

# Fisheries



## South Australian Sardine (*Sardinops sagax*) Fishery: Stock Assessment Report 2012



T.M. Ward, P. Burch and A.R. Ivey

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SARDI Aquatic Sciences  
PO Box 120 Henley Beach SA 5022

November 2012

Report to PIRSA Fisheries and Aquaculture



Government  
of South Australia

**South Australian  
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## **PREFACE**

This is the ninth stock (fishery) assessment report for the South Australian Sardine (pilchard) Fishery (SASF). Since 1998, SARDI Aquatic Sciences has assessed the status of the stock of Australian sardine (*Sardinops sagax*) off South Australia and provided scientific advice to Primary Industries and Regions SA (PIRSA) Fisheries and Aquaculture to assist management of this fishery. Under the current Service Level Agreement with PIRSA Fisheries and Aquaculture, SARDI Aquatic Sciences provides biennial spawning biomass and stock assessment reports in alternate years. The spawning biomass report documents the biennial application of the Daily Egg Production Method (DEPM). The current report updates the previous stock assessment report by Ward et al. (2010). It synthesises information relevant to the SASF, assesses the status of the stock (including explicit consideration of uncertainty), comments on the biological suitability of current management arrangements and identifies future research needs.

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## EXECUTIVE SUMMARY

1. This report synthesises information relevant to the South Australian Sardine Fishery (SASF), assesses the status of the stock (including explicit consideration of uncertainty) and comments on future management and research needs.
2. The SASF began in 1991 and is conducted mainly in southern Spencer Gulf. It was initiated to provide fodder for the mariculture industry for southern bluefin tuna (*Thunnus maccoyii*). In recent years an increasing proportion of the catch has been used for human consumption, recreational fishing bait and pet food.
3. The Total Allowable Commercial Catch (TACC) was set at 1,000 t in 1991-92, and increased to 3,500 t during 1993-97. In 1998, the TACC was set at 12,500 t, but this was reduced to 3,500 t in 1999 and 2000, after the 1998 mass mortality event. The current Harvest Strategy indicates that the TACC will be set at 30,000 t while the spawning biomass is between 150,000 t and 300,000 t. Since 2007, the TACC has been set at 30,000 t. However, since 2010, an additional exploratory TACC of 4,000 t has been made available for capture outside the traditional fishing areas. In 2012, a temporary additional quota of 2,000 t was added to the TACC in July for capture outside the traditional fishing areas, which will be deducted from the TACC for 2013.
4. Total annual catch estimated from logbooks increased from ~7 t in 1991 to ~40,000 t in 2005 and declined to ~24,000 t in 2006. Between 2007 and 2010, estimated catches were relatively stable at ~27,000-30,000 t. In 2011, the estimated catch was ~32,000 t.
5. Total annual catches reported in Catch and Disposal Records used for quota monitoring increased from 2,597 t in 1995 to 42,475 t in 2005, before falling to 25,137 t in 2006 and stabilising at ~30,000 t in 2007-2009. In 2010, the CDR catch rose to 32,262 t after additional quota was allocated outside the traditional fishing area. In 2011, the CDR catch was 31,592 t.
6. Between 2001 and 2005, annual effort increased from 285 boat-nights (406 net-sets) to 1,233 boat-nights (1,274 net-sets). Effort declined to 713 boat-nights (840 net-sets) in 2006 and has remained stable at 713 to 902 boat-nights (840-1077 net-sets) between 2007 and 2011.
7. Effort and catches are usually low in August-October but increase during November-February and usually peak in March-June.
8. Between 2006 and 2010 size frequencies ranged from 120 to 200 mm FL with a mode at 150 mm FL. Since 2010 size frequencies have declined with modal sizes of 140 and 130 mm FL in 2011 and 2012 (January-June), respectively.
9. Growth rates of 1+ and 2+ year old fish have been relatively low in recent years. These decreases are likely to have been driven by environmental factors but do not entirely explain the recent reduction in the size of fish taken in catches.

10. From 2001 to 2004, 3+ year old fish dominated catches. Fish caught between 2005 and 2010 were comprised primarily of 2+ and 3+ year olds. Since 2010 there has been an increase in the proportion of younger fish in catches, with 2+ year olds dominating in 2011 and 2012 (January-June only).
11. Between 2001 and 2009 the proportion of immature fish in catches from southern Spencer Gulf was below 30%. Since 2010, the immature proportion of the catch has increased markedly to be >80% in both 2011 and the first six months of 2012.
12. Spawning biomass estimates obtained using the DEPM remained stable at 152,000-170,000 t during 2003 to 2005, and rose to ~263,000 t in 2007 before declining to ~172,000 t in 2009. The DEPM-based spawning biomass in 2011 was ~193,000 t.
13. Model-generated estimates of spawning biomass were stable from 2003 to 2011 under all scenarios tested, typically at levels of ~170,000-190,000 t. Under the majority of scenarios, spawning biomass is predicted to remain above the limit reference point of 150,000 t over the next two years for TACCs up to 45,000 t.
14. DEPM-based (~193,000 t) and model-generated (~180,000 t) estimates of spawning biomass are above the Lower Limit Reference Point of 150,000 t. These estimates provide evidence that the South Australian sardine stock is being fished within sustainable limits.
15. The decline in size/age of sardines caught in the southern Spencer Gulf over the last few seasons suggests that the proportion of the catch being taken from southern Spencer Gulf may be too high. There is a need to consider increasing the proportion of the SASF catch that is taken outside southern Spencer Gulf.
16. The Lower Limit [Target] Reference Point for spawning biomass in the previous SASF Harvest Strategy of 150,000 t is ~75% of the current spawning biomass. Recent studies have shown that maintaining biomasses of low trophic level species, such as sardine, above 75% of unfished biomass protect ecosystem function and biodiversity. This limit is considered to be particularly conservative for Australian ecosystems. This suggests that 150,000 t would be suitable as the Limit Reference Point for spawning biomass in the next Harvest Strategy for the SASF. Maximum Sustainable yield for species such as sardine is typically at around 40% of unfished biomass and a Lower Limit Reference Point of ~75,000 t may be appropriate.
17. The next Harvest Strategy for SASF may include a spatial management framework that provides the capacity to limit the proportion of the catch taken from southern Spencer Gulf. There is also a need to consider including an age-based Biological Performance Indicator in the next Harvest Strategy.

## 1.0 GENERAL INTRODUCTION

### 1.1 Rationale and Objectives

This is the ninth stock (fishery) assessment report by SARDI Aquatic Sciences on the South Australian Sardine Fishery (SASF). It informs management of the SASF and summarises data collected on Australian sardine, *Sardinops sagax* (Jenyns 1842, Clupeidae) since 1991, when the fishery began, up until December 2011. The objectives of the report are to: review scientific literature on the biology, ecology and fisheries for *S. sagax* and describe the development and management of the SASF (Chapter 1); present catch, effort, catch-per-unit-effort (CPUE) and size/age composition data for 1991 to 2011 (Chapter 2); describe the age, growth, condition and reproductive biology of *S. sagax* in South Australia (Chapter 3); report on inter-annual trends in spawning biomass during 1995-2011 (Chapter 4); use an age structured model that synthesises existing data to predict future biomasses under a range of catch levels (Chapter 5); and assess the current status of the resource (including explicit consideration of uncertainty) and comment on current management arrangements and future research needs (Chapter 6).

### 1.2 Literature Review

#### 1.2.1 Taxonomy

Historically, five species have been recognised in the genus *Sardinops*: *S. ocellatus* off southern Africa; *S. neopilchardus* off southern Australia and New Zealand; *S. sagax* off the west coast of South America; *S. caeruleus* off the west coast of North America; and *S. melanostictus*, around Japan (Whitehead 1985). However, most fisheries scientists now follow the taxonomy for the genus proposed by Parrish et al. (1989), who suggested that the genus *Sardinops* is mono-specific with no valid sub-species and that the name *Sardinops sagax* (Jenyns 1842) has taxonomic priority. This finding was confirmed by Grant and Leslie (1996). Grant et al. (1998) suggested that cluster and parsimony analyses of haplotypic divergences supported the hypothesis that there were three lineages within the genus: southern Africa (*ocellatus*) and Australia (*neopilchardus*); Chile (*sagax*) and California (*caeruleus*); and Japan (*melanostictus*). Polymorphic micro-satellites that have been isolated have the potential to help resolve some of the minor taxonomic questions that remain for this species (Pereya et al. 2004).

The common and scientific names of the Australian sardine or pilchard have been debated for many years. The names pilchard and *Sardinops neopilchardus* (Steindachner 1879) were used in many Australian publications prior to 2000. However, the terms Australian

Sardine and *Sardinops sagax* (Jenyns 1842) have been listed in the Standard Fish Names List for Australia since May 2006. *Sardinops sagax* is also the name used in the Australian Faunal Directory and Eschmeyer's Catalogue of Fishes. For the sake of brevity in this report, we use the term sardine to refer to *S. sagax* in Australia and elsewhere.

### 1.2.2 Distribution

Sardine occurs in cool temperate to sub-tropical waters of the Northern and Southern Hemispheres where, along with a local species of anchovy (*Engraulis* spp.), it commonly dominates the fish biomass in upwelling regions. Sardine supports important fisheries in the eastern boundary current systems off the west coasts of northern and southern Africa and North and South America. These globally important upwelling regions are known as the Canary, Benguela, California and Humbolt Current Systems, respectively. The sardine is also found in the rich coastal waters surrounding Japan, and the generally less productive waters off southern Australia and around New Zealand.

In Australia, sardine is found throughout temperate and sub-tropical waters between Rockhampton (Queensland) and Shark Bay (Western Australia), including northern Tasmania (Gomon et al. 1994). Waters off South Australia are the centre of the sardine's Australian distribution and support the largest component of the Australian population (Ward et al. 2006). Shelf waters off South Australia also include Australia's largest upwelling system, the Finders Current system (Ward et al. 2006c). Sardine is the dominant clupeid off South Australia, occurring in the southern portions of Gulf St Vincent and Spencer Gulf and over the continental shelf (Ward et al. 2001a, b, c). As in other upwelling systems worldwide, one species of engraulid, the Australian anchovy (*Engraulis australis*) is also abundant off South Australia (Dimmlich et al. 2009). When sardine biomass is high, the Australian anchovy occurs mainly in the northern gulfs, but when sardine biomass is low this species has the capacity to increase in abundance and expand its distribution into shelf waters (Ward et al. 2001a; Dimmlich et al. 2004; Dimmlich and Ward 2007).

### 1.2.3 Movement patterns and stock structure

*Sardinops* is known to undergo extensive migrations. For example, off Africa, sardine migrate along the coast to access conditions that are favourable for spawning and the survival of recruits (van der Lingen and Huggett 2003). Similarly, it has been hypothesised that schools of sardine migrate into waters off northern New South Wales and southern Queensland during winter-spring to spawn (Ward and Staunton-Smith 2002). The movement patterns of sardine in waters of South Australia are largely unknown although there is evidence of an ontogenetic shift in distribution with larger, older fish most

commonly found in shelf waters and smaller, younger fish mainly found in embayments including Spencer Gulf (Rogers and Ward 2007).

In the absence of alternative information, estimates of spawning biomass obtained from the spawning aggregation that occurs annually in northern New South Wales (NSW) and southern Queensland have been assumed to be reflective of abundance for the entire east coast and used to inform fisheries management decisions (e.g. Ward and Rogers 2007a). The need to investigate patterns of movement along the east coast has been identified by fisheries managers for the NSW Ocean Haul Fishery and the Commonwealth Small Pelagic Fishery. As fishing effort in the SASF is concentrated in southern Spencer Gulf but a substantial proportion of the spawning biomass is located in the Great Australian Bight, understanding the rate of movement between shelf and gulf waters has been identified as a high priority for the fishery. A collaborative project funded by the Fisheries Research and Development Corporation (Izzo et al. 2012) provided some evidence of a level of separation between the southern Australian and east coast populations using otolith shape and elemental composition. Shape and elemental analysis of sardine otoliths indicated a level of inter-annual variation in the level of population sub-structuring off South Australia (Izzo et al. 2012), possibly driven by environmental variations.

There appears to be a high level of genetic heterogeneity within the Australian population of sardine, but there is limited evidence of spatially consistent stock structure (e.g. Ward et al. 1998; Izzo et al. 2012). The existence of separate eastern and western stocks has been proposed for species, including sardine, that are targeted in the Commonwealth Small Pelagic Fishery (Bulman et al. 2008), with Bass Strait suggested as a significant barrier to genetic flow. The annual migration of sardine along the east coast suggests that this region may not include more than one stock.

The existence of a possible stock division for small pelagic species, such as sardine, off south-western Australia (between western Tasmania and southern west coast of Western Australia) was also proposed by Bulman et al. (2008), but the level of separation was considered less distinct than for the east coast because of the lack of a clearly defined barrier to movement (like Bass Strait). However, there is evidence that some sardine larvae spawned in Western Australia may be transported into South Australian waters by the Leeuwin Current (Gaughan et al. 2001).

Several studies on stock structure of sardine in Australia have been conducted off Western Australia, where it has been suggested that there are distinct breeding stocks off

the west and south coasts (Edmonds and Fletcher 1997; Gaughan et al. 2001, 2002). The south coast stock consists of three assemblages that are targeted separately in the fisheries that operate out of Albany and Bremer Bay. The two breeding stocks and three assemblages are managed as independent units with separate Total Allowable Commercial Catches. Larvae may mix between these Western Australia regions, as they are transported eastwards by the Leeuwin Current. Westward migration of juveniles may be required to maintain functionally distinct adult aggregations.

#### 1.2.4 Schooling behaviour

The schooling behaviour of sardine is complex, and varies at both spatial and temporal scales. For example, Barange and Hampton (1997) found that schools remained at similar densities throughout the day, whereas Misund et al. (2003) found that schools were highly dynamic and densities changed during the afternoon. Similarly, in some parts of the California Current system (e.g. Oregon) schools of sardine have also been observed displaying patterns consistent with anti-predator behaviour, dispersing at night, reforming rapidly at sunrise and remaining in schools throughout the day (Kaltenberg and Benoit-Bird 2009). However, at other locations in the California Current system (e.g. Monterey Bay) transitions between day and night behaviour occur more gradually (Kaltenberg and Benoit-Bird 2009). Factors such as habitat heterogeneity, food availability, predation levels and vessel noise can influence schooling behaviour (Freon 1993; Giannoulaki et al. 2003). Schooling behaviour of sardine in Australian waters varies within and among locations, days, months and years and complicates the application of acoustic techniques for stock assessment (T.M. Ward, unpublished data). There is some evidence that spawning influences schooling behaviour as well, with spawning females under-represented in commercial catches (Ward et al. 2011a).

#### 1.2.5 Food and feeding

Sardine has two feeding modes: filter-feeding on micro-zooplankton and phytoplankton and particulate-feeding on macro-zooplankton. Sardine switch between these two modes depending on relative prey density (van der Lingen 1994; Louw et al. 1998; van der Lingen 2002). Sardine tend to feed on smaller zooplankton than anchovies (e.g. Espinoza et al. 2009). The prey composition of sardine varies among regions. For example, in the northern Humboldt Current system, sardine tend to feed on larger prey and consume more euphausiids than in the northern Canary and southern Benguela systems.

In South Australian waters, sardine have been found to consume 12 prey taxa with krill (29.6% biomass) and unidentified crustacean (22.2% biomass) contributing the highest

biomass (Daly 2007). Krill occurred in greater numbers (65.3%) than crustaceans (27.0 %). Crab zoea, other decapods, copepods, polychaetes, fish eggs and larvae and gelatinous zooplankton were also present in the diet of sardine in South Australia (Daly 2007).

#### 1.2.6 Reproduction

Sardine are serial spawners with asynchronous oocyte development and indeterminate fecundity. Serial or batch spawning fishes release numerous batches of pelagic eggs throughout an extended spawning season. The number of eggs in a batch, or batch fecundity, is correlated with female size and varies among locations and years (Lasker 1985).

In Australia, sardine usually spawn in open waters between the coast and shelf break (Blackburn, 1950; Fletcher and Tregonning 1992; Fletcher et al. 1994). In most locations there is one spawning season per year, but off Albany in Western Australia there are two (Fletcher 1990). The size and age at which sardine reach sexual maturity vary between locations, and range from 100 to 180 mm FL, and 1.8 to 2.8 years, respectively (Blackburn 1950; Joseph 1981; Stevens et al. 1984; Fletcher 1990; Macewicz et al. 1996).

The reproductive biology of sardine in South Australian waters is relatively well known. Ward and Staunton-Smith (2002) found approximately 50% of males and females reach sexual maturity ( $L_{50}$ ) at 146 and 150 mm, respectively. Spawning occurs during the summer-autumn upwelling period of January-April (Ward et al. 2001a, b; Ward and Staunton-Smith 2002). Females spawn batches of 10,000-30,000 pelagic eggs approximately once per week during the extended spawning season. Eggs are abundant in the southern gulfs and shelf waters (Ward et al. 2001a, b; Ward et al. 2003, 2004).

#### 1.2.7 Early life history and recruitment

Sardine has a relatively long larval phase: eggs hatch approximately two days after fertilization and larvae metamorphose at 1-2 months of age (Neira et al. 1998). Survival rates of sardine eggs and larvae strongly affect recruitment success (Lo et al. 1995). The large variations in abundance that characterise sardine populations worldwide have been attributed to fluctuations in recruitment, which can be influenced by environmental factors, regime shifts and over-fishing (e.g. Galindo-Cortes 2010). Larval survival is a key determinant of recruitment success but the factors affecting survivorship may vary spatially and temporally. The effects of food availability on larval survival have been discussed at

length (Lasker et al. 1975) but there has been less consideration of the effects of egg and larval predation on recruitment success (e.g. Agostini 2007).

