys. It is important to recognise that each estimate of spawning biomass relates to the area covered by each survey, and therefore it is difficult to directly compare between years when the survey areas were different (i.e., 2013 and 2018). The differences in survey design are accounted for in the estimates of fishable biomass from the SnapEst model." There seem to have been a number of differences between surveys over time in each gulf. Given that the biomass estimates from these surveys are used as absolute estimates of biomass, it would seem important to take account of these differences, and also variations in survey design and circumstances year to year. The last sentence quoted above suggests that this is accounted for in the SnapEst model. The approach is discussed in comment 15 below. Another (less direct) way to reflect this variation would be to increase the variance on the individual DEPM estimates that are used in the stock assessment (something I also suggested in my last review).
8. Page 77. "There was a 38% decrease in spawning biomass when the adult parameters were re-estimated after the removal of these fish. Nevertheless, there was a negligible (<1%) difference in the subsequent estimate of fishable biomass from SnapEst (Appendix 9.5.3)." Here and in later comments I caution against over-interpreting this sort of information. Making a substantial change to one data point in a stock assessment will often not result in a lot of change in an output statistic because of the limited weight of that point relative to all the others informing the assessment. If such points are to be made, you really need to show the difference in the fit of the model to the alternative data points, as argued more generally in comment 10 and later.
9. Chapter 5. First, a general comment on this chapter and the accompanying appendices. Unlike the previous (2020) assessment that I reviewed, nowhere in this report are any fits of the stock assessment model to the data shown. This is a major hindrance to any serious attempt to understand the assessment and should be rectified (in the appendix). More specific suggestions are provided below.
10. Another general comment about chapter 5 is that some of the data should be shown more explicitly in tables or figures. I am thinking particularly of the catch data, and for example how the periods between and after recreational catch series have been interpolated, and what has been assumed (if anything) about catch levels post fishery closures.
11. P 80. "The model does not attempt to discern a stock-recruitment relationship and the yearly recruitment number for each stock was freely estimated." This is a reasonable approach given the highly variable and episodic recruitment patterns. However, it does raise questions later about what it means to overfish such stocks, if there is no impact of depleting the stock on future recruitment. Later, the report seems to assume that there is some sort of stock recruitment relationship: "More recent years are excluded because they

potentially reflect recruitment reduced by high exploitation and so may not be typical of pre-fishing levels" (p81). See also comment 27.

12. Page 80. "Instantaneous natural mortality was set to M = 0.05 (consistent with previous versions of Snapper stock assessment models in South Australia)." This is consistent with fish living to a considerable age, and the implication of the sentence is that this value for M has been accepted for some time. However, this value of M seems completely inconsistent with the later information (Table 6-1, page 94) that the trigger reference point for the harvest fraction is "above 32% (int. standard)". I strongly suspect that the value for M is correct and that the harvest fraction (even as a trigger) is way too high. Presumably the "int. standard" refers to some level accepted in a scientific publication but how old is it and was the maximum age of snapper well understood when it was adopted? Was it for snapper in general or this species in particular? A suitable target for harvest fraction should be reviewed and is more likely on the order of 5 to 10%. I return to this point in commenting on the output statistics for the model.

13. Page 81. "Egg production is computed as the estimated total number of eggs produced in each summer spawning season, assuming a 50:50 sex ratio, a fecundity-versus-length formula and all Snapper aged 2 years and older are mature." Is it really the case that all Snapper aged 2 years and older are mature? This helps explain why there is little difference in trends between exploitable biomass and egg production.

14. Page 82. "In fitting to DEPM spawning biomass estimates, it was necessary to account for the fact that the model regions sometimes extended beyond the area covered by the DEPM surveys. In 2019 and 2021, the survey area covered about half of the SG/WCS. To extrapolate beyond the SG/WC survey area, commercial catch by MFA block was used. Model biomass for the whole region was scaled downward before fitting to approximate the spawning biomass estimate 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 surveys were used. For fitting to the results from the surveys in 2019/20 and 2021/22, when the two gulfs were closed to fishing from November 2019, the 2018/19 spatial catches by MFA block (Oct-Mar) were also used to construct the 2019/20 survey and 2021/22 survey spatial catch proportions. Because the GSV region was fully covered by the 2019/20 and 2021/22 surveys, this extrapolation using catch by MFA was not needed in GSV for those years." I don't understand what is being adjusted here. Does this explain the earlier reference to adjustments for survey design (comment 8 above)? What is the extent of the adjustments? This all needs to be explained (and justified) more clearly.