In the Californian Current system recruitment success and large population sizes have coincided with warm periods characterised by reduced primary productivity, when the centre of spawning shifts northward (e.g. Lluch-Belda et al. 1992). Surprisingly, a recent study linked high zooplankton abundance to poor recruitment of sardine, suggesting that decreased predation pressure on eggs and larvae may be a key factor contributing to strong recruitment in the Californian Current system (Agostini et al. 2007). Similarly, a reduction in zooplankton abundance was followed by an increase in sardine recruitment in Peru (Carrasco and Lozano 1989) and the sardine population off South Africa collapsed when zooplankton abundance increased (Verheye 2000).

In South Australia, sardine larvae are highly abundant at temperature and salinity fronts that form near the mouths of the two gulfs during summer and autumn (Bruce and Short 1990) and in mid-shelf waters off the Eyre Peninsula. Spawning coincides with the summer-autumn upwelling period in South Australia (e.g. Ward et al. 2006a). It is predicted that the intensity and duration of upwelling events could increase under climate change scenarios (Hobday et al. 2009). The effects of such changes on the future recruitment success of sardine in South Australia are poorly understood.

#### 1.2.8 Age, growth and size

Age determination studies of sardine have involved counting growth increments in scales (Blackburn 1950) and sagittae (Butler et al. 1996; Fletcher and Blight 1996), and modelling the formation of marginal increments (Kerstan 2000). Daily deposition of growth increments in the otoliths of larvae and juveniles has been validated in laboratory trials (Hayashi et al. 1989). Validation studies involving the capture and maintenance of sardine and other clupeoids have proven to be problematic owing to logistical difficulties (Fletcher 1995) and sensitivity to handling (Hoedt 2002; Rogers et al. 2003). Other methodological approaches have been used to show that translucent zones form annually in the sagittae of 1+ year old sardine off South Africa (Waldron 1998),  $\leq 2+$  year olds off North America (Barnes et al. 1992) and  $\geq 4+$  year olds off Western Australia (Fletcher and Blight 1996). Despite this apparently sound theoretical basis for using increment-based age-determination methods, the application of these standard approaches has proven to be problematic in Western Australia, South Australia and California due to difficulties associated with interpreting and counting opaque and translucent zones (Butler et al. 1996; Fletcher and Blight 1996; Rogers and Ward 2007).

Studies of growth dynamics of sardine in the Benguela and California Current systems suggest that growth rates of larvae (up to 0.85 mm.day<sup>-1</sup>) and juveniles (0.48-0.63 mm.day<sup>-1</sup>) are high (Butler et al. 1996; Kerstan et al. 1996; Watanabe and Kuroki 1997; Quinonez-Velazquez et al. 2000; Skogen 2005). In South Africa, sardine were found to reach larger asymptotic sizes ( $L_{\infty} = 221$  mm) and have lower growth constants ( $k = 1.09$  year<sup>-1</sup>) than those off southern California ( $L_{\infty} = 205$  mm,  $k = 1.19$  year<sup>-1</sup>, Thomas 1979, 1983, 1984, 1985; Butler et al. 1996). Growth parameter estimates available for sardine in Western Australia (Fletcher and Blight 1996) suggest that growth in this area is slower and that fish reach smaller asymptotic sizes than those in the more productive eastern boundary current systems.

A detailed study by Rogers and Ward (2007) showed that the growth rates of sardine are higher in South Australian waters than off other parts of the Australian coastline, but lower than those in more productive boundary current ecosystems (Ward et al. 2006c). A notable finding of the study was that fish in commercial catches were younger (and smaller) than those obtained in fishery-independent samples. This finding has significant implications for the use of age structured models (based on fishery samples) for stock assessment of the SASF (see Chapter 5).

#### 1.2.9 Mass mortality events

Mass mortality events in 1995 and 1998/99, spread throughout the entire Australian range of sardine and are thought to have killed more fish over a larger area than any other single-species fish-kill recorded (Jones et al. 1997). Characteristics of the mortalities, such as their focal origin, rapid spread throughout the entire geographical range of the population and high mortality rates suggest both were caused by an exotic pathogen to which Australian sardine were naïve (Jones et al. 1997; Gaughan et al. 2000; Whittington 2008).

Each event was estimated to have killed over 70% of the spawning biomass in South Australian waters (Ward et al. 2001b). Recovery of the South Australian population following these events has been monitored using the daily egg production method (DEPM, Lasker 1985; Ward et al. 2001c). Assessment of the impacts of the mortality events on the age structure of the South Australian population of sardine and the use of age structured models to assess the capacity of the population to recover, has been impeded by difficulties associated with obtaining reliable estimates of age (Ward et al. 2005a).

#### 1.2.10 Role in the ecosystem

Small pelagic fishes such as sardine live mainly in the upper layers of the water column, and convert energy produced by phytoplankton into a form that is available to higher vertebrates. Sardine are an important food source for many predatory fishes (Hoedt and Dimmlich 1995; Alheit and Niquen 2004; Ward et al. 2006c), squid (O'Sullivan and Cullen 1983), seabirds (Montevecchi et al. 1995; Dann et al. 2000; Crawford 2003) and marine mammals (Ohizumi et al. 2000; Page et al. 2005). Populations of predators fluctuate in response to changes in productivity in the regions where they feed. Therefore, aspects of the reproductive and feeding ecology of predators have been used to monitor the health of marine ecosystems, assess changes in environmental conditions, and monitor effectiveness of fishery management regimes (Wanless et al. 1982, Rindorf et al. 2000, Boyd et al. 2006). An FRDC study undertaken to assess the importance of sardine in the diet of marine predators in waters of the eastern Great Australia Bight (Goldsworthy et al. 2011) found no evidence of ecological impacts from the SASF.

The effect of the sardine mortality events on marine predators provides evidence of the important role in the pelagic ecosystems off southern Australia. In Victoria, negative impacts of the sardine mortality events on apex predators were documented for little penguins (*Eudyptula minor*) and little terns (*Sterna albifrons*) in 1995/1996 (Dann et al. 2000; Taylor and Roe. 2004), and Australasian gannets (*Morus serrator*) in 1998/1999 (Bunce and Norman 2000; Bunce et al. 2005). In South Australia, McLeay et al. (2008) investigated the diet, age structure, and morphology of a population of crested tern (*Sterna bergii*) to determine whether survival and growth were reduced for cohorts reared in years immediately following sardine mortality events. Age-specific information collected from banded adults indicated that the cohort reared after the first sardine mortality event in 1995 exhibited significantly lower rates of recruitment than predicted by life-table analyses, suggesting that survival was reduced in response to the absence of sardine. Females from cohorts reared one year after the end of each sardine mortality event (in 1996 and 1999) were smaller than other age classes, suggesting that chick growth was reduced during periods of low sardine abundance.

#### 1.2.11 Stock assessment

##### *Daily Egg Production Method*

The Daily Egg Production Method (DEPM, Parker 1980; Lasker 1985) has been used to estimate the spawning biomass of sardine in South Australia since 1995. Estimates of spawning biomass are the key biological indicator in the management plan for the SASF

(Shanks 2005). The advantage of this approach is that it provides direct estimates of spawning biomass on which to base management decisions.

The DEPM was originally developed for direct stock assessment of the northern anchovy, *Engraulis mordax*, off the west coast of North America (Parker 1980). The method relies on the premise that the biomass of spawning adults can be calculated by dividing the mean number of pelagic eggs produced per day throughout the spawning area, i.e. total daily egg production, by the mean number of eggs produced per unit mass of adult fish, i.e. mean daily fecundity (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 batch fecundity (number of oocytes in a batch,  $F$ ) and mean spawning fraction (proportion of mature females spawning each day/night,  $S$ ) by mean female weight ( $W$ ). Spawning biomass ( $SB$ ) is calculated according to the equation:

$$SB = P_0 \cdot A / (R \cdot F \cdot S / W).$$

The DEPM can be applied to fishes that spawn multiple batches of pelagic eggs over an extended spawning season (e.g. Parker 1980). Data used to estimate DEPM parameters are typically obtained during fishery-independent surveys. The key assumptions of the method are that: surveys are conducted during the main (preferably peak) spawning season; 2) the entire spawning area is sampled; 3) eggs are sampled without loss and identified without error; 4) levels of egg production and mortality are consistent across the spawning area; and 5) representative samples of spawning adults are collected during the survey period (Parker 1980; Alheit 1993; Hunter and Lo 1997; Stratoudakis et al. 2006).

The DEPM has been used for stock assessment of at least fifteen species of small pelagic fishes, mostly clupeoids (e.g. Stratoudakis et al. 2006). Although the method is used widely, a range of problems have been encountered and estimates of spawning biomass are generally considered to be accurate (unbiased) but relatively imprecise (e.g. Alheit 1993; Hunter and Lo 1997; Stratoudakis et al. 2006). The imprecision that characterises the DEPM is mainly due to uncertainties associated with the estimation of total daily egg production, i.e.  $P_0$  and  $A$ . (Fletcher et al. 1996; McGarvey and Kinloch 2001; Ward et al. 2001a; Gaughan et al. 2004; Stratoudakis et al. 2006). However, many studies have also been impeded by difficulties associated with obtaining representative samples of spawning adults for estimation of adult reproductive parameters, especially  $S$  (see Stratoudakis et al.

2006). There are relatively few published examples where the DEPM has been applied for extended periods and robust estimates of all parameters have been obtained consistently.

At least two reviews have concluded that the DEPM may be better tailored to anchovies (*Engraulis* spp.) than sardine (Alheit 1993; Stratoudakis et al. 2006). The main argument used to support this assertion is that a higher proportion of anchovies are actively spawning during the peak spawning season, daily fecundity can be estimated more precisely for anchovies than sardine (e.g. Alheit 1993; Stratoudakis et al. 2006). Despite the apparent limitations of the method for stock assessment of sardine, the DEPM is currently considered the best technique available for this species off the west coast of North America (e.g. Lo et al. 1996, 2005) and the western and southern coasts of Australia (Fletcher et al. 1996; Ward et al. 2001b; Gaughan et al. 2004). However, there are important differences between locations in the manner in which egg and adult samples are collected, data are analysed and estimates of spawning biomass are used to support fisheries management. For example, a Continuous Underway Fish Egg Sampler (CUFES) has been used routinely in surveys off California, but not Australia (e.g. Lo et al. 2001). In addition, adult samples have usually been collected by mid-water trawling off California, purse-seining off Western Australia (Gaughan et al. 2004) and gill-netting off South Australia (Ward et al. 2001b). Furthermore, estimates of spawning biomass obtained from DEPM surveys are used directly for fisheries management in South Australia, but are incorporated into age-structured stock assessment models in California and Western Australia.

A recent study re-analysed data collected between 1998 and 2007 in waters off South Australia to optimise the use of DEPM for estimating the spawning biomass of sardine (Ward et al. 2011b). Estimates of each parameter were calculated using the full range of data available (i.e. samples obtained from research and commercial vessels) and the main analytical methods described in the literature for each DEPM parameter. The most appropriate sampling methods and statistical approaches were identified on the basis of model diagnostics and the logical consistency of inter-annual trends. Estimates of each parameter obtained using the most appropriate sampling method and statistical approach were used to calculate estimates of *SB* between 1998 and 2007. Methods established in this review were followed in subsequent DEPM estimates in 2009 and 2012.

The analyses by Ward et al. (2011b) confirmed that estimates of spawning biomass (*SB*) are imprecise and show that if inappropriate analytical methods are used they can also be biased. Estimates of spawning biomass are most affected by observed levels of variation

in estimates of mean daily egg production ( $P_0$ ), total spawning area ( $A$ ) and mean spawning fraction ( $S$ ). The log-linear egg mortality model (with one egg added to each day class of eggs at each positive site) should be used to estimate  $P_0$  because it fits strongly over-dispersed sardine egg density data better and provides more logically consistent estimates of  $P_0$  than the exponential mortality model or the Generalised Linear Models (GLM) that were tested. Most GLMs produced inflated estimates of  $P_0$  when egg data were strongly over-dispersed. The Voronoi natural neighbour method should be used to estimate  $A$ , because it reduces subjectivity in sub-division of the sampling area. Most importantly, the area surveyed should be sub-divided into a large number (e.g. 300) of similar sized grids. Potential biases in  $S$  can be minimised by including all spawning females, i.e. those with hydrated oocytes, day-0, day-1 and day-2 post-ovulatory follicles (POFs), in calculations of this parameter. Research priorities to improve the precision of estimates of spawning biomass include: developing methods for using data from a CUFES to enhance estimation of  $P_0$  and  $A$ ; measuring the degeneration rates of POFs in a range of locations; and comparing the vulnerability of females with different POF stages to capture in various sampling gears.

#### *Acoustic techniques*

Acoustic techniques have been used widely for the stock assessment of small pelagic fishes (Beckley and van der Lingen 1999; van der Lingen and Huggett 2003; Simmonds and MacLennan 2005) and these studies have significantly contributed to the understanding of sardine movement (Barange et al. 1999), stock structure (Barange and Hampton 1997), relationships with oceanographic features (Lynn 2003; Tameishi et al. 1996), predator-prey interactions and inter-annual variability in abundance (Barange et al. 1999). Sources of error in acoustic estimation of biomass include: variations in target strength due to spatial or temporal differences in fish behaviour (Freon et al. 1993), difficulties associated with discrimination of species and size classes and spatial sampling error (Simmonds and MacLennan 2005). Acoustic surveys often produce biased estimates of biomass and require more sophisticated and expensive infrastructure, higher levels of technical support and expertise, and a longer developmental phase than the DEPM.

#### 1.2.12 Management procedures in small pelagic fisheries

Modern fisheries management aims to constrain exploitation rates within biologically sustainable limits while maximising potential yields, and establishing performance indicators and reference points that underpin harvest strategies (Gabriel and Mace 1999). Management procedures that include agreed operational targets and decision rules have been successfully incorporated into the management systems of several commercial

fisheries, including the South African Pelagic Fishery, Western Australian Pilchard Fishery, Pacific Sardine Fishery and SASF (Cochrane et al. 1998; De Oliveira. et al. 1998; Cochrane 1999; Gaughan et al. 2004; Gaughan and Leary 2005a, b; Hill et al. 2005).

In the South African Pelagic Fishery and Pacific Sardine Fishery, operational targets and decision rules are based around the outputs of age-structured models that use survey data and other information to generate estimates of 1+ biomass. Similarly, the TACs for the sardine fisheries in Western Australia are based on estimates of spawning biomass generated using an age-structured model (Gaughan and Leary 2005a, b). In these Western Australian fisheries, where the stocks are recovering from substantial declines in abundance, the decision rules indicate that exploitation rates should not exceed 15-20% of the spawning biomass (Cochrane 1999; Gaughan and Leary 2005a, b).

### **1.3 The South Australian Sardine Fishery**

The SASF is currently managed by the *Fisheries Management (Marine Scalefish Fisheries) Regulations 2006* and *Fisheries Management Act 2007*. Management goals for the SASF are consistent with the objectives of the *Fisheries Management Act 2007* and are outlined in the Management Plan (Shanks 2005). Management measures include entry limitations, gear restrictions and individual transferable quotas. Purse seine nets must not exceed 1,000 m in length or 200 m depth with meshes of 14 to 22 mm. There are currently 14 license holders with several companies operating multiple licenses. The costs of the policy, compliance and research programs that are needed to manage the SASF are recovered through license fees collected by PIRSA Fisheries and Aquaculture.

The Total Allowable Commercial Catch (TACC) was set at 1,000 t in 1991 and 1992, and increased to 3,500 t during 1993-1997 (Figure 1-1). In 1998, the TACC was set at 12,500 t, but this was reduced to 3,500 t in 1999 and 2000, after the mass mortality event in late 1998. The TACC increased rapidly to reach 50,000 t in 2005. Since 2007, the TACC has been set at 30,000 t. Since 2010, an additional exploratory TACC of 4,000 t was made available for capture outside the traditional fishing areas (i.e. the Spencer Gulf region (Figure 1-1 and 1-2)) to investigate the feasibility of sardine fishing in the West Coast region. In July 2012, a temporary additional quota of 2,000 t was allocated to the fishery for the 2012 quota period for capture outside the traditional fishing areas, in response to an unanticipated shortfall of feed supply to the southern bluefin tuna farming sector. The 2013 TACC will be reduced by 2,000 tonnes.

The key biological performance indicator for the SASF is the estimate of spawning biomass obtained using the DEPM. From 1997 to 2006, the TACC for the following calendar year was set as a proportion of the spawning biomass (i.e. 10.0-17.5%, depending on the size of the spawning biomass). More recently, the indicative TACC has been set at 30,000 t (PIRSA 2007). The Harvest Strategy indicates that the indicative TACC will be established as the effective TACC while the latest estimate of spawning biomass obtained using the DEPM remains between 150,000 and 300,000 t (which correspond to exploitation rates of 20% and 10%, respectively) and there is no other evidence of a significant decline in stock status.

For the purposes of this report the SASF is divided into three broad spatial regions: Western Region, Spencer Gulf and Eastern Region (Figure 1-2).

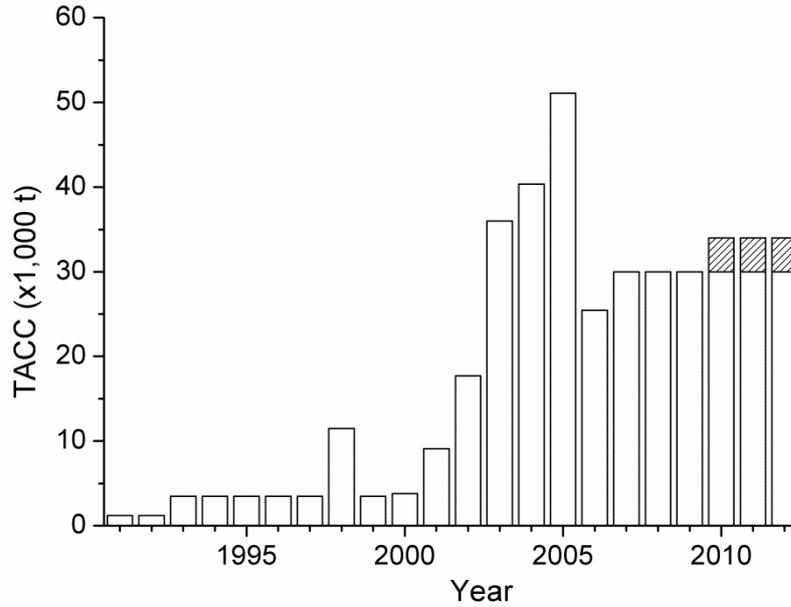


Figure 1-1. TACC for the SASF between 1991 and 2012 (hashed TACC for 2010-2012 is to be caught outside traditional fishing area).

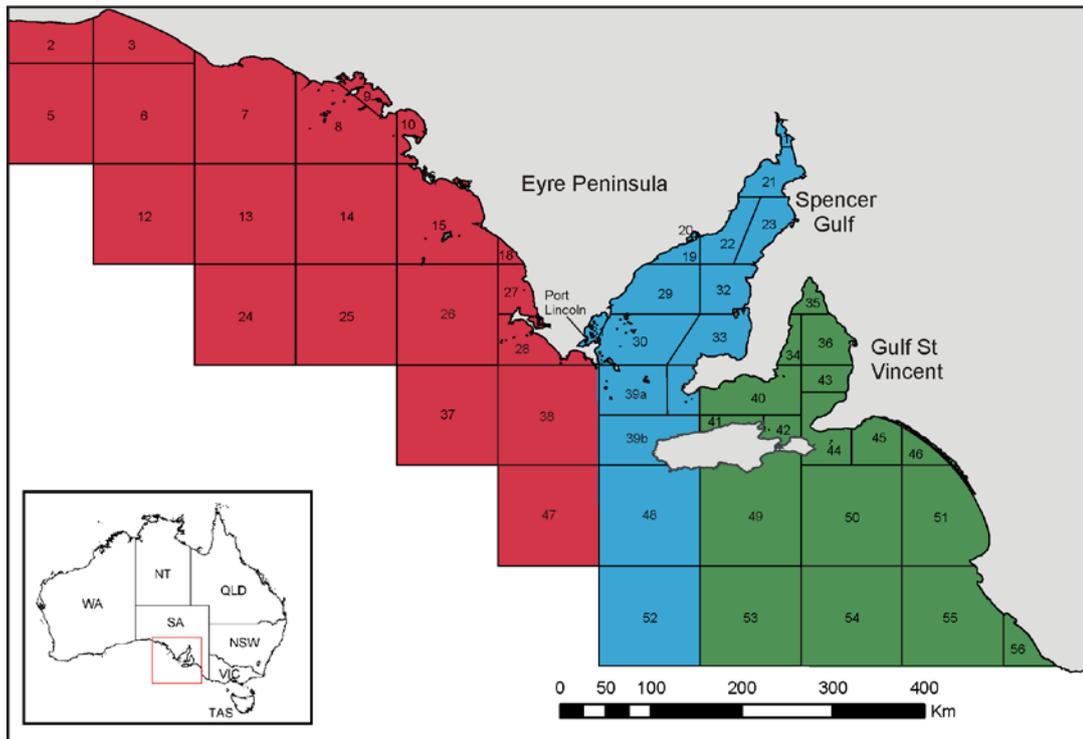


Figure 1-2. Locations mentioned in the text of this report. Colours show the division of Marine Fishing Area (MFA) fishery blocks used for spatial analysis (red – Western Region, blue – Spencer Gulf and green – Eastern Region) of catch, effort and biological data between 1991 and 2012.

## 2.0 FISHERY INFORMATON

### 2.1 Introduction

This chapter presents catch, effort, catch-per-unit-effort (CPUE) and size and age composition data from the start of the SASF on 1 January 1991 to 31 December 2011, size composition data for 2012 to date has also been included. Information is used to describe spatial and temporal patterns in fishing activities and to make inferences about stock status. Data presented in this chapter are key inputs to the population model (Chapter 5).

### 2.2 Methods

#### 2.2.1 Data collection

Catch and effort data were collated from data in fishery logbooks. Prior to 2001, effort and catch were reported with respect to MFA (Figure 1-2). After 2001, effort and catch were reported by latitude/longitude and aggregated with respect to 10 x 10 km grid squares. From 2001 additional information on effort (net-sets) and lost target catch were also recorded. Estimated annual catches are aggregates of daily catches recorded in logbooks. CPUE is estimated from aggregates of catch and effort data in Fishery Logbooks. Actual total annual catches were estimated from Catch Disposal Records (CDR) collated by PIRSA Fisheries and Aquaculture.

#### 2.2.2 Commercial catch sampling

Between 1995 and 2012, samples of the commercial catch were collected from vessels under a range of sampling protocols, most recently by independent observers who are present on about 10% of fishing trips. Size frequencies were constructed from caudal fork lengths (FL), aggregated into 10 mm length classes for all samples. Age determination methods are described in Chapter 3.

#### *Sex ratio*

The proportion of females in commercial catch samples was calculated using the equation:

$$SR = \frac{nF}{(nF + nM)},$$

where,  $nF$  is the number of females and  $nM$  is the number of males in samples. Two sided exact ratio tests (Fay 2010) were used to determine if the proportion of females in the population was significantly different from 0.5. To account for the application of multiple significance tests a Bonferroni adjustment was made to the alpha value at which the null hypothesis is accepted (Kutner et al. 2005). Note that sex was not recorded for commercial samples obtained in 2007.

## 2.3 Results

### 2.3.1 Effort, catch and CPUE

#### *Annual patterns*

The SASF expanded quickly after its inception with total effort (purse seine) and catches recorded in logbooks increasing from 5 boat-nights and approximately 7 t during 1991 to 736 boat-nights and 3,241 t in 1994 (Figure 2-1). However, total effort and catch were reduced significantly in 1995 as a result of the first mass mortality event, but increased rapidly to reach 530 boat-nights and 5,973 t in 1998. In 1999, after the second mass mortality event in late 1998, effort and catch declined to 345 boat-nights and 3,081 t, respectively.

Since the second mortality event, the fishery has expanded rapidly, with total effort reaching 1,274 net-sets across 1,233 boat-nights in 2005, with an estimated total catch of 39,831 t (Figure 2-1). Since 2007, total effort has stabilised at approximately 850-1100 net-sets over 700-900 boat-nights with an estimated catch of 27,500-32,000 t. In 2011, there were 1077 net-sets made over 847 boat-nights for an estimated catch of 31,864 t.

Actual catches recorded in CDRs have exceeded catches estimated in logbooks in most years. Actual catches increased from 2,597 t in 1995 to 42,475 t in 2005, and fell to 25,137 t in 2006 (Figure 2-1). Since 2007, actual catches have ranged from 29,854 to 32,262 t. In 2011, 31,592 t were recorded in CDRs.

Mean CPUE was calculated in boat-nights only between 1991 and 2001 and both net-sets and boat-nights between 2001 and 2009 (Figure 2-1). Mean  $CPUE_{(boat-night)}$  increased from 1.3 t.boat-night<sup>-1</sup> in 1991 to 11.3 t.boat-night<sup>-1</sup> in 1999 and reached 35.6 t.boat-night<sup>-1</sup> in 2003 before declining to 32.3 t.boat-night<sup>-1</sup> in 2005.  $CPUE_{(boat-night)}$  then rose to 37.4 t.boat-night<sup>-1</sup> in 2008 and was 37.6 t.boat-night<sup>-1</sup> in 2011. Mean  $CPUE_{(net-set)}$  increased from 7.6

t.net-set<sup>-1</sup> in 2001 to 32.4 t.net-set<sup>-1</sup> in 2008. CPUE<sub>(net-set)</sub> has remained between 28.2 and 32.2 t.net-set<sup>-1</sup> since 2008 and was 29.6 t.net-set<sup>-1</sup> in 2011.

#### *Intra-annual patterns*

Between 1991 and 2011, there has been a reasonably consistent intra-annual pattern in fishing effort (Figure 2-2). Relatively little fishing is usually conducted during August to October. Effort and catches often begin to increase in November/December, prior to the start of the southern bluefin tuna farming season. Catches continue to increase during January-February and usually peak in March-June. The peak fishing season reflects the extended periods of calm weather between April and June and the high demand for sardine to feed southern bluefin tuna following their capture during summer.

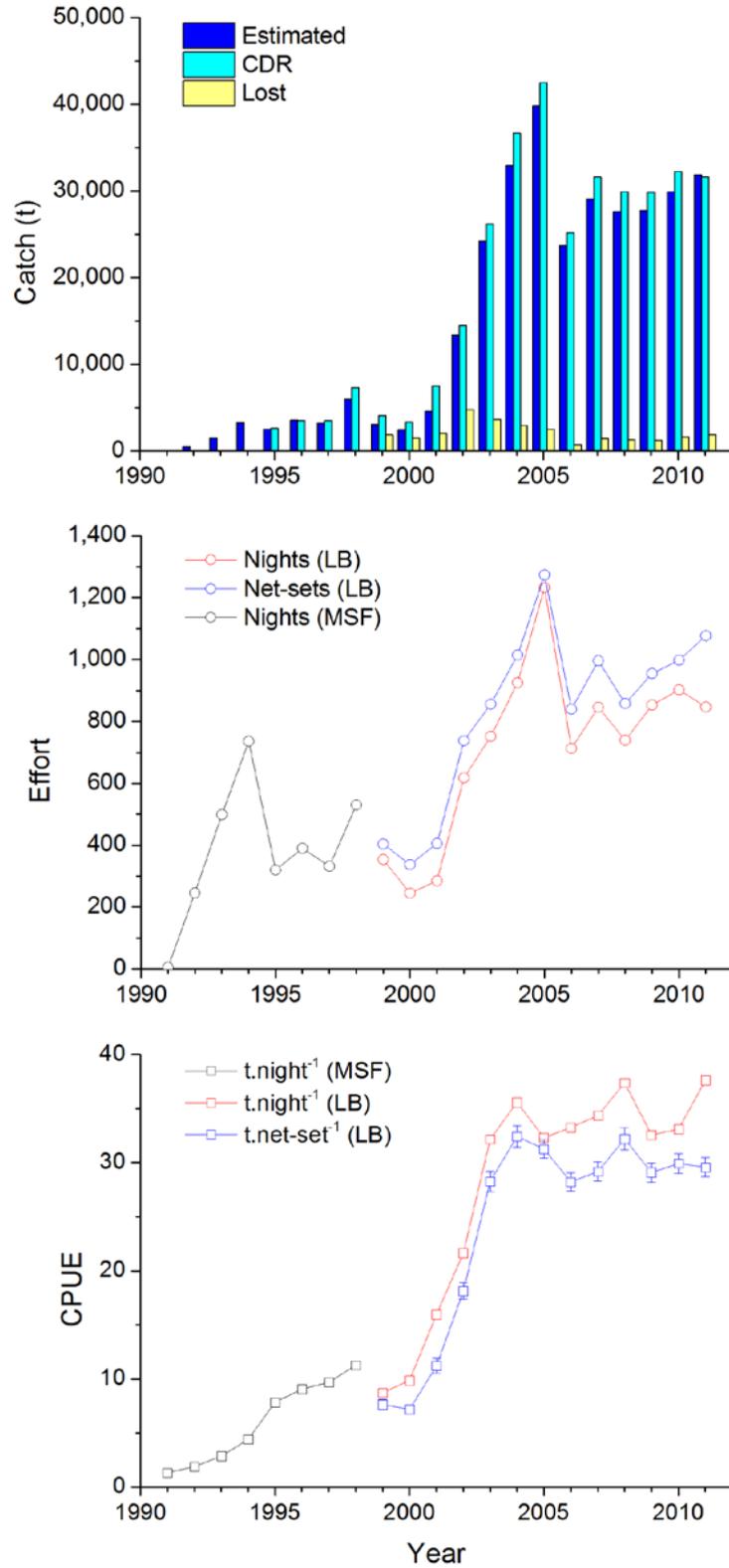


Figure 2-1. Total catches (logbooks, CDR) and estimated lost catches, fishing effort (nights, net-sets), and mean annual CPUE (nights, net-sets,  $\pm$ SE). Data prior to 1999 is derived from marine scalefish records, specific SASF logbooks were introduced in 1999.

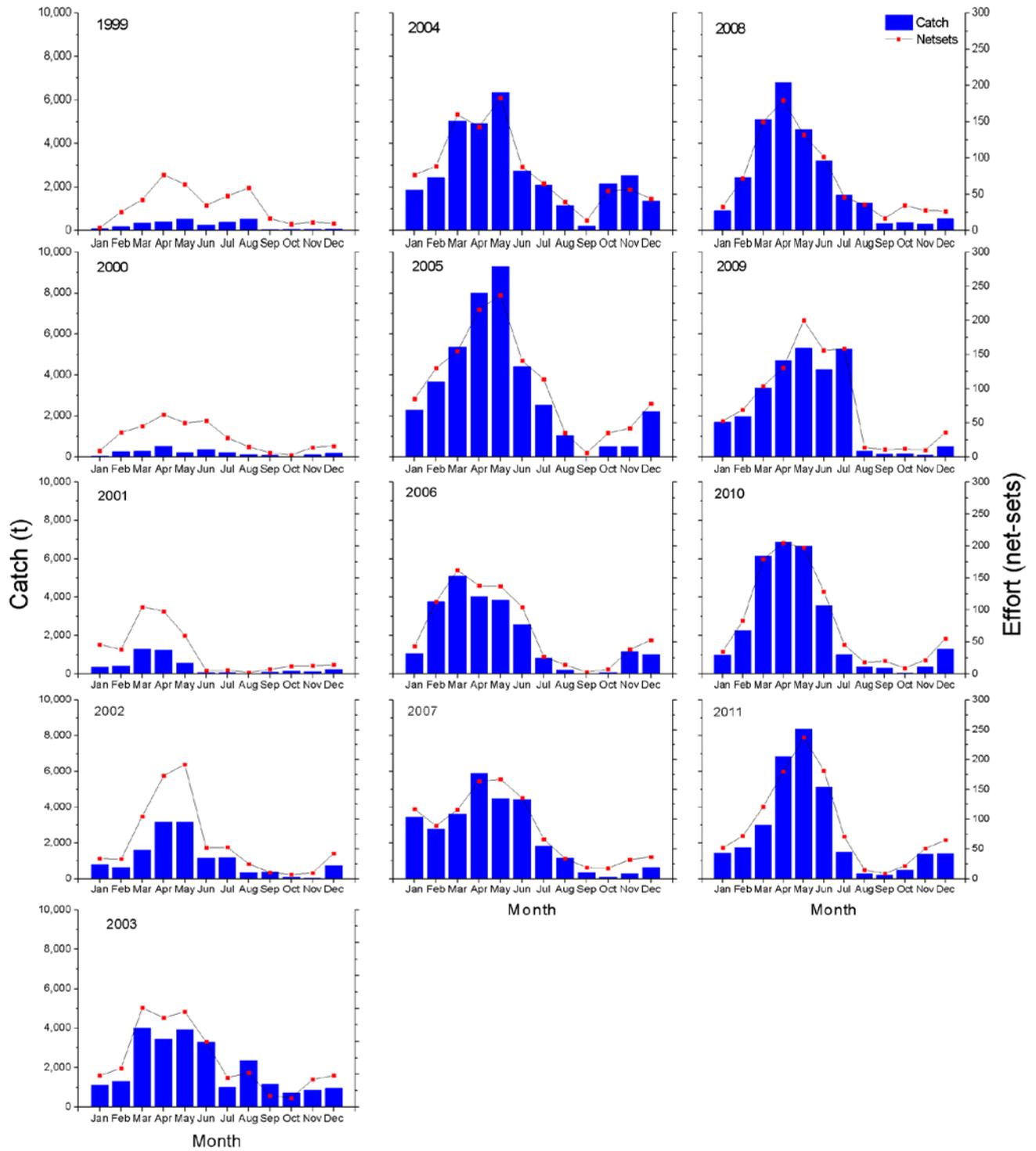


Figure 2-2. Intra-annual patterns in catch (blue bars) and effort (red circles) between 1999 and 2011.

### *Spatial patterns*

There is no record of fishing taking place outside of the three regions considered in this report: Western, Spencer Gulf and Eastern (Figure 1-2). Most fishing has been conducted in Spencer Gulf, although the Western Region was significant in some years (Figure 2-3, 2-4).

In 1991, a small amount of fishing (<10 t) was conducted in Spencer Gulf and the Eastern region. From 1992, up until the first mortality event in 1995, most sardines were taken from Spencer Gulf (Figure 2-3). In 1996 and 1998, most of the catch was taken from the Western region. The catch from the Western Region declined between 1998 and 2000 (Figure 2-3, 2-4). In 2002, ~90% of the catch was taken north-east of Thistle Island and ~9.5% was taken off Coffin Bay (Figure 2-4). In 2003, as the TACC increased the fishery expanded northwards in Spencer Gulf and an increasing proportion of the catch taken closer to shore. Further spatial expansion occurred during 2004, and a large proportion of the total catch was taken near Wedge Island and east of Althorpe Island. Significant catches were also taken along the northern coast of Kangaroo Island, between Cape Borda and Marsden Point. In 2005, the TACC was set at 50,000 t and the spatial extent of the SASF expanded substantially. The eastern Great Australian Bight and inshore areas around the Sir Joseph Banks Group and along the west coast of Spencer Gulf were fished more heavily than in previous years. The TACC was reduced to 26,000 t in 2006 and the area fished contracted, with the largest catches taken from southern Spencer Gulf, near Thistle, Wedge and Althorpe Islands and further north near Arno Bay. Relatively large catches were also taken in Investigator Strait. Between 2007 and 2009, the area fished remained relatively consistent with the majority of fishing continuing to occur in southern Spencer Gulf (Figure 2-5). Since 2010, when additional quota was allocated with conditions that it be taken outside of the Spencer Gulf Region, a significant amount of catch has been taken from Investigator Strait, Gulf St Vincent and waters between Anxious Bay and Flinders Island

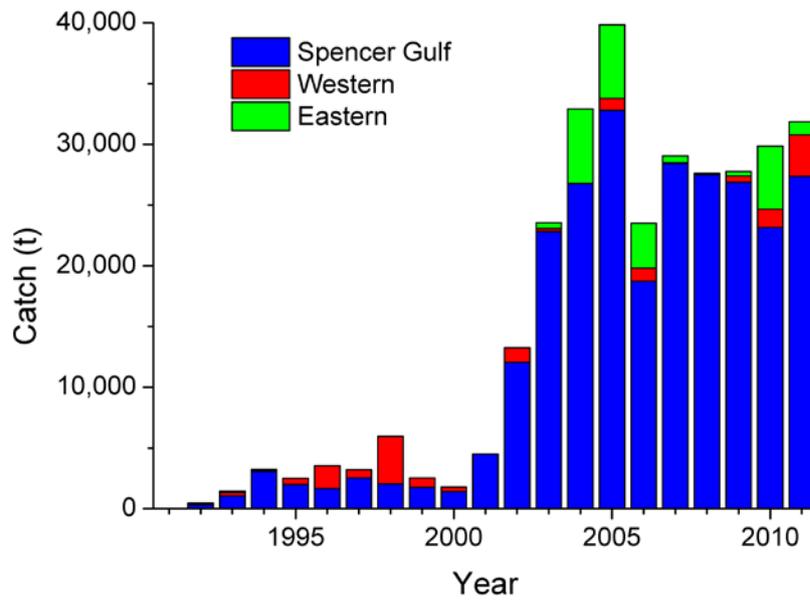


Figure 2-3. Spatial trends in catch and between 1991 and 2011.