15. Results for the SG/WC Stock.

    a. Trends in biomass. P 83. "a flat to marginal increase in fishable biomass over the last two years". This is despite the considerable <u>decrease</u> in the DEPM biomass estimate from 2020 to 2021. It would be very useful to see a plot of this recent biomass trajectory (say, post 2012) showing the fit to the DEPM estimates, and with the CIs shown for both the DEPM values and the model-estimated values for biomass. This would provide some basis for assessing whether the model is overconfident in its estimates of uncertainty.

    b. Trends in recruitment, Figure 5-1. This figure clearly shows just how much the fishery for this stock relies on episodic recruitment, and how poor that recruitment has been for two decades. Regarding the impact of depleting the stock on

subsequent recruitment, comparing the biomass and recruitment trajectories shows pretty clearly that the dramatic decline in recruitment occurred while the stock was still at relatively high biomass levels (though arguably low egg production – see below). It might at least be noted that the only years of high recruitment have come from years of higher biomass, but high biomass is clearly no guarantee of good recruitment. Some discussion on stock and recruitment and its implications for stock recovery should be included.

c. Trends in harvest fraction. P 83 and Figure 5-1. "From 1984 to 2014, estimates of harvest fraction generally varied between 0.1 and 0.2. These increased from 2014 and reached 0.45 in 2019 (Figure 5-1)." See comment 13 above on appropriate levels for the harvest fraction. Arguably (at least in retrospect) this stock has been overfished for decades! This is consistent with the next point below.

d. Trends in egg production. P 83 and Figure 5-1. "The trends in model-estimated egg production over time largely mirror those of fishable biomass, showing a steep and large decline from 2005 onward." This comment fails to note that egg production has been below its trigger (limit?) reference point for the whole period since 1984! The fact that it is now estimated to be at 2% of unfished levels puts this stock (and GSV) in a league of its own with respect to stocks that I have reviewed (and there have been a lot of them)!

e. Summarising points a to d here, while the assessment report provides the graphs and describes the results, in my view it considerably underplays the gravity of the situation.

16. Page 84, Figure 5-1. Following my comment in the 2020 review, I continue to be dubious about just how tight the CIs are on all these graphs. Showing the fits to the data would help clarify this comment and concern.

17. Results for the GSV Stock.

    a. Many of the comments for SG/WC can be replicated for GSV. Figure 5-2 shows that this stock has benefited from both more frequent and more recent episodes of strong recruitment (strong year classes) than SG/WC. The relationship between stock size and strong recruitment is equally tenuous. If R is recruitment and B is biomass, it seems that R can drive B, but B does not seem to drive R. The longer-term values for harvest fraction are lower than for SG/WC and closer to appropriate values, but the more recent history is even more calamitous with harvest fraction peaking at over 70%. The current state of egg production is nearly identical to SG/WC and was only ever briefly above the trigger level (as far back as records are available).

18. Results for the SE Region.

    a. The report implies but should emphasise that this is an analysis of the SA component only of a wider stock. The use of SnapEst to assess only part of the stock, ignoring catches and data from Victoria, should be justified more clearly. The report does note that the SE region is likely a sink population for the wider stock but also notes the possibility of some local spawning. It is not clear what the results for egg production mean for only part of the overall stock.