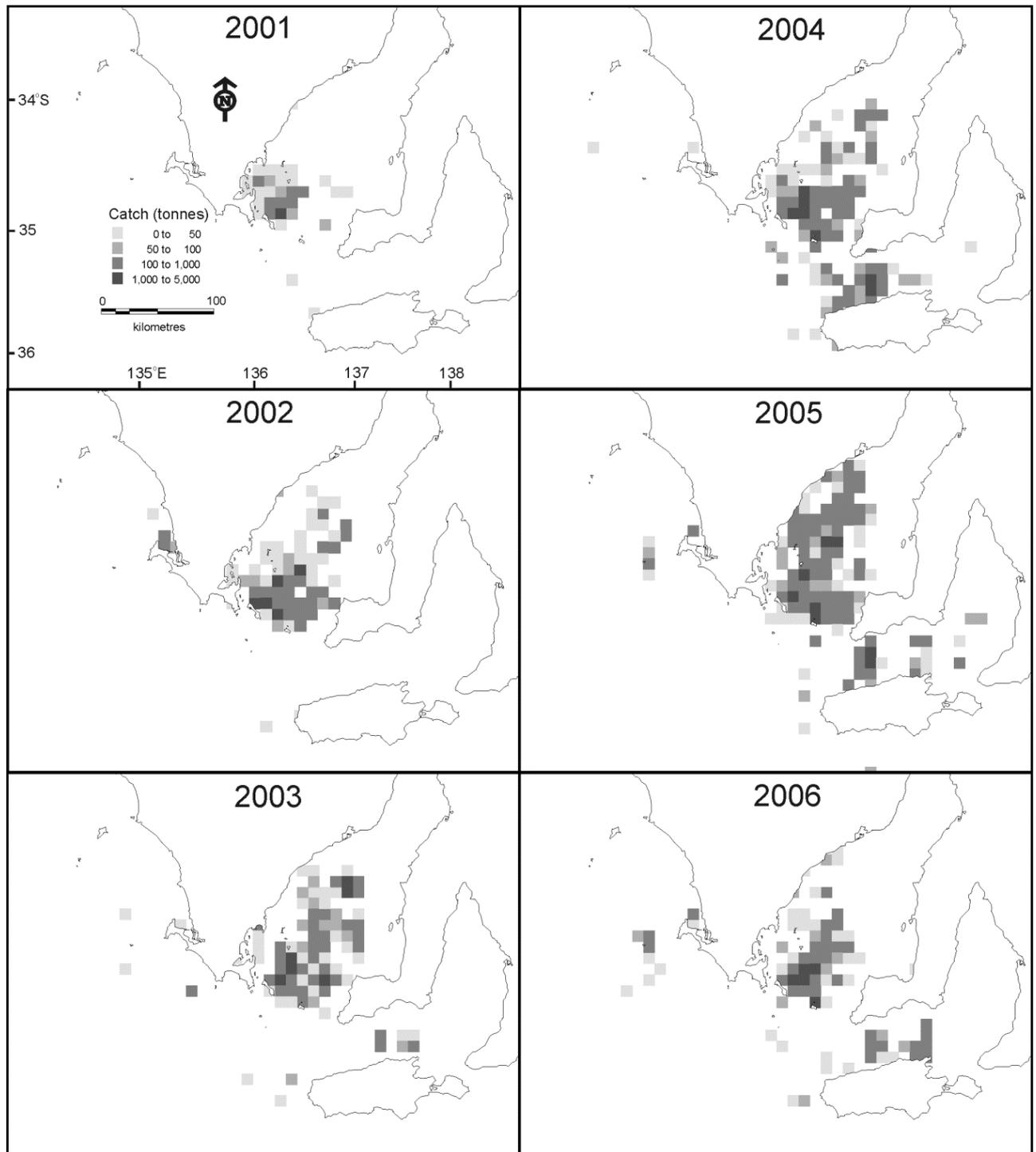


Figure 2-4. Spatial trends in annual catches between 2001 and 2006.

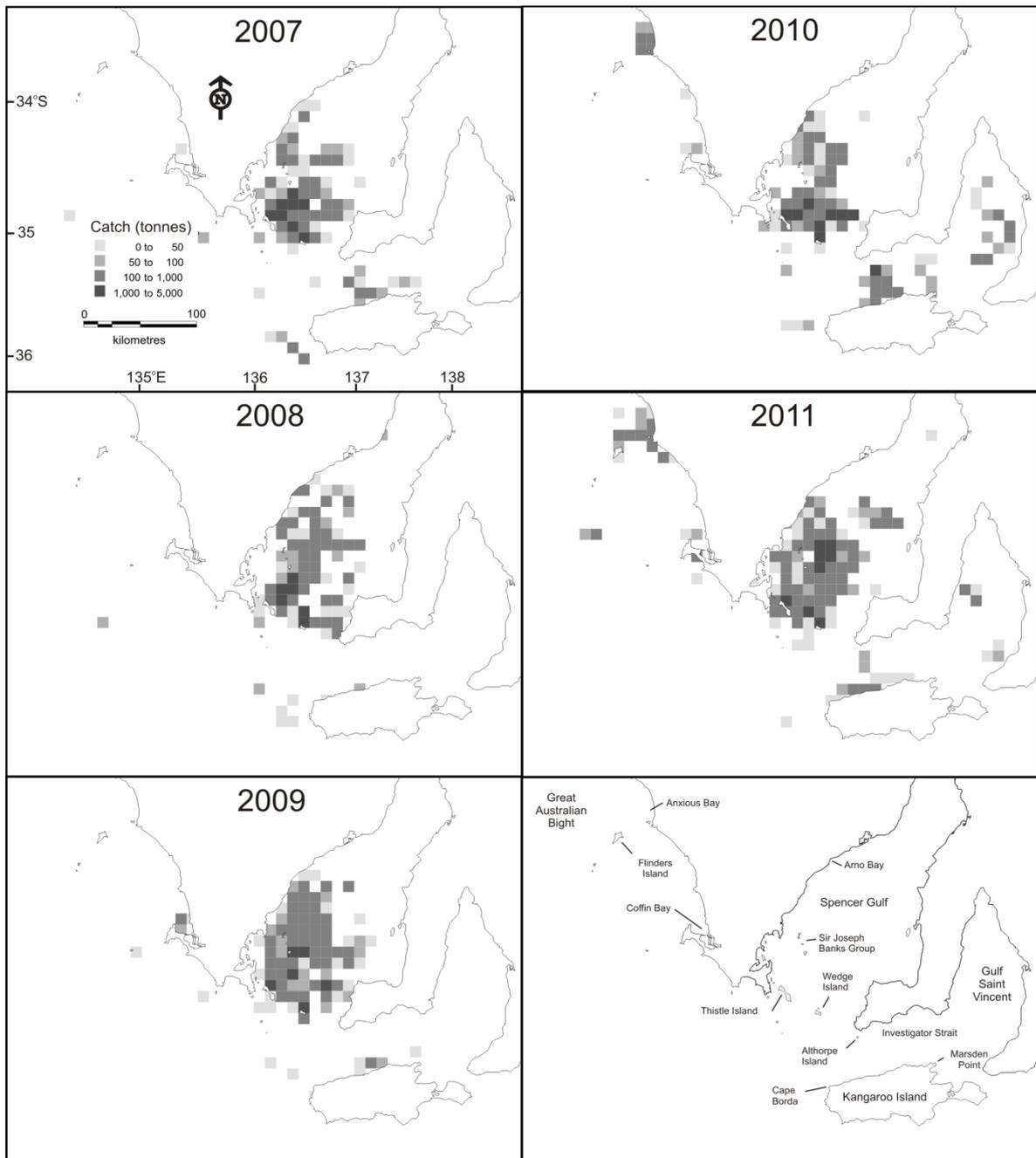


Figure 2-5. Spatial trends in annual catches between 2007 and 2011.

### 2.3.2 Catch composition

#### *Size frequency*

Between 1995 and 1999, sardine taken from Spencer Gulf were mostly 120-170 mm FL, with modes at 130-140 mm (Figure 2-6). Between 2000 and 2002, sardine were mostly >150 mm FL with modes between 160 and 170 mm FL. In 2003 and 2004, catch samples were bimodal as significant quantities of juveniles (80-120 mm) were caught in addition to adults (150-180 mm). Prior to 2003, no catch samples included sardine  $\leq 100$  mm FL. Between 2005 and 2009, size distributions for commercial catches remained stable with a mode at 150 mm FL and fish ranging from 120 to 200 mm. In 2010, the modal size of fish remained the 150 mm FL class but for the first time since 1999 there were more fish below than above the mode (Figure 2-6). The mode has subsequently continued to decline being 140 and 130 mm FL in 2011 and 2012 respectively (Figure 2-6, 2012 January – June only).

In the Western Region, sardine of 150-180 mm FL dominated catches between 1995 and 1998 (Figure 2-7). In 1999, after the second mortality event the modal length fell to 130 mm FL, but increased to 150 mm FL in 2000. Catches from the Western Region were bimodal in 2002 and 2003, but sample sizes were relatively small. In 2004, the modal size was 140 mm FL and this increased to 170-190 mm in 2005-07. In 2008, the modal length was 160 mm FL but sample size was again small in this year. Since 2010, the modal size for fish from the Western Region has remained 170 mm FL, although an increasing proportion has been juvenile sized fish (Figure 2-7). Catches from the Eastern Region were comprised of relatively large sardines in 2003 (mode 190 mm FL) with a smaller mode (~150 mm FL) in 2006 (Figure 2-8). Since 2009, fish from the Eastern Region have mostly been juveniles (<150 mm FL, Figure 2-8).

#### *Proportion of immature fish*

Along with decreasing average fish lengths there has been an increasing proportion of the juvenile fish in the catch since 2005 (Figure 2-9 and 2-10). Significant proportions of the catch from Spencer Gulf were immature between 1995 and 2000 (Figure 2-10), however catches during this period were relatively small and the fishery was still developing. Between 2001 and 2009 (the period of the fishery expansion and subsequent stability at ~30,000 t TACC), the proportion of immature fish remained below ~30%. Since 2010, the immature proportion of the catch has increased markedly to be >80% in both 2011 and the first six months of 2012. Recent catches from the Eastern Region have also been dominated by juvenile fish with only catches from the Western Region containing greater than 50% mature fish.

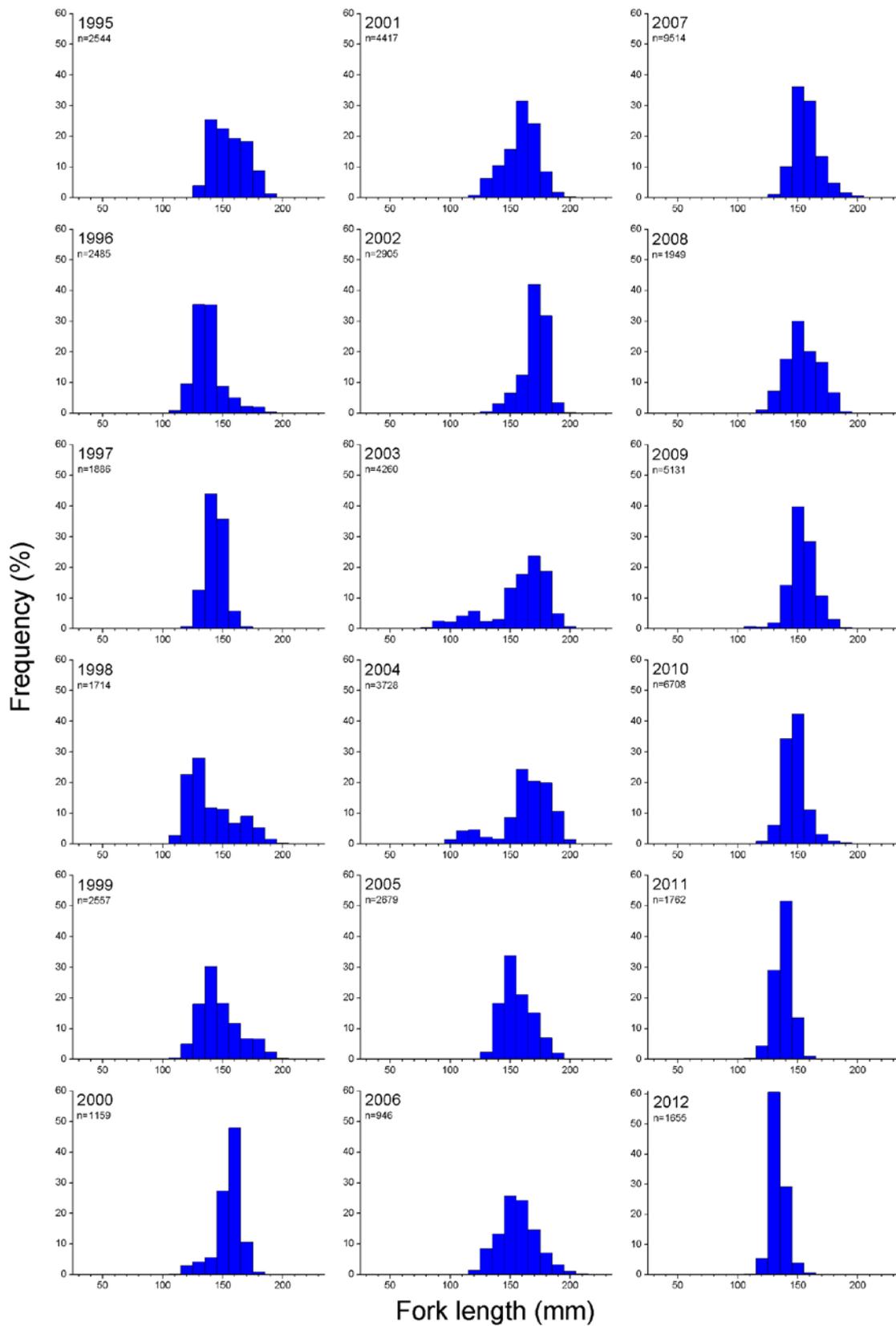


Figure 2-6. Length frequency distributions for commercial catch samples from Spencer Gulf between 1995 and 2012 (January to June only for 2012).

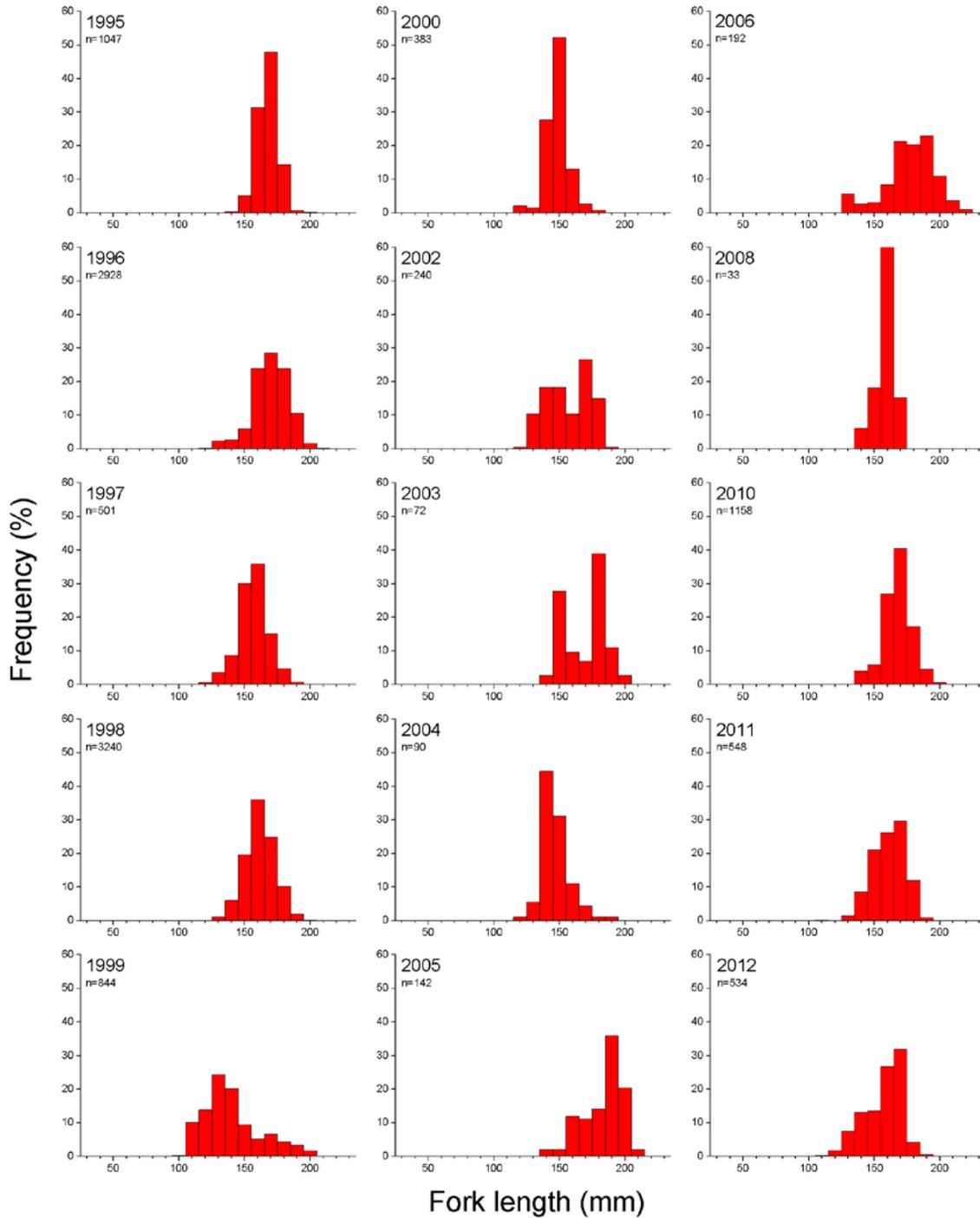


Figure 2-7. Length frequency distributions for commercial samples from the Western Region between 1991 and 2012 (January to June only for 2012).

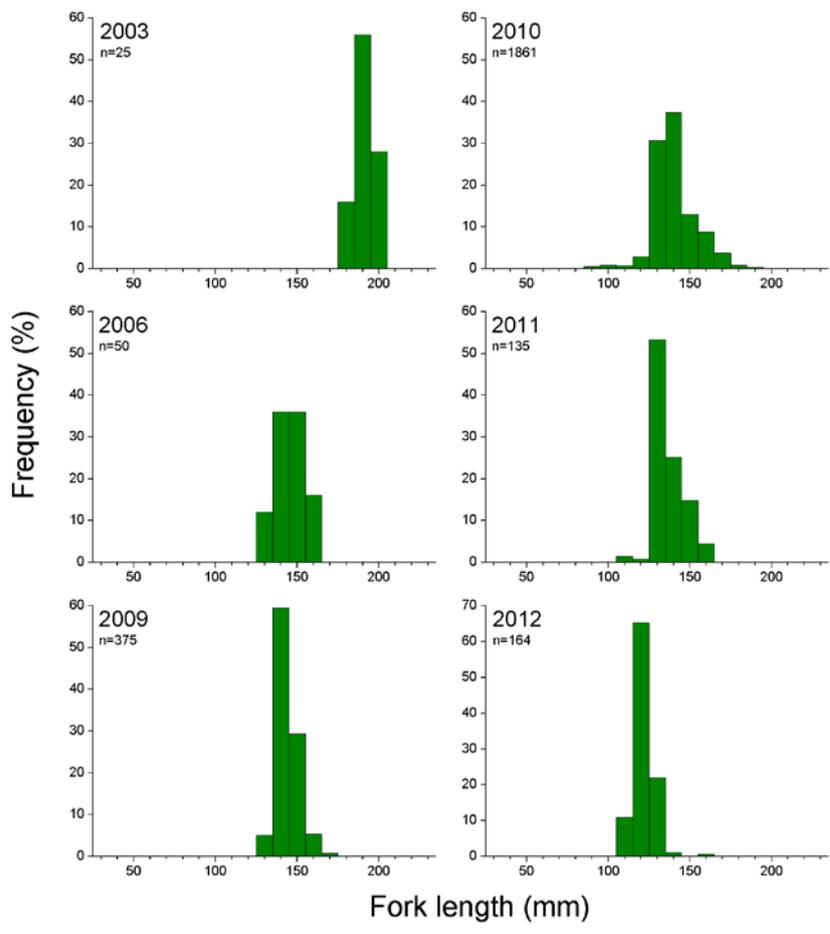


Figure 2-8. Length frequency distributions for commercial samples from the Eastern Region between 2003 and 2012 (January to June only for 2012) for years when samples were available.

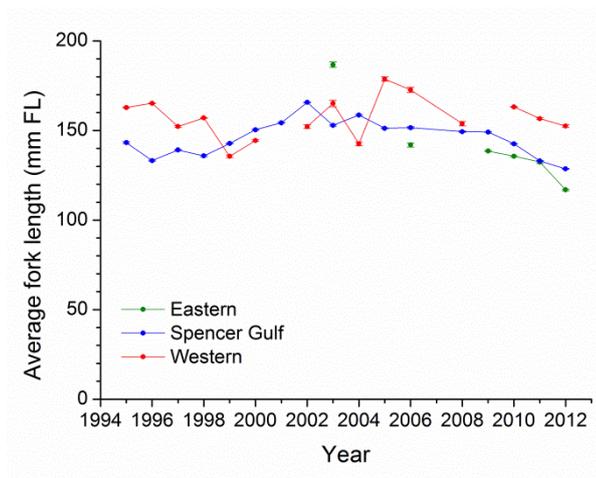


Figure 2-9. Average fork length (FL) by year for the three regions of the SASF, error bars are standard error (January to June only for 2012).

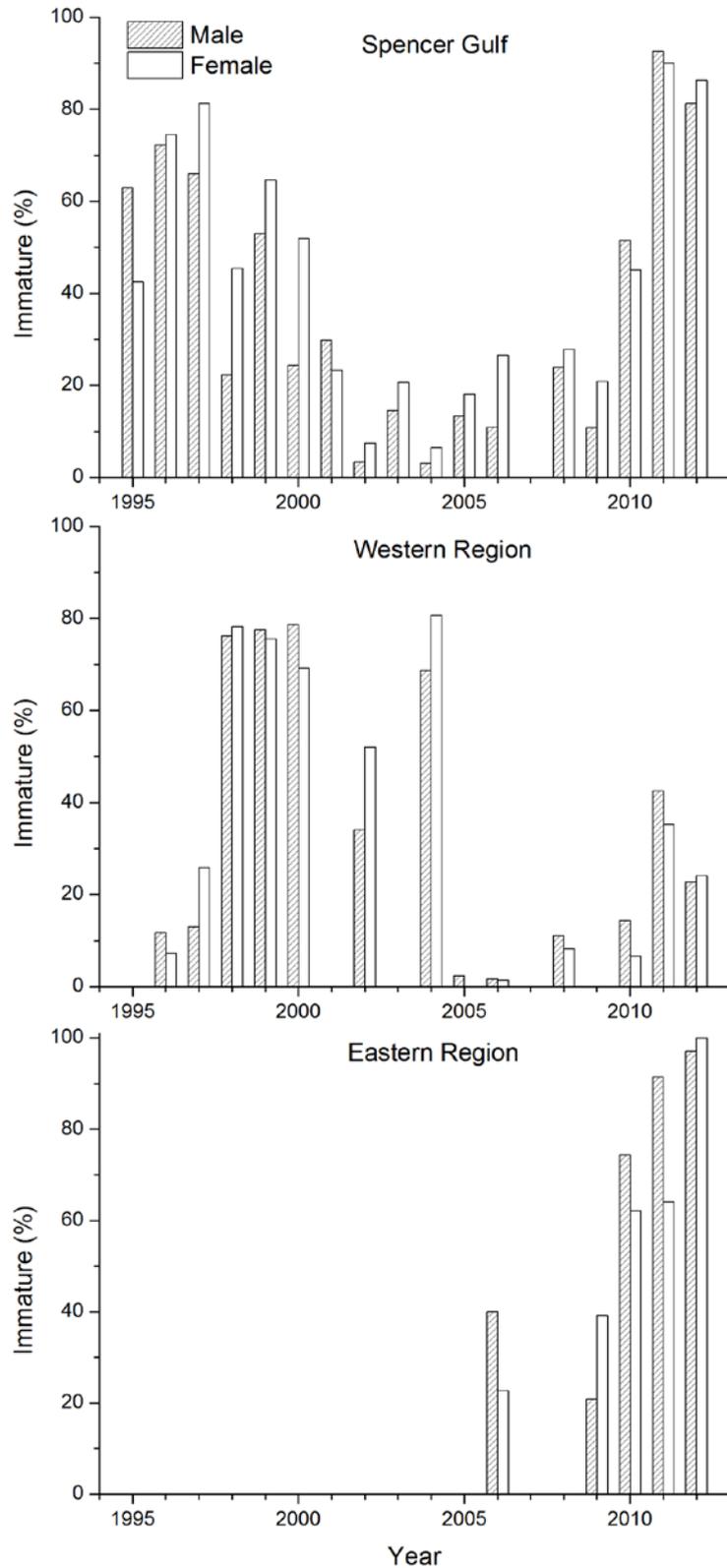


Figure 2-10. Proportion (by number) of catch immature by region between 1995 and 2012 (January to June only for 2012), fish of indeterminate sex are excluded.

### *Age structure*

Age structures from commercial catches in Spencer Gulf were available from 1995 to 2012, with the exception of 2007 when no otoliths were collected. Ages ranged from 1+ to 6+ years. In 1995, 2+ and 3+ year olds dominated catches, but in 1996-1998, 1+ and 2+ year olds were most abundant. In 1999, 2+ year olds (fish that were juveniles in 1998 and not affected by the mortality event) dominated the catch. Fish that were spawned during 1997 and 1998 continued to dominate catches as 3+ year olds in 2000. From 2001 to 2010, 3+ year olds dominated the catch in all years except 2005 and 2006, when 2+ year olds were most abundant in catch samples (Figure 2-11). Since 2011, 2+ year olds have dominated the catch from Spencer Gulf comprising more than 60% of the catch.

### *Sex ratio*

Mean sex ratios from commercial catch samples from Spencer Gulf and the Western Region from 1995 to 2012 are shown in Table 2-1 and 2-2. Females were more abundant than males in most seasons in both locations. Males were more abundant than females in catches from Spencer Gulf in 2000, 2006, 2010 and 2011 (Figure 2-13) and the Western Region in 2002 and 2003. Females comprised more than 60% of the catch in Spencer Gulf in 1995, 2001 and 2005 and in the Western Region in 1995, 1996, 1999, 2004, 2005, 2006, 2008 and 2010. When all seasons were combined, females were more abundant than males in all months, both in Spencer Gulf and in all regions combined. Sex ratios in all regions combined and in Spencer Gulf were significantly different in all months except October, with females dominating. In all regions combined, sex ratios were significantly different in most seasons with females predominating in all seasons, except 2000.

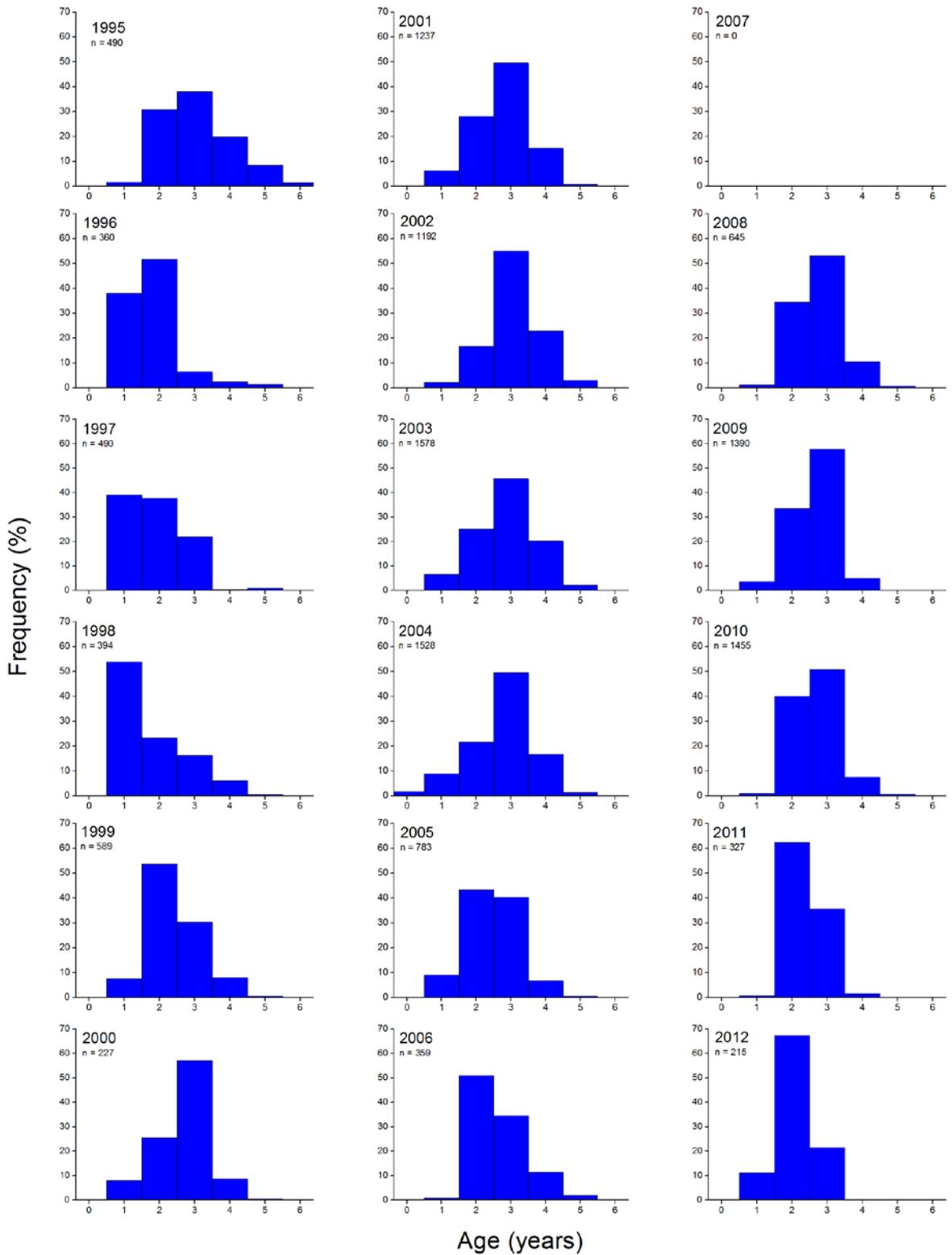


Figure 2-11. Age distributions for commercial catch samples from Spencer Gulf between 1995 and 2012 (January to June only for 2012). Note that no data were available for 2007. Ages are derived from otolith weight age relationships calculated for all years from readability 1 and 2 otoliths and applied to all weighed otoliths of readability 3 or poorer for that year.

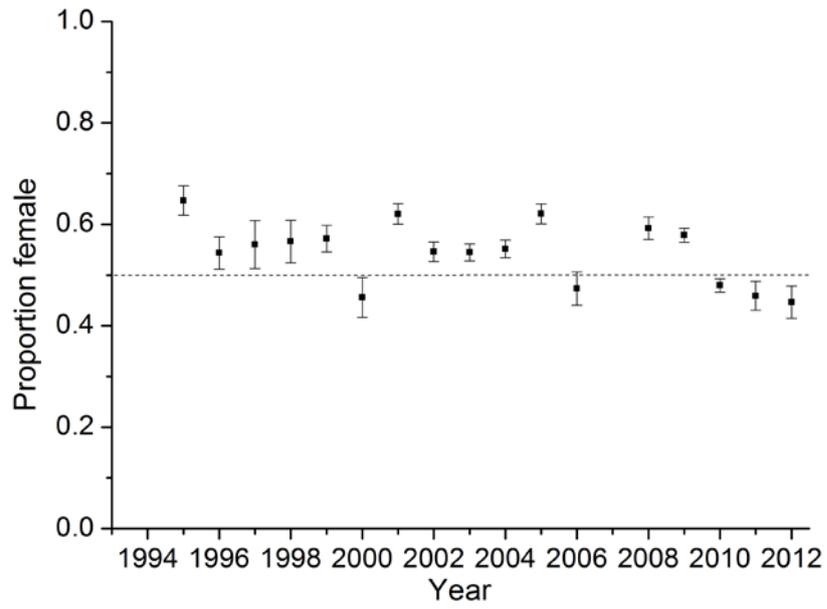


Figure 2-12. Sex ratio ( $F/(F+M)$ ) of commercial catch samples from Spencer Gulf from 1995-2012 (January to June only for 2012). Error bars are 95% confidence intervals.

Table 2-1. Mean annual sex ratios for commercial catch samples from Spencer Gulf (January to June only for 2012). (\*P-values <0.0031 represent evidence of a significant difference in sex ratio after Bonferroni adjustment).

Year	Female (n)	Male (n)	nf/(nm+nf)	p-value
1995	713	389	0.65	<0.001
1996	525	441	0.54	0.008
1997	251	197	0.56	0.012
1998	320	245	0.57	0.002
1999	809	606	0.57	<0.001
2000	289	345	0.46	0.029
2001	1423	870	0.62	<0.001
2002	1469	1221	0.55	<0.001
2003	1958	1637	0.54	<0.001
2004	1796	1461	0.55	<0.001
2005	1502	917	0.62	<0.001
2006	429	477	0.47	0.118
2007	-	-	-	-
2008	1144	788	0.59	<0.001
2009	2907	2115	0.58	<0.001
2010	2703	2930	0.48	0.003
2011	561	661	0.46	0.005
2012	431	534	0.45	0.001

Table 2-2. Mean annual sex ratios for commercial catch samples from West Coast Coffin Bay (January to June only for 2012). (\*P-values <0.0036 represent evidence of a significant difference in sex ratio after Bonferroni adjustment).