19. Page 90. I have concerns about possible misinterpretation of some of the statements in the second paragraph on page 90. While it correctly points out the differences between SG/WC and GSV found in the sensitivity tests involving exclusion of DEPM estimates, it potentially goes too far in arguing that the similarity in outputs for GSV is of great significance. The report argues that "For the GSVS, even complete exclusion of all estimates of spawning biomass from the DEPM reflect similar model outcomes to when DEPM is included. <u>This implies that the standard datasets available for the GSVS, particularly the extensive long-term age structures, were sufficient to inform model estimates without a fishery-independent measure of biomass</u>" and goes on to argue that "The close agreement between estimates of fishable biomass from SnapEst and estimates of spawning biomass from the DEPM, both in absolute biomass and in the trend from 2014 to 2019, provides mutual validation of this large biomass decline". In my view, also expressed in my 2020 review, this correspondence is fortuitous, as the wide discrepancy for SG/WC illustrates. The report notes the "extensive long term age structures" for GSV, but these are also available for SG/WC where there is no correspondence. The fact is that in the absence of reliable CPUE, the ongoing DEPM surveys are critical to anchoring the assessment, the fortuitous correspondence in results without the surveys for GSV notwithstanding. I don't think that the authors disagree with this point, but the text underlined might lead some readers to think otherwise. Perhaps a final sentence emphasising the fundamental importance of the DEPM surveys would not go amiss?

20. Page 94, Table 6-1. I realise why this table has to be here – it is in the management plan. And I also understand why the management plan, which was due for revision in 2018, has not been updated – the MSF restructure. The Table is a great illustration of what is wrong with the current management plan and (perhaps) why things have gone so badly for two of SA's Snapper stocks. I can't help passing a few comments:

    a. There are far too many performance indicators and their potential value in informing an effective harvest strategy varies enormously. The combination of positive and negative PIs is also confusing. This raises the risk that the significance of a few key indicators will be swamped by the noise from a large number of uninformative indicators.

    b. The fact that there are only trigger reference points and not target or limit reference points is also a problem. This is particularly a problem where the triggers relate to relative changes in indicators over a period of time, rather than clearly specifying targets and limits.

    c. I mention again the highly dubious trigger for harvest fraction. The level indicated (32%) seems way too high even for a limit reference point. The fact that it was clearly breached by 2015 in GSV and perhaps a year later in SG/WC without immediate consequences perhaps illustrates the concern about point a above.

    d. The choice of primary and secondary indicators is also far from clear. For example, egg production would be a primary indicator in most assessments but is here relegated to secondary. The fact that the trigger has been breached in every single year since 1984 in SG/WC and SE and in nearly every year in GSV, without any apparent consequences, is also of great concern. Would having egg production as a primary indicator have made any difference?

   e. Just how potentially uninformative this "dashboard" approach to indicators can be is illustrated in the results shown in Table 6-3 on page 97. All three stock show a mix of greens and reds, but two are severely depleted and one is classified as sustainable.

21. Page 101. Overall, this summary of the results for the SG/WC stock mentions all the key points. The sentence "Model outputs indicate that the decline in fishable biomass relates to a prolonged period of poor recruitment throughout the 2000s and from 2010, and increasing harvest fractions, caused by the continued fishing of a depleting stock." is a good summary of the situation and its causes. The sentence "It is evident that the biomass and recruitment of the SG/WCS remains at low levels with no evidence of measurable stock recovery following the closure of the fishery" is also useful.

22. Page 101. "The estimate of spawning biomass in 2021 was 108 t (± 65; SE)". "The SnapEst model estimates of fishable biomass declined by 90% from a peak of 5,244 t (± 104; SE) in 2005 to 543 t (± 65; SE) in 2022". There is a relatively small but statistically significant discrepancy between the DEPM estimate and the model estimate of recent biomass. This highlights the need, mentioned elsewhere, to show model fits to the data, particularly the fits to the DEPM estimates. It also potentially sheds light on data weighting.

23. Page 105. "The reliance of DEPM inputs for the SG/WCS is demonstrated by the fits to HL CPUE (Figure 9-5.2), 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 the model inputs (Figure 9-5.5)." This is confusing. I went looking for fits to data in the figures indicated but of course they are not there. It seems to be just another chance to make a point made repeatedly in this report (see comment 20). If the CIs on the results in Figures 9-5.2 and 9-5.5 were shown, it would be easier to interpret the results.