Year	Female (n)	Male (n)	nf/(nm+nf)	p-value
1995	534	339	0.61	<0.001
1996	978	608	0.62	<0.001
1997	54	46	0.54	0.484
1998	768	683	0.53	0.027
1999	291	160	0.65	<0.001
2000	78	61	0.56	0.175
2002	73	88	0.45	0.270
2003	24	48	0.33	0.006
2004	31	16	0.66	0.040
2005	99	42	0.70	<0.001
2006	134	58	0.70	<0.001
2008	24	9	0.73	0.014
2010	732	424	0.63	<0.001
2011	323	221	0.59	<0.001
2012	249	241	0.51	0.752

## 2.4 Discussion

One of the most notable features of the SASF has been its rapid growth. The TACC in 2012 (34,000 t) was 34 times the TACC in 1992 (1,000 t) and more than eight times the TACC in 2000. The occurrence of this rapid growth despite the impacts of two mass mortalities, each of which are thought to have killed more fish than any other single species mortality event ever recorded (Jones et al. 1997; Ward et al. 2001), provides evidence of the success of the assessment and management arrangements that have been established for the SASF.

Another notable feature of the SASF is the stability in catches that has been achieved in recent years. This stability has been achieved by establishing a harvest strategy that addresses the imprecision in estimates of spawning biomass obtained using the DEPM. Under the previous harvest strategy for the SASF, and the management arrangements for many other sardine fisheries (e.g. US Pacific Sardine Fishery, Hill et al. 2005, 2008), TACCs can vary dramatically among years in response to changes in estimates of spawning biomass that may often be more reflective of the imprecision in the DEPM rather than of real changes in stock size.

The concentration of fishing effort in a relatively small proportion of the total area over which the managed population is distributed has been a notable feature of the SASF. In recent years (since 2010) increasing effort has occurred outside of Spencer Gulf. Management incentives of additional quota to be caught outside of traditional fishing areas have contributed to this shift in effort, although the larger size fish available on shelf waters of the Western Region may also provide an incentive.

The declining size and age structures since 2010, particularly in southern Spencer Gulf is of concern. When catches increased rapidly in 2003 and 2004 catches included a small proportion of juvenile sardine but the modal size was above 150 mm FL. Between 2005 and 2009, over a period of high catches, modal fish lengths (>150 mm), ages (2-3 year olds) and the proportion of immature fish (<30%) in the catch remained reasonably stable. However, catch samples since 2010 have included larger proportions of small (<140 mm), young (2+ year old) and immature (>80%) fish. This pattern has been particularly evident in southern Spencer Gulf. The driving force behind this change in catch composition is not entirely clear. There is some evidence that growth rates have fallen in recent years (Chapter 3), which may partially explain the reduction in fish lengths. However, as the age

of fish taken has also declined and the proportion of immature fish in the catch has increased, it seems likely that the high proportion of the total catch taken from southern Spencer Gulf has at least contributed to the issue. This finding suggests that consideration may need to be given to establishing an enhanced spatial management framework for the SASF which provides the capacity to increase the proportion of the TACC taken outside southern Spencer Gulf.

## **3.0 AGE, GROWTH, CONDITION AND REPRODUCTION**

### **3.1 Introduction**

This chapter describes the age, growth, condition and reproductive patterns of sardine in South Australian waters. Methods described in this chapter were used to develop the catch-at-age data presented in Chapter 2. Age, growth and reproductive data are key inputs to the population model presented in Chapter 5.

### **3.2 Methods**

#### 3.2.1 Age-determination

##### *Otolith preparation and interpretation*

Sagittal otoliths were collected from sub-samples of the commercial catch sample (n = 10-20) and from fishery-independent samples. Otoliths were soaked overnight in 10% sodium hypochlorite solution to remove excess tissue, rinsed in distilled water and dried in IWAKI™ plastic microplates. Translucent zone counts were made for one whole otolith from each fish under reflected light against a flat black background (Butler et al. 1996).

##### *Readability indices (RI)*

Sardine otoliths were classified as 1 = excellent, 2 = good, 3 = average, 4 = poor and 5 = unreadable based on standard criteria relating to their interpretability (see Rogers and Ward 2007).

##### *Relationship between age and otolith weight and length*

Three methods of assigning age to fish were tested, these were: (1) decimal age calculated from the date of capture and an assumed universal birthdate from otoliths with readability indices 1 or 2; (2) determination of an age-otolith weight relationship estimated from otoliths with readability indices of 1 or 2, that was then applied to all otoliths with recorded weights; and (3) generation of an age-length key from fish with readability indices of 1 or 2 and then applying it to all fish with recorded lengths.

##### *Decimal age estimated from annuli counts*

To estimate decimal age for adults with a translucent zone count of one or more an arbitrary birth-date of March 1 was assigned, representing the time of peak spawning. The midpoint of translucent zone formation was assumed to be mid-winter (Rogers and Ward 2007a). Decimal age ( $A$ ) was then:

$$A = \begin{cases} (\alpha - \beta_p)/365 + TZC + 0.334 & \alpha \leq \beta_s \\ (\alpha - \beta_s)/365 + TZC + 0.334 & \alpha > \beta_s, \end{cases}$$

where  $\alpha$  is the date of capture,  $\beta_s$  is the assumed translucent zone formation date from the same year as  $\alpha$ ,  $\beta_p$  is the assumed translucent zone formation date from the previous year,  $TZC$  is the translucent zone count and 0.334 (4 decimal months) adjusts for the difference between the assigned birth-date and the approximate timing of the first translucent zone.

#### *Age estimated from otolith weight*

The relationship between age and otolith weight was determined using a generalised linear model (McCullagh and Nelder 1983) fitted to decimal age data from those otoliths with readability scores of 1 and 2. A Gaussian GLM with an identity link function was used to estimate age from otolith weight. The low number of readability 1 and 2 otoliths in some years (Table 3-2) prevented the consideration of the season of capture (biological year July to June) as a covariate in this analysis.

Aged otoliths from commercial catch samples between 1995/96 and 2010/11 and from fishery-independent samples between 1997/98 and 2004/05 were pooled for the analysis. Due to the change in the spatial patterns of fishing over time it was not possible to separate annual effects from any regional effects (i.e. region and season were confounded), so data from all regions have been used in the analysis.

#### *Age estimated using an age-length key*

An age-length key was constructed from the decimal age of commercial and fishery-independent age samples that had otoliths with readability scores of 1 or 2. Fish were aggregated into 10 mm length bins with all fish below 130 mm or above 180 mm pooled to obtain sufficient sample sizes in all bins. All fish below 100 mm and above 220 mm FL were excluded. Decimal age was rounded to the nearest whole number and an age-length key was generated by dividing the number of fish at each age in each length class by the total number of fish in that length class. The age-length key was then applied to all commercial and fishery-independent length samples to estimate the age structure of the population for use in the stock assessment model (see Chapter 5).

### 3.2.2 Growth

Recent growth rates were investigated using the von Bertalanffy growth model, however, there was insufficient numbers of fish >2+ years old in the samples from 2010/11 and 2011/12 to obtain meaningful estimates of growth parameters.

#### *Annual variability in age, otolith weight and length relationships*

Difficulties with obtaining sufficient numbers of otoliths with readability scores of 1 or 2 in several years prevented the consideration of inter-annual variability in the relationship between age estimate from annuli counts and otolith weight. This relationship between age and otolith weight has been shown to vary between years in sardine and other small pelagics (Worthington et al. 1995; Fletcher 1995; Gaughan 2008). This is a result of reduced sampling of the commercial catch and lack of fishery independent ageing samples in recent years.

As an alternative to direct measures of age we also investigate inter-annual variability in the relationship between otolith weight and fish length as a proxy for growth rate. It is well established that for some species otolith weight is a better measure of fish age than fish length (Beckman 1991; Rogers and Ward 2007; Gaughan 2008). As a result, slow growing fish will have a heavier otolith at a given length than fast growing fish. The slope of the regression can be interpreted as a proxy for growth rate in the form of somatic growth (mm FL) per otolith growth (mg). This analysis was applied to 1+ and 2+ year old fish from Spencer Gulf to reduce the effect of changes in growth rates over time due to slower growth in older fish and/or among year environmental variability.

### 3.2.3 Fish condition

Fish condition indices calculated from fish lengths and weights have been used to assess the health of populations (Blackwell et al. 2000). A number of condition indices have been proposed, both in the fisheries literature (Bolger and Connolly 1989; Froese 2006) and in the general biology literature (Stevenson and Woods 2006).

#### *Le Cren's Condition Index (Kn)*

The 2010 assessment identified a modified version Le Cren's condition index (Blackwell et al. 2000) as the most biologically appropriate of the condition indices examined. Le Cren's relative weight was defined as,

$$K_n = \frac{W}{W_s},$$

where  $W$  is the observed gonad free weight and  $W_s$  is the gonad free weight obtained from a length weight relationship. The standard weight is defined as,

$$W_s = aFL^b,$$

where  $FL$  is caudal fork length in mm, and the parameters  $a$  and  $b$  were estimated using nonlinear regression from all commercial samples obtained in Spencer Gulf with length, weight and gonad weight recorded.

### *Upwelling*

Upwelling occurs off the South Australian coast during summer and autumn and is driven by the prevailing, alongshore south-easterly winds (Ward et al. 2006c). These upwelling events bring cold nutrient rich water to the surface and have been associated with high levels of primary production (van Ruth et al. 2010). The upwelling index was defined as the along shore component of wind stress ( $\tau$ ) from observed wind strength and direction at Neptune Island in southern Spencer Gulf (van Ruth et al. 2010).

$$\tau = \rho_a C_D U |U|,$$

where  $\rho$  is the density of air,  $C_D$  is the drag coefficient and ( $U$ ) is the averaged three hourly wind data. Positive values of  $\tau$  indicate upwelling favourable conditions.

### *Correlation of fish condition with upwelling and environmental covariates*

The relationship between  $K_n$  from commercial samples and environmental covariates was investigated for the upwelling months of November to April using a Gaussian GLM with an identity link function. Environmental covariates considered were mean upwelling of the month of capture ( $\tau_0$ ), 1, 2 and 3 month lags of mean monthly upwelling ( $\tau_1$ ,  $\tau_2$  and  $\tau_3$ ) and mean upwelling between November and April of each year. The season of capture (biological year July to June) was also included as a covariate in the model.

Since the index of upwelling was obtained from Spencer Gulf, only commercial samples obtained from this region were considered. Fishery-independent samples were excluded because they were primarily collected during DEPM surveys undertaken in February and March. Over 13,000 fish were available for this analysis. For large sample sizes AIC tends

to over-fit so the Bayesian Information Criteria (BIC) was used to assess the performance of candidate models (Burnham and Anderson 2002).

#### 3.2.4 Size at maturity

Ovaries were staged macroscopically where stage 1 = immature, stage 2 = maturing, stage 3 = mature, stage 4 = hydrated (spawning) and stage 5 = spent (recently spawned). Testes were staged where stage 1 = immature, stage 2 = mature and stage 3 = mature (running ripe).

The length at which 50% of the population was mature ( $L_{50}$ ) was estimated using Binomial GLM with a logit link function (logistic regression) from fish sampled during the spawning season. The model was fitted to the numbers of immature (stage 1) and mature (stages  $\geq 2$ ) fish in 5 mm size classes, separately for males and females. The use of a GLM allowed weighting by sample size.

Fish of indeterminate sex and those without gonad stage information were excluded along with those fish sampled outside of the spawning season (1 December to 31 March). Sample sizes were insufficient to estimate size maturity in all years or for regions other than Spencer Gulf.

#### 3.2.5 Gonosomatic index (GSI)

Mean monthly gonosomatic indices were calculated from commercial samples using the equation:

$$GSI = \left[ \frac{Gwt}{Fwt_{gonadfree}} \right] \cdot 100$$

where  $Gwt$  is gonad weight and  $Fwt$  is gonad-free fish weight for fish with gonads of macroscopic stages  $\geq 2$ . The mean of all fish above size at maturity was used for both males and females to determine spawning season. It is important to note that it is sometimes difficult to macroscopically distinguish between Stage 2 and Stage 5 gonads in frozen samples.

### 3.3 Results

#### 3.3.1 Age-determination

Between 1994/95 and 2010/11, a total of 20,220 otoliths from commercial and fishery-independent samples were read. Only 0.2% were assigned a Readability Index (RI) score of 1, while 5.6%, 45.6% and 27.7% were assigned scores of 2, 3 and 4, respectively. Approximately 20.9% were assigned an RI of 5 (Figure 3-1, Table 3-1 and 3-2).

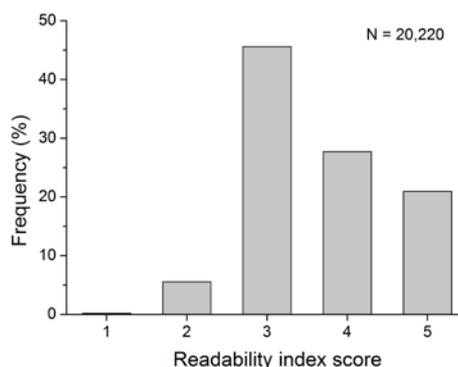


Figure 3-1. Readability Index scores assigned to otoliths from all commercial catch samples between 1994/95 and 2010/11.

Table 3-1. Summary of otolith Readability Index scores for otoliths collected between 1994/95 and 2011/12. #/ denotes incomplete biological years of all samples. \*/ denotes incomplete biological years of commercial samples.

Year	Readability					Total
	1	2	3	4	5	
1994/95#	0	43	130	49	0	222
1995/96	1	92	510	188	3	794
1996/97	0	241	312	49	8	610
1997/98	15	161	472	147	6	801
1998/99	3	61	618	406	21	1109
1999/00	2	88	594	115	2	801
2000/01	0	61	1089	458	72	1680
2001/02	0	90	2500	1248	192	4030
2002/03	0	36	539	545	143	1263
2003/04	1	73	651	568	334	1627
2004/05	9	62	538	385	437	1431
2005/06	0	6	152	114	336	608
2006/07*	0	4	76	67	184	331
2007/08*	0	9	143	183	303	638
2008/09	0	25	281	334	714	1354
2009/10	4	54	418	542	1025	2043
2010/11	0	15	162	169	398	744
2011/12#	1	4	33	39	57	134
All Years	36	1125	9218	5606	4235	20220

Table 3-2. Ring counts of readability 1 and 2 otoliths by season. #/ denotes incomplete biological years of all samples. \*/ denotes incomplete biological years of commercial samples.

Season	0	1	Ring Count	4	5	Total 1&2	Total	%	1&2
1994/95#		27	12	3		1	43	222	19.4
1995/96		27	40	17	9		93	794	11.7
1996/97		198	38	2	3		241	610	39.5
1997/98		123	24	20	8	1	176	801	22.0
1998/99		20	20	21	3		64	1109	5.8
1999/00		13	44	31	2		90	801	11.2
2000/01			18	33	9	1	61	1680	3.6
2001/02			14	46	28	2	90	4030	2.2
2002/03		2	4	13	15	2	36	1263	2.9
2003/04	3	15	26	20	10		74	1627	4.5
2004/05	25	12	11	15	7	1	71	1431	5.0
2005/06	2		1	2	1		6	608	1.0
2006/07*	1		1	1	1		4	331	1.2
2007/08*			4	3	2		9	638	1.4
2008/09		9	11	5			25	1354	1.8
2009/10		12	30	14	1	1	58	2043	2.8
2010/11			6	9			15	744	2.0
2011/12#			4	1			5	134	3.7
Total	31	458	308	256	99	9	1161	20220	5.7

### Otolith weight relationship

The relationship between age from annuli counts and otolith weight provided a reasonable fit to the observed data ( $R^2 = 0.612$ ; Figure 3-2). Ages predicted by the model were biased, with age over-estimated in younger fish and under-estimated in older fish.

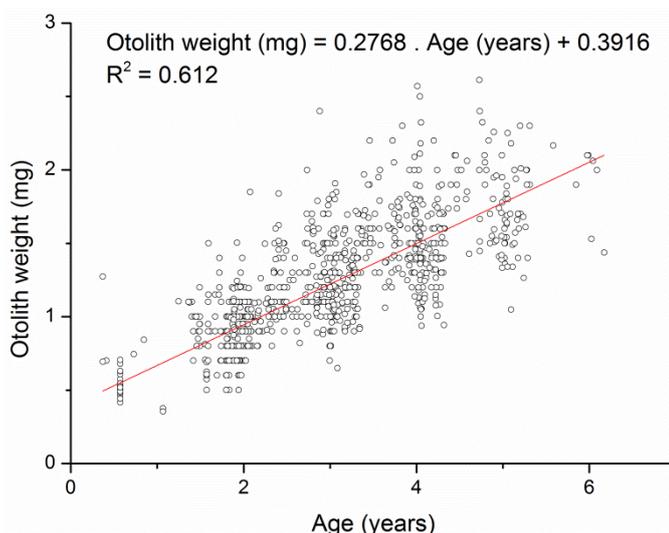


Figure 3-2. Regression of decimal age and otolith weight for sardine otoliths of readability 1 and 2 from commercial and fishery-independent samples collected between 1994/95 and 2011/12.

### *Age-length key*

The age-length key was constructed from all readability 1 and 2 otoliths from commercial and fishery-independent samples obtained between 1994/95 and 2010/11 (Table 3-1). Fish from 100-150 mm were most likely to be 1+ year olds, those 150-160 mm 2+ year olds, 160-180 mm 3+ year olds and 180-220 mm 4+ year olds. The large range of sizes of sardine in each age class demonstrates the difficulties associated with estimating sardine age reliably from length data (Table 3-3).

Table 3-3. Probability of age at length derived from all otoliths of readability 1 and 2 collected between 1994/95 and 2010/11 from both commercial and fishery-independent samples.

Length bin (mm FL)	Age					
	1+	2+	3+	4+	5+	6+
100-130	0.86	0.12	0.02	0.00	0.00	0.00
130-140	0.50	0.39	0.10	0.01	0.00	0.00
140-150	0.26	0.44	0.25	0.05	0.00	0.00
150-160	0.08	0.28	0.36	0.23	0.03	0.01
160-170	0.03	0.20	0.41	0.29	0.06	0.01
170-180	0.01	0.06	0.37	0.35	0.20	0.01
180-220	0.00	0.08	0.23	0.51	0.19	0.00

### 3.3.2 Growth patterns

The growth measure based on the relationship between otolith-weight (OW) and fish length (FL) indicates a pattern of declining growth rate for one and two year old fish since a peak of ~90 mm FL.mgOW in 2003/04 to a low of ~30 mmFL.mgOW in 2011/12. Growth rates measured prior to 2003 should be interpreted with caution due to limitations in the accuracy of the scales used to weigh these otoliths. Otoliths obtained during that period will be re-weighed over the next year.

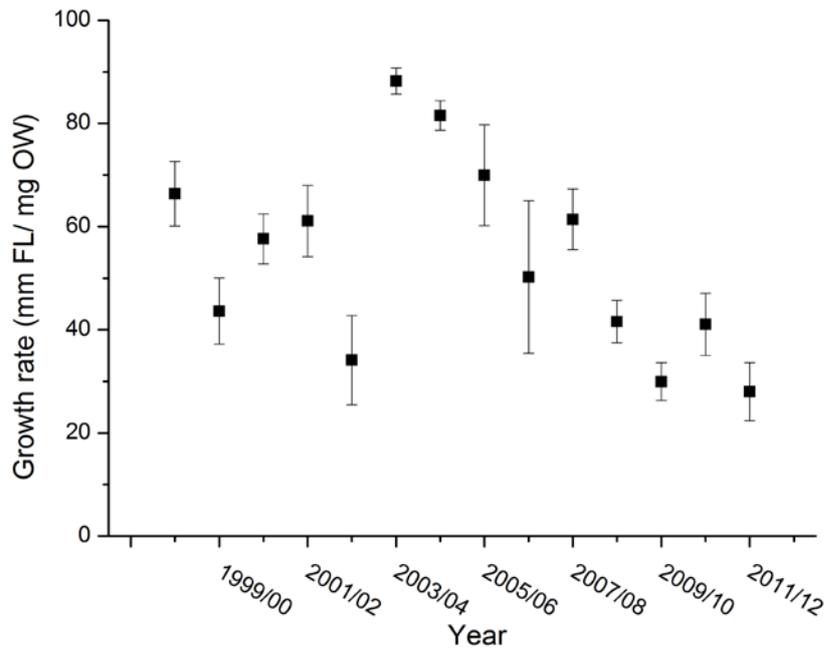


Figure 3-3. Growth rate proxy for 1+ and 2+ year old sardine collected from Spencer Gulf between 1998/99 and 2011/12. Error bars are 95% confidence intervals. N = 6,965.

### 3.3.3 Fish condition

Analysis of fish condition was conducted by biological year (July to June) to allow the correlation with upwelling events that are known to occur in summer-autumn. There was a distinct correlation between fish condition factor and upwelling index in Spencer Gulf when examined by season (Figure 3-4). A one month delay was evident for both minimum and maximum fish condition (September and March respectively) to the minimum and maximum of upwelling index (August and February, respectively; Figure 3-4). Mean seasonal fish condition and upwelling during the upwelling months of November to April did not show any consistent correlation with condition index (Figure 3-5).

GLM results show the correlation of environmental variables to fish condition (Table 3-4). The most significant term in the model was season (biological year), the next most significant term was a 2 month lag of upwelling ( $\tau$ ).

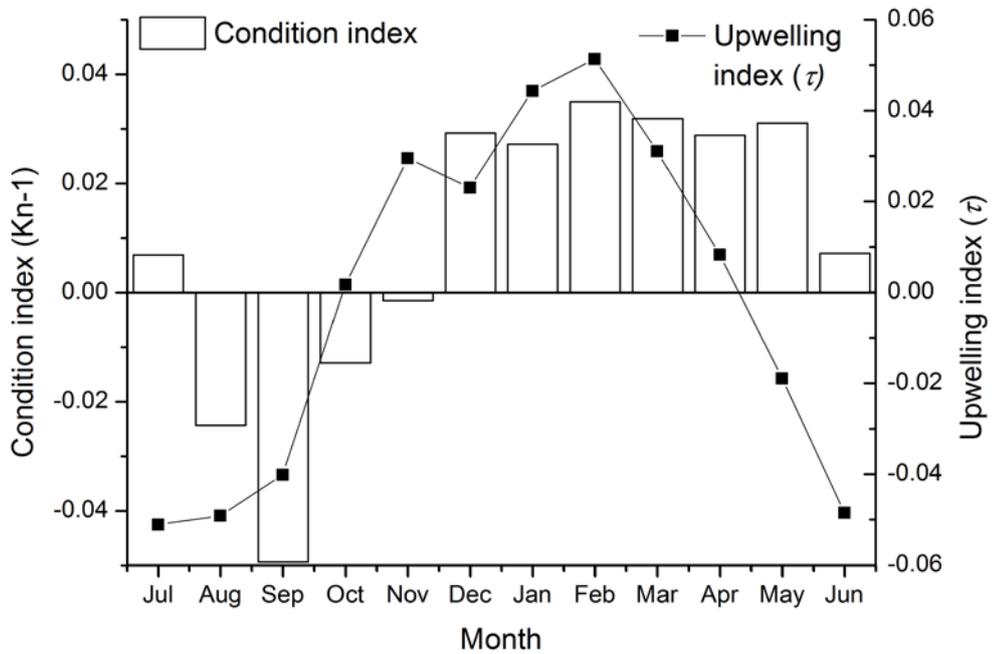


Figure 3-4. Condition index (observed / expected gonad free weight) and upwelling index by month, average across all biological years from 1995/96-2010/11 for Spencer Gulf.

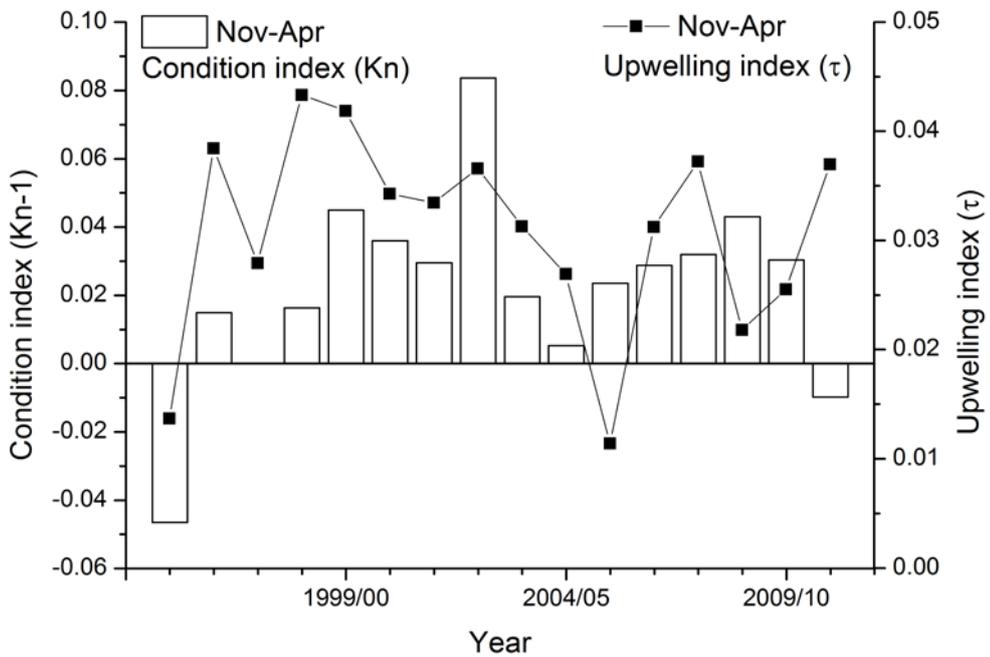


Figure 3-5. Condition index (observed / expected gonad free weight, for fish captured between November and April) and upwelling index by year from 1995/96-2010/11 for Spencer Gulf averaged across upwelling favourable months (November – April).

Table 3-4. Generalised linear model (GLM) results modelling the relationship between fish condition ( $Kn$ ) and indices of upwelling fitted to commercial catch samples obtained from Spencer Gulf between November and April from 1996/97 to 2010/11. Four lags of mean monthly upwelling were included ( $\tau_0, \tau_1, \tau_2$  and  $\tau_3$ ) along with mean upwelling for the season ( $\xi$ ). Biological year (July to June) was also included as a covariate (Season) in the analysis. Notation: P = number of parameters; Dev = residual deviance; AIC = Akaike's information criterion; BIC = Bayesian information criterion;  $\Delta$ BIC is the change in AIC between the best and candidate model. Model 1 is the null model (constant only), model 6 included all main effects except  $\xi$  which was not significant.

Model	P	Dev	AIC	BIC	$\Delta$ BIC
1 $Kn \sim 1$ (null)	1	90.93	-41673	-41665	1884
2 $Kn \sim$ Season	14	82.52	-43327	-43218	331
3 $Kn \sim$ Season + $\tau_2$	15	81.54	-43532	-43415	134
4 $Kn \sim$ Season + $\tau_2$ + $\tau$	16	80.99	-43645	-43521	28
5 $Kn \sim$ Season + $\tau_2$ + $\tau$ + $\tau_3$	17	80.88	-43666	-43534	15
6 $Kn \sim$ Season + $\tau_2$ + $\tau$ + $\tau_3$ + $\tau_1$	18	80.77	-43689	-43549	0

### 3.3.4 Size at maturity

Size at maturity (SAM,  $L_{50}$ ) estimated for sardine from the commercial catch samples from Spencer Gulf between 1995/96 and 2011/12 varied slightly among seasons (Figure 3-6). However, small sample sizes produced wide confidence intervals with no statistically significant differences detected between seasons. All males below 116 mm and females below 118 mm had immature gonads.

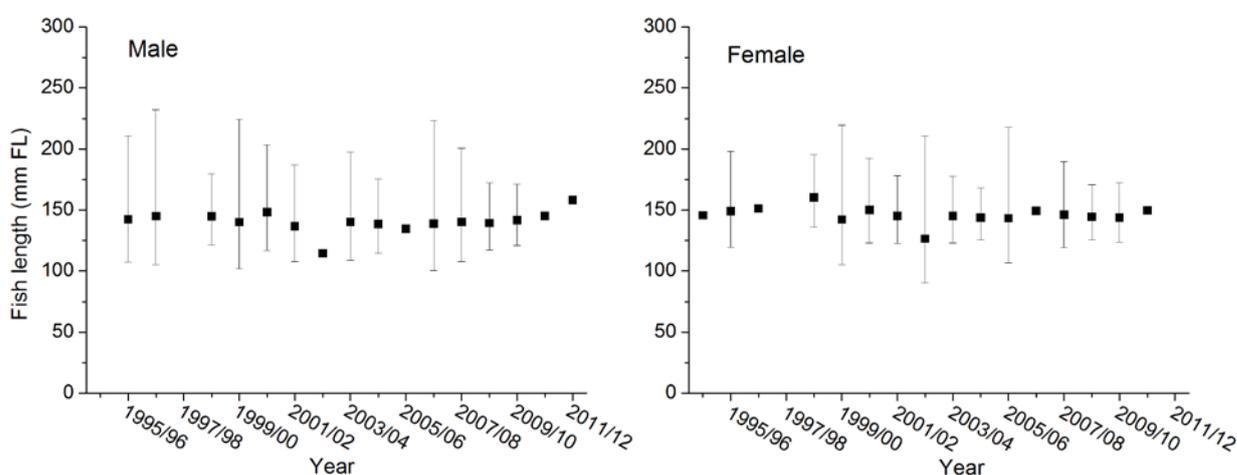


Figure 3-6. Size at 50% maturity ( $L_{50}$ ) for male and female sardine collected in Spencer Gulf by biological year between 1995/96 and 2011/12. Some years omitted due to low sample size. Error bars are 95% confidence intervals, absent where insufficient data were available for estimation.

### 3.3.5 Gonosomatic index (GSI)

There was a large amount of seasonal and regional variability in GSI. However, sample size was variable, with sufficient samples obtained only from Spencer Gulf in most years and West Coast Coffin Bay in a few years. GSI peaked between January and March (Figure 3-7). Higher mean GSI values were observed for males than females, this is due to male gonads not decreasing in size as much as females after each spawning event. Higher mean GSI values were also observed from larger fish.

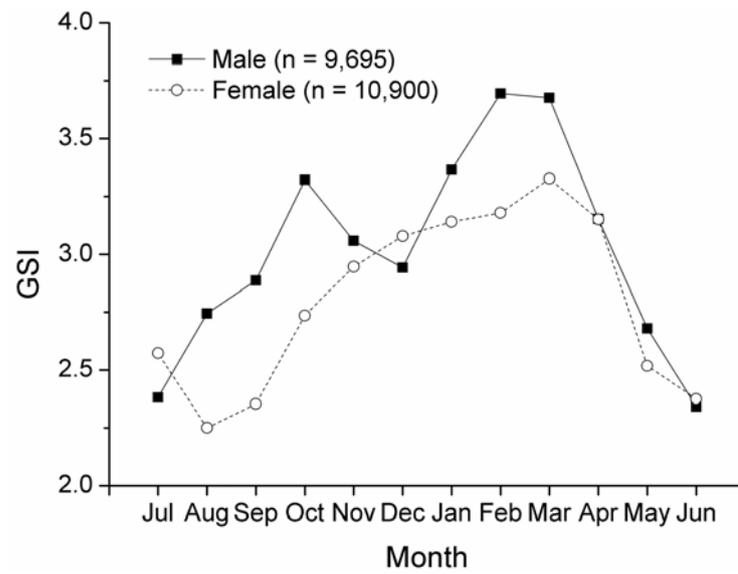


Figure 3-7. Mean monthly gonosomatic index of mature males and females from commercial samples from 1995-2011 combined. Fish below the size at 50% maturity have been excluded.

### 3.4 Discussion

The relatively high level of uncertainty associated with the estimation of the age of sardine from annual growth increments in otoliths has been noted elsewhere (Fletcher 1994; Rogers and Ward 2007; Ward et al. 2010). This issue can be partly overcome by using an age-otolith weight relationship developed from otoliths with high readabilities to estimate the age of sardine with otoliths that are more difficult to read. This approach requires a large number of otoliths to be read each year to provide adequate numbers of otoliths with high readability to determine inter-annual variations in growth rate.

Our analyses suggest that the length-at-age of sardine exhibits considerable spatial and temporal variation. The growth rates of individuals also vary with age, (Rogers and Ward 2007), which reflect the moderate to high growth rates prior to the onset of sexual maturity and slower growth rates as adults. This finding is consistent with the results of Fletcher (1994) who found that fish length displayed no consistent modal progression through samples, whereas, cohorts could be tracked using otolith weight (Rogers et al. 2004). This variability in growth rates limits the effective use of age-length keys for estimating the age of sardine. In Chapter 5, we show that biases in age estimates associated with the use of fish length, rather than otolith weight, can substantially affect the outputs of the age structured model that has been developed for the fishery.

In this report it was necessary to consider an indirect measure of growth rate using the otolith-weight/fish length-relationship. This approach provided evidence of progressively decreasing growth rates over the last nine years. The low growth rates of one and two year old fish taken from Spencer Gulf over the last four years may partially explain the declines in size of fish taken from that region. It is likely that this reduction in growth rates was driven by environmental factors. The summer's of 2010/11 and 2011/12 were characterised by a strong La Niña influence across much of Australia, these weather patterns can have several effects on oceanographic patterns in the Great Australian Bight including suppressed upwelling and an increased penetration of the Leeuwin Current (e.g. Middleton and Bye 2011). However, the reduction in the mean age of fish and increase in the proportion of immature fish taken from southern Spencer Gulf in recent years (Chapter 2) suggests that the high concentration of fishing effort in that region may also be a contributing factor to this reduction in fish size.

The relationship between monthly fish condition and upwelling index supports previous suggestions that the timing of spawning and other aspects of the biology of sardine in South Australia are strongly related to productivity. Like previous studies (e.g. Ward and

Staunton-Smith 2002), our results show that in South Australia sardine spawn mainly in summer and autumn, with the peak occurring in February-March. The high condition factor (which is based on gonad free-weight) during the spawning season when significant energy is being directed at gonad development suggests that upwelling has a strong influence on the total annual energy budget of sardine in South Australia.

Although the most important fishing months are between March and June (i.e. partially overlap with the spawning season), actively spawning females (Stage 4) comprised only a small proportion (<1%) of the catch. This finding provides evidence that spawning sardine may be poorly sampled using purse seine gear (Hewitt 1985) and further demonstrates the need to obtain fishery-independent samples to estimate the reproductive parameters required for application of the DEPM.

## 4.0 TRENDS IN ESTIMATES OF SPAWNING BIOMASS

### 4.1 Introduction

This chapter presents the time series of estimates of sardine spawning biomass for South Australia that have been obtained using the DEPM between 1995 and 2011. This information is used to make inferences about the recent status of the South Australian sardine resource and is a key input to the population model in Chapter 5.

### 4.2 Methods

#### 4.2.1 Daily Egg Production Method

SARDI Aquatic Sciences has used the DEPM to estimate the spawning biomass of sardine in South Australia since 1995. This method relies on the premise that spawning biomass can be calculated from estimates of the number of pelagic eggs produced per day in the spawning area (daily egg production,  $P_0$ ) and the number produced per female (daily fecundity). Spawning biomass ( $B$ ) is calculated according to the equation:

$$B = \frac{P_0 \cdot A \cdot W}{R \cdot F \cdot S}$$

where  $P_0$  is mean daily egg production,  $A$  is the spawning area,  $W$  is the mean weight of mature females,  $R$  is the sex ratio,  $F$  is the mean batch fecundity and  $S$  is the mean spawning fraction (Lasker 1985; Parker 1985; Alheit 1993).