24. Page 105. "SnapEst is primarily reliant on estimates of spawning biomass as its index of relative abundance". The DEPM estimates are fitted as absolute, not relative abundance.

25. Page 105. "Thereafter, the decline in fishable biomass was associated with a prolonged period of poor recruitment, concurrent with the continued exploitation of the population". The latter point, continued exploitation of the population, should be emphasised further. Not only did exploitation continue, but the harvest fraction (exploitation rate) increased dramatically. The report could (should?) also mention that this followed a very long duration of exploitation rates (harvest fractions) that were almost certainly too high for the inherent productivity of the stocks. If lessons are to be learned for future management, this longer history needs to be emphasised.

26. Page 106. "Furthermore, because the two populations are heavily depleted and spawning potentials (i.e., egg production) are at a historically low levels, the probability of a strong recruitment event is accordingly very low". I agree with expressing this concern but see previous comment 12 about stock recruitment relationships.

27. Page 106. "The timeframes for the expected recovery of the SG/WCS and GSVS are unknown". While this statement is true, I think more could be said. Given 1) the absence of any strong incoming year classes and 2) the time it would take for any future strong year class to fully recruit to and influence the fishable biomass, the minimum possible time to recovery would be at least "X" years. You might also discuss what level of recovery would be required to enable any resumption of targeted fishing. I realise that the latter is really a management question, but it can also be informed by science. The larger point here is that

the disruptions caused by having a key Tier 1 stock closed to fishing are likely to endure for some time. It would be remiss not to state this very clearly.

28. Section 7.5 Directions for future research. I have a couple of comments on this section.
    a. On improved monitoring of 0+ juveniles, does this pre-suppose that most density dependent determinants of year class strength occur at early life history stages, rather than say at juvenile stages? This seems to be supported by the PPB results. With stocks at such extremely low levels, does depensation become a possible factor preventing the emergence of stronger year classes and stock recovery?
    b. Page 107. "The estimates of spawning biomass using the DEPM have become an important component of the stock assessment for Snapper in SA". Vital, I would have said! Continuing to understand and improve the basis for these estimates is clearly a high priority. The question of how often the surveys should be conducted is also worth discussing.
    c. Page 108. I strongly support the proposal to develop a forecasting capability for the SnapEst model. This would have several immediate and longer-term benefits, including 1) allowing a proper analysis of suitable reference points for quantities like Harvest Fraction (see comments 13, 16c and 21c); 2) as noted in the assessment report, allowing exploration of possible recovery scenarios; and 3) allowing evaluation of future harvest strategies.
    d. Page 109. "Snapper is highly susceptible to barotrauma through fishing activities". This highlights a point raised earlier about ongoing incidental mortality. Even without the research project being completed, it is not clear to me why the current mortality due to fishing for SG/WC and GSV is (apparently) assumed to be zero.
29. Appendices. Limited time for this review has prevented an in-depth consideration of all the detail in the appendices but several of the sections repeat material that I have considered in earlier reviews. <u>My main concern is the absence of any figures showing fits of the model to the data</u>. Such figures are essential for any well-informed review of a model-based stock assessment. These should be provided for the main assessments for each stock and would also be informative for several of the sensitivity tests undertaken, including variations in use of the DEPM. Fits to both the DEPM estimates and the CPUE should be shown for the main assessments. Fits to the age structure would also be useful, particularly for more recent years.
30. Page 147. "This implies that the standard data set available in Gulf St. Vincent, with strong and long-term age samples, was sufficient to inform model estimates without a fishery-independent measure of stock biomass". See comment 20 above.
31. Page 147. "we added two further runs where the initial parameters were variously modified in both the baseline version of SnapEst and in the run with DEPM entirely omitted". These modifications do not seem to be described anywhere.

9.7.2. Reviewer 2

Review of Snapper (*Chrysophrys auratus*) Stock Assessment Report 2022

Department of Primary Industries and Regional Development (WA)

19 October 2022

Stringent fishery closures for Snapper were implemented from 1st November 2019 for the Spencer Gulf / West Coast Stock (SG/WCS) and Gulf St. Vincent Stock (GSVS). A stock assessment delivered in 2020 determined that the SG/WCS remained 'depleted' and that the status of the GSVS had further deteriorated necessitating its reclassification to 'depleted'.

The current South Australia (SA) assessment report provides a comprehensive account of the 2022 status of the biological stock for each of the SG/WCS and the GSVS and reporting for South-East Region of the State (SE Region). Noting the latter stock forms a component of Victoria's snapper assessment and it is their report where stock status is determined.

Overall, the assessments presented in the SA report demonstrates that the SA snapper stocks in each of SG/WCS and GSVS are severely overfished (likely requiring substantial stock protection over an extended period to allow stock rebuilding). The overall conclusion regarding stock status is consistent with the highly-truncated age structures and low DEPM biomass estimates. As this species can exhibit long periods of low recruitment, it will be necessary to protect the biomass resulting from recruitment in recent years (given lack of contribution to biomass from earlier recruitments).

The SA report (i) summarises the data that were used to determine stock status; (ii) assesses the status of the resource; (iii) identifies areas of uncertainty; and (iv) identifies future research needs. This review will consider these components providing feedback, however, given time constraints for the review, responses will be limited to the provision of generalised comments and recommendations.

DATA

Both fishery-dependent and fishery-independent data have been collected, analysed and subsequently used in assessments. Fishery-dependent data included commercial fishery statistics (to 2019), recreational and charter retained catch information, length and age population structures from targeted adult sampling and commercial market sampling. Fishery-independent data were estimates of spawning biomass derived from the application of the daily egg production method (DEPM) for Spencer Gulf (SG) and Gulf St. Vincent (GSV).

General comments and recommendations on chapters 2-4 are:

Chapter 2: FISHERY STATISTICS

- Clarify Fig 2.2 – Is "Other" = Charter?
- In the methods section, clarify what the expected fishery performance level

(Prop200kgTarLL & HL) is and why used? Text on page 20 suggests these don't show much trend but there is a clear peak (exceeds upper reference line) and then decline, possibly related to recruitment. Is the decline for LL not related to stock decline?

- Catch and effort based (general) reference levels based on a reference period from 1984- 2021 (Table 6.1), despite substantial variations over this time in response to changing abundance/recruitment and spatial shifts in effort – how valuable are these indicators?
- Is it reasonable to use the same fishery performance levels for GSV and SE as SG/WC, given that they are different stocks and may have different productivity?
- Add some clarity regarding any catches pre-1984, which appears to be the first year of catch data used in the assessment, and any assumptions made about recreational and charter catches in years without available data or survey estimates.
- Provide clarification and detail on CPUE time series e.g. has any statistical standardisation analysis been undertaken to account for effects of spatial blocks, temporal effects (daily/monthly), vessels, targeting classification etc, or any changes in reporting methods. Would also be useful to present uncertainty of annual/seasonal CPUE time series.
- State whether release rate data were available from historical charter or recreational fishing data sets, prior to prohibition on releasing fish.
- Additional description of the success or challenges to acquiring rigorous recreational sector data via the compulsory logbook would be useful, e.g. are there issues with any non- reporting?

Chapter 3: REGIONAL LENGTH AND AGE STRUCTURES

- Clarify source of biological data with regards to sampling methods used.
- Add sample sizes to age composition plots (like presented for length structures).
- The report only provides plots of length and age structure information collected since 2010, despite the methods section stating that market sampling has been undertaken since 2000. Suggest including age frequency distributions for earlier years (e.g. as reported in Fowler 2000; 2002), to understand changes over longer time period.

Chapter 4: REGIONAL ESTIMATES OF SPAWNING BIOMASS (DEPM)

- Acknowledge research done by SARDI since 2013-2014 FRDC project to improve application of DEPM for snapper stocks.
- Timing of surveys in 2021-2022 was well aligned with peak snapper spawning based on long- term understanding of reproductive biology.
- Good coverage of potential spawning locations with large number of stations in both SG and GSV (waters >10 m depth).
- High confidence in accuracy of identification (molecular) of snapper eggs (ISH), and in egg development stage classification.
- Adequate numbers of adult samples collected in SG and GSV in 2021-2022 (when fishery was closed) across broad geographic area, representative samples obtained across size/age.
- Correspondingly, confidence in estimated adult parameters.
- Also acknowledge methods used to estimate P_0 (arguably the most challenging component of the DEPM) uses a pre-specified value of instantaneous daily egg mortality (Z) and has been published (McGarvey et al. 2018).

- Conclusion would be that the estimates of spawning biomass from the most recent surveys provide a credible assessment of spawning biomass in SG and GSV at that time.
- Contractions in spawning areas between 2019 and 2021 surveys - lowest spawning areas on record, might just reflect lowest spawning biomass on record. Not convinced by the argument that reduction in spawning area may relate to the shift in the spatial distribution and abundance of adults (due to lack of disruption from fishing), rather than the decrease in spawning biomass. Experience from DEPM surveys in Shark Bay indicated no change in the spatial distribution of spawning activity through fishery closure period through to re-opening of fishery rather we observed increased intensity of spawning (reflected in increased egg production) at main spawning locations as stock rebuilt after fishery closure (5 yr).

ASSESSMENT

Assessments are based on the use of indicators associated with (i) 'general' fishery performance (ii) outputs from the SnapEst model applied to each biological stock and (iii) prevalence of fish > 10 y in age compositions.

The performance of the SG/WCS, GSVS and SE Regional population were determined based on the assessment of the fishery performance indicators against reference points defined in the Management Plan. 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 2021 was compared against those calculated for the reference period of 1984 to 2021 and assessed using several trigger reference points. The estimates of Prop200kgTarHL and Prop200kgTarLL for 2021 were compared against those from the reference period of 2004 to 2021, using the same trigger reference points that are used for the general performance indicators.

The SnapEst model integrates fishery-dependent, fishery-independent, and biological datasets to produce a time series of 'biological' fishery performance indicators. Collectively, the indicators of (i) fishable biomass, (ii) recruitment, (iii) harvest fraction, and (iv) egg production underpinned this stock assessment. Respective trigger reference points are indicated in Table 6-1. A fifth 'biological' fishery performance indicator is the age composition based on the catch proportions by age, derived from commercial fish market sampling and targeted adult sampling. 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.

Combined comments and recommendations on chapters 5 and 6 are:

- Include model fitting diagnostics to enable readers to assess level of fit of model to age, CPUE and DEPM biomass estimates.
- Clarify years of age composition data to which the model was fitted.
- Provide details regarding calculations of 'initial conditions' in the stock assessment model. Confirm if this is still based an estimated level of fishing mortality for the fished stock at equilibrium as described by McGarvey and Feenstra (2014)?
- It might be useful to show time series for estimated spawning biomass or relative spawning biomass (to compare with egg production). Some discussion on initial conditions, with respect to very low starting egg production, and how this relates to knowledge of development of the fishery (e.g. early statewide catches, as shown in McGlennon (1999) and early age compositions, as shown in Fowler (2000)), would be useful.

- Suggest reviewing value for natural mortality (M=0.05 y^{-1}) used in stock assessment model. It appears that Gilbert et al. (2006) used M=0.075 y^{-1} (see Table 1). Consider a sensitivity analysis, using an estimate of M from an age-based empirical estimator (e.g. Hoenig (1983) fish equation, Hamel & Cope (2022), Dureuil et al. (2021) or Dureuil & Froese (2021).
- Since CPUE is used in the absence of DEPM estimates, it might be useful to clarify if they provide similar trends where they overlap. If CPUE is considered as valid index in years prior to DEPM estimates, is there justification why it is not considered valid for the entire series, and therefore fit model to both trends? Has any efficiency been applied to CPUE time series?
- Biomass is scaled in the model to the survey area (using relative catch levels between blocks) before fitting to the DEPM estimates. DEPM estimates could be scaled outside the model (using the same catch by block scaling method) to provide scaled DEPM estimates for the area accounted for by the model region, which could be presented as inputs to model.
- Reported uncertainty seems low for many estimated quantities – e.g. model estimates of trends in fishable biomass, harvest fraction, proportion of pristine egg production time series – figure 5-1, 5-2 etc. For future assessments, consider allowing for more uncertainty, e.g. associated with model inputs (such as M, observed uncertainty of CPUE time series etc.).
- Consider, for future assessments, producing some 'lower-level' model-based assessments to provide more 'lines of evidence' to support stock status determination, e.g. catch curve analysis (allowing for variable recruitment (e.g. Deriso et al., 1985, as used by Coulson et al., 2009, see also Schnute and Haigh, 2007) to age data, to produce ('equilibrium') mortality estimates, and per recruit analyses.
- Consider, for future assessments, adding additional empirical metrics such as trends in mean age and mean length over time, proportion of mature and large fish in catches etc., to further monitor stock status changes.
- For future assessments, suggest providing some model projections, based on differing levels of F or Catch, to show likely timeframes (e.g. 10-20 y) required for stock rebuilding, for this relatively long-lived species.
- Reference points for the general (non-biological) indicators such as catch and CPUE are based on values observed during a reference period that spans the full history of the fishery (1984-2021, Table 6-1). As these indicators have fluctuated substantially over time in response to changes in abundance and shifting fishing effort between regions, the value of these indicators to measure changes in stock status is not clear.
- Suggest highlighting the key importance of the indicators from the model, particularly harvest fraction and egg production, for monitoring stock sustainability (i.e. as there are quite a few other indicators that are being monitored).

UNCERTAINTIES and FUTURE RESEARCH

Seven priorities have been identified for monitoring and assessing the status of Snapper in SA. These are: (i) continuation of the adult sampling program to access biological data and monitor regional age structures; (ii) better understand and monitor inter-annual recruitment variability of 0+ juveniles; (iii) continuation and refinement of DEPM surveys to estimate spawning biomass; (iv) develop forecasting capability in the SnapEst model; (v) improve the understanding of the Snapper

population on the WC; (vi) improve the information on recreational catches; and (vii) understand post-release mortality.

The proposed priorities for future monitoring and assessment of snapper stock status are exhaustive and there is only limited scope for provision of additional research beyond that identified in the SA report. These may include:

- The report describes the current harvest strategy and the need for this be reviewed. During this review, it would be valuable to give consideration to the indicators that will be used to measure the recovery of the stocks. As a long-lived species with highly variable recruitment, the ability to sustain future increases in catch as the fishery re-opens will be closely linked to the proportion of larger and older mature fish in the stocks to ensure resilience during periods of lower-than-average recruitment.
- It may be useful to consider the use of likely stock rebuilding times (e.g. with respect to MSC guidelines/generation times), and build these into current stock rebuilding strategy and future harvest strategy with explicit performance measures and control rules outside of 'times of crisis'. This is the case with the harvest strategy (includes a rebuilding strategy with timelines and performance measure for the stock/s) for West Coast Demersal resource in WA that includes snapper as one of the key indicator species and that is receiving local/national attention currently.
- Using the current indicator of ensuring 20% of fish in the population/stock are above 10 years of age may not always reflect an age structure that comprises appropriate proportions of older fish. Instead, the proportion of fish above 10 years of age would be influenced by the interannual recruitment variation that occurs, which could result in a highly variable proportion of fish above 10 years. Instead, fitting appropriate catch curves (that take into account recruitment variation) and identifying relevant reference points may provide a more useful reflection of appropriate age compositions.
- Noting that there is an FRDC currently underway to develop a 0+ recruitment index, that will become available to inform future modelling.

Final comment – Noting the tight timetable, any one of the DPIRD team that assisted with this review and provided specific comments would be prepared to discuss further via email or phone, if that is useful to the SARDI assessment team.