The method used to estimate spawning biomass was recently the subject of a comprehensive review (Ward et al. 2011b) that resulted in minor modifications to the methods used in previous reports (e.g. Ward et al. 2007). Four generalised linear models (GLM), the exponential model (Lo et al. 1985) and the log linear model were tested. The optimal model was determined to be the log linear model applied to all stations with eggs of stages 2 to 11 with one egg added to all day 1 and day 2 densities (Ward et al. 2011b). Spawning area is now calculated using the Voronoi natural neighbour (VNN) method to assign station boundaries objectively. A temperature-dependent model is used to assign ages to eggs based on the sea surface temperature, stage and time of sampling. Egg density is calculated using the log linear model applied to all stations with eggs of stages 2 to 11. One egg is added to all day 1 and day 2 densities. Revised estimates of spawning biomass are presented in this chapter.

### Sampling area

Between 1995 and 2007, annual ichthyoplankton and adult sardine sampling surveys of ~28 days duration were conducted during each spawning season (January-March) from the *RV Ngerin*. Since 2009, surveys have been biennial. The location of plankton sampling stations used in 2011 is shown in Figure 4-1. The number of stations and orientation of transects varied among years as the survey design was refined. During 1995 and 1996, few stations were sampled, as the primary goal was to identify the main spawning area. After 1997, transects were orientated northeast-southwest (*c.f.* north-south) to improve sampling efficiency. In 2006, stations that had not yielded any eggs in the preceding decade were excluded in favour of additional stations in Spencer Gulf.

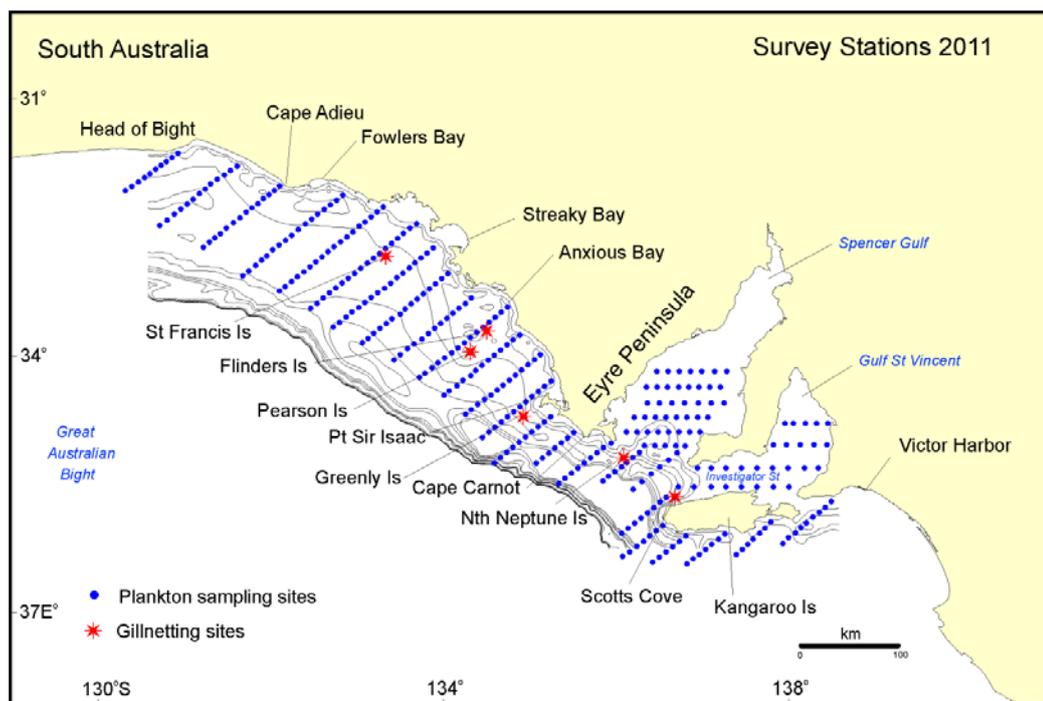


Figure 4-1. Map showing stations where sardine egg and adult samples were collected during the 2011 DEPM surveys.

### Plankton sampling

Plankton samples were collected at each station using Californian Vertical Egg Tow (CaIVET) plankton nets. CaIVET nets had an internal diameter of 0.3 m, 330  $\mu$ m mesh and plastic cod-ends. During each tow the nets were deployed to within 10 m of the seabed at depths <80 m or to a depth of 70 m at depths >80 m. Nets were retrieved vertically at a speed of ~1  $\text{m}\cdot\text{s}^{-1}$ . General Oceanics™ flowmeters were used to estimate the distance

travelled by each net. Samples from the two cod-ends were combined and stored in 5% buffered formaldehyde and seawater.

#### *Egg distribution and abundance*

Sardine eggs were identified, counted, staged and assigned ages according to descriptions and temperature-development keys in White and Fletcher (1996). The number of eggs of each stage under one square metre of water ( $P_i$ ) is estimated at each station according to equation 2:

$$P_i = \frac{C.D}{V}$$

where  $C$  is the number of eggs of each age in each sample,  $V$  is the volume of water filtered ( $m^3$ ), and  $D$  is the depth (m) to which the net was deployed (Smith and Richardson 1977).

#### *Spawning time*

Sardine eggs in each sample were counted and staged according to criteria in White and Fletcher (1996). The age of each developmental stage was estimated using a temperature dependent egg development model that incorporated time of sampling, sea surface temperature and egg stage (Ward et al. 2011b).

The peak daily spawning time calculated from combined data between 1995 and 1999 was 0200 hours (Ward et al. 2001b). This was used as the standard spawning time in subsequent egg production and spawning biomass calculations.

#### *Spawning area*

A key premise of using the DEPM is to accurately estimate the entire spawning area (Lasker 1985; Somarakis 2004). After the surveys are completed, the survey area is divided into a series of contiguous grids approximately centred on each station using the Voronoi natural neighbour method (Figure 4-2). The area represented by each station ( $km^2$ ) was calculated using MAPINFO® software. The spawning area ( $A$ ) is defined as the total area of grids where live sardine eggs were found (see Fletcher et al. 1996).

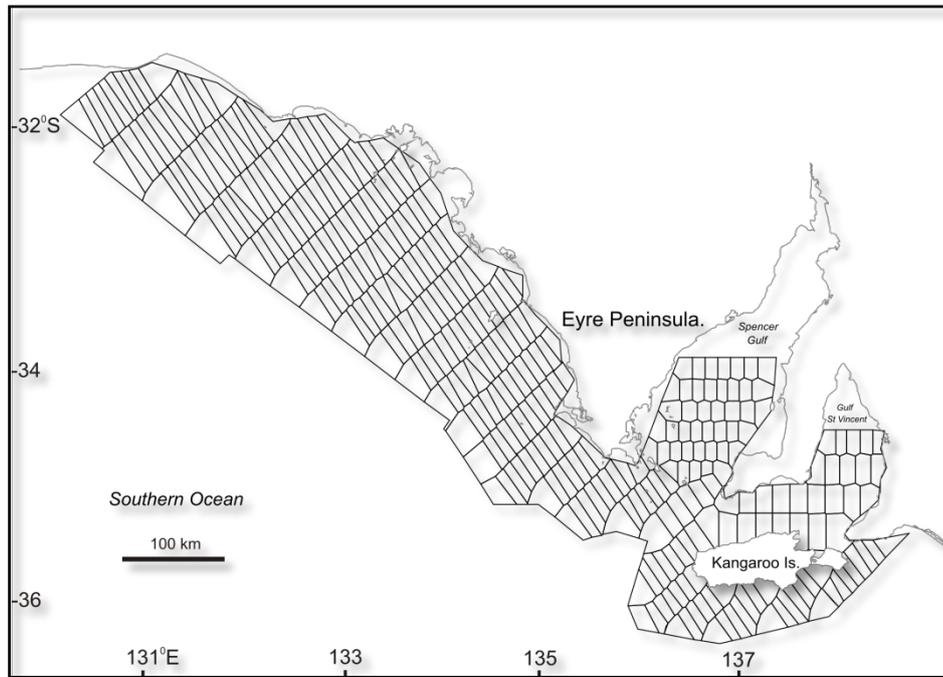


Figure 4-2. Spatial grids generated using the Voroni natural neighbour method to estimate the spawning area of sardine in 2011.

#### *Egg production $P_0$*

Biased mean daily egg production ( $P_b$ ) was calculated by fitting the linear version of the exponential egg mortality model to estimates of egg age and density at each station (Picquelle and Stauffer 1985). To allow the inclusion of data from stations where either day 1 or day 2 eggs were absent, one egg was added to the counts of both day 1 and day 2 eggs at every positive station. The linear version of the exponential egg mortality model is:

$$\ln P_b = \ln(P_i) - Zt$$

where  $P_i$  is the density of eggs of age  $t$  at site  $i$  and  $Z$  is the instantaneous rate of egg mortality.

Estimates of  $P_b$  obtained using the linear version of the exponential mortality model have a strong negative bias, therefore a bias correction factor was applied following the equation of Picquelle and Stauffer (1985):

$$P_0 = e^{(\ln P_b + \sigma^2/2)}$$

where  $\sigma^2$  is the variance of the estimate of biased mean daily egg production ( $P_b$ ).

#### *Adult reproductive parameters*

Between 1995 and 1997, there were difficulties collecting samples of mature sardine. Mid-water trawling and sampling from commercial catches during the spawning season were attempted with minimal success. The resultant paucity of data and uncertainty associated with estimates of adult reproductive parameters reduced confidence in estimates of spawning biomass over this period (Ward et al. 2001a).

From 1998 to 2011, samples of mature sardine were collected during research surveys in the eastern Great Australian Bight, southern Spencer Gulf and the Investigator Strait using surface and sub-surface lights and a multi-panelled gillnet (Ward et al. 2001a). Upon retrieval of the net, fish were removed and dissected by ventral incision. Mature and immature males and females were counted. Mature females were fixed in 5% buffered formaldehyde solution and immature females and males were frozen. Calculations of female weight, sex ratio, batch fecundity and spawning fraction were based on samples collected from Investigator Strait, southern Spencer Gulf and in the eastern Great Australian Bight.

#### *Mean female weight (W)*

Mature females from each research sample were removed from formalin and weighed ( $\pm$  0.01 g).  $W$  was calculated from the average of sample means weighted by proportional sample size:

$$W = \left[ \overline{W_i} * \frac{n_i}{N} \right]$$

where  $\overline{W_i}$  is the mean female weight of each sample  $i$ ;  $n$  is the number of fish in each sample and  $N$  is the total number of fish collected in all samples.

#### *Sex ratio (R)*

For fishery-independent samples,  $R$  was calculated from the average of sample means weighted by proportional sample size:

$$R = \left[ \overline{R_i} * \frac{n_i}{N} \right]$$

where  $n_i$  is the number of fish in each sample,  $N$  is the total number of fish collected in all samples and  $\overline{R}_i$  is the mean sex ratio of each sample calculated from the equation:

$$\overline{R}_i = \frac{F}{(F + M)}$$

where  $F$  and  $M$  are the respective total weights of mature females and males in each sample,  $i$ .

### *Spawning fraction*

Ovaries of mature females were examined histologically. The ovaries were sectioned and stained with haematoxylin and eosin then examined to determine the presence/absence of post-ovulatory follicles (POFs). POFs were aged according to the criteria developed by Hunter and Goldberg (1980) and Hunter and Macewicz (1985). The spawning fraction of each sample was estimated as the mean proportion of females with hydrated oocytes plus day-0 POFs ( $d0$ ) (assumed to be 0-23 hrs old), day-1 POFs ( $d1$ ) (assumed to be 24-48 hrs old) and day-2 POFs ( $d2$ ) (assumed to be 48+ hrs old). The mean spawning fraction of the population was then calculated from the average of sample means weighted by proportional sample size from:

$$S = \left[ \overline{S}_i * \frac{n_i}{N} \right]$$

where  $n_i$  is the number of fish in each sample,  $N$  is the total number of fish collected in all samples and  $\overline{S}_i$  is the mean spawning fraction of each sample calculated from equation:

$$\overline{S}_i = \frac{[(d0 + d1 + d2POFs)/3]}{n_i}$$

where  $d0$ ,  $d1$  and  $d2$  POFs are the number of mature females with POFs in each sample and  $n_i$  is the total number of females within a sample. Note:  $d0$  includes hydrated oocytes.

### *Batch fecundity*

Batch fecundity was estimated from ovaries containing hydrated oocytes using the methods of Hunter et al. (1985). Both ovaries were weighed and the number of hydrated

oocytes in three ovarian sub-sections were counted and weighed. The total batch fecundity for each female was calculated by multiplying the mean number of oocytes per gram of ovary segment by the total weight of the ovaries. The relationship between female weight (ovaries removed) and batch fecundity was determined by linear regression analysis and used to estimate the batch fecundity of mature females in all samples.

#### *Confidence intervals*

Confidence intervals were obtained from a two stage bootstrap process with 100,000 iterations (Efron and Tibshirani 1993). The two stage bootstrap method allowed the incorporation of the covariance of adult parameters within individual samples, improving their precision (Ward et al. 2011b).

### **4.3 Results**

#### 4.3.1 Daily Egg Production Method

##### *Sampling area*

The total area sampled during the DEPM surveys varied from 46,525 to 120,468 km<sup>2</sup> between 1998 and 2009. In 2011, the total survey area was 114,745 km<sup>2</sup>. Plankton samples were collected at 341 stations on 34 transects.

##### *Egg distribution and abundance*

The distribution and abundance of sardine eggs has varied considerably among years. Important spawning areas include the eastern Great Australian Bight, between Coffin Bay and Ceduna, southern Spencer Gulf and the western end of Investigator Strait (Figures 4-3, 4-4). Mass mortality events in 1995 and 1998 had substantial effects on both the abundance of eggs and their spatial distribution (Ward et al. 2001a). The densities of sardine eggs in southern Spencer Gulf during 2011 were relatively low compared to previous years.

##### *Spawning area*

Estimates of spawning area varied among years and reflected both the size of the sampling area and the status of the spawning biomass. The spawning area declined substantially following the two mass mortality events in 1995/96 and 1998/99, from 68,260 km<sup>2</sup> in 1995 to 17,990 km<sup>2</sup> in 1996 and from 31,510 km<sup>2</sup> in 1998 to 14,876 km<sup>2</sup> in 1999 (Figure 4-5). The spawning area increased between 2001 and 2007 from 34,182 km<sup>2</sup> to 49,628 km<sup>2</sup>. The overall mean spawning area during this period was 40,298 km<sup>2</sup>

(SD = 6,606). In 2011, the spawning area was 42,075 km<sup>2</sup>. It is important to note that the survey design also varied between years, e.g. four transects were excluded in the central Great Australian Bight in 2004 due to bad weather.

#### *Egg production $P_0$*

Figure 4-5 shows estimates of egg production and their associated 95% CI between 1995 and 2011. Following the second mass mortality event, egg production declined from 111.78 eggs.day<sup>-1</sup>.m<sup>-2</sup> in 1998 to 38.12 eggs.day<sup>-1</sup>.m<sup>-2</sup> in 1999. In 2001 and 2002, estimates of egg production were 74.53 and 59.75 eggs.day<sup>-1</sup>.m<sup>-2</sup>, respectively. These estimates increased further to 66.31 and 120.91 eggs.day<sup>-1</sup>.m<sup>-2</sup> in 2003 and 2004, respectively. In 2005, egg production rates were calculated to be 55.66 eggs.day<sup>-1</sup>.m<sup>-2</sup>. Egg production then rose again to reach 102.98 eggs.day<sup>-1</sup>.m<sup>-2</sup> in 2007 and then declined in 2009 to 63 eggs.day<sup>-1</sup>.m<sup>-2</sup>. In 2011, the estimated egg production was 44.0 eggs.day<sup>-1</sup>.m<sup>-2</sup>.

#### *Sex ratio ( $R$ )*

Estimates of sex ratio from fishery-independent samples ranged between 0.44 in 2003 and 0.64 in 2006 (Figure 4-5). The small variability in estimates of  $R$  reflects the stability of this parameter between years and the relatively low potential to influence estimate of spawning biomass.

#### *Mean female weight ( $W$ )*

Estimates of  $W$  ranged from 46.3 g in 1998 to 78.7 g in 2004 (Figure 4-5). Fishery independent samples were mainly taken from shelf waters and were typically larger than commercial catch samples that were mainly taken from Spencer Gulf.

#### *Batch fecundity ( $F$ )*

Batch fecundity was not estimated in 1995 and 1996 because no hydrated females were caught in adult sampling. Between 1997 and 2009, estimates of  $F$  from fishery-independent samples ranged between 13,965 and 22,304 hydrated oocytes (Figure 4-5). Variations in  $F$  between years mainly reflected variations in gonad-free female weight (and  $W$ ).

#### *Spawning fraction ( $S$ )*

Estimates of  $S$  varied between 0.044 in 2011 and 0.18 in both 1999 and 2001 (Figure 4-5). Estimates of  $S$  for 2003 and 2004 were based on less than 500 females.

### *Spawning biomass*

The spawning biomass estimate for 1995 of 165,000 t may be negatively biased, as the survey did not coincide with the peak spawning season or sample the entire spawning area, and also because estimates of adult reproductive parameters were obtained from other studies. Hence, there is no reliable estimate of the spawning biomass prior to the first mass mortality event in late 1995. The estimate of spawning biomass declined to 37,000 t in 1996 following this mass mortality event but increased to 59,000 t in 1997 (Figure 4-6; Ward et al. 2001a).

Estimates of spawning biomass since 1998 (Figure 4-6) have been recalculated after a review by Ward et al. (2011b). Spawning biomass estimates increased from 22,906 t (95% CI = 13,518-41,151) in 1999 to 180,787 t (103,493-337,037) in 2002. Estimates of spawning biomass remained between 152,000 and 170,000 t for 2003 to 2005, and then rose to 202,624 t (119,317-363,363) in 2006 and 263,049 t (161,321-435,205) in 2007. The estimate of spawning biomass in 2009 was 171,532 t (122,100-242,479). The estimate of spawning biomass in 2011 was 193,201 t (104,151-381,961). Coefficients of variation (CV) from spawning biomass estimates were high between 0.35 in 2009 and 2011 and 0.63 in 2002.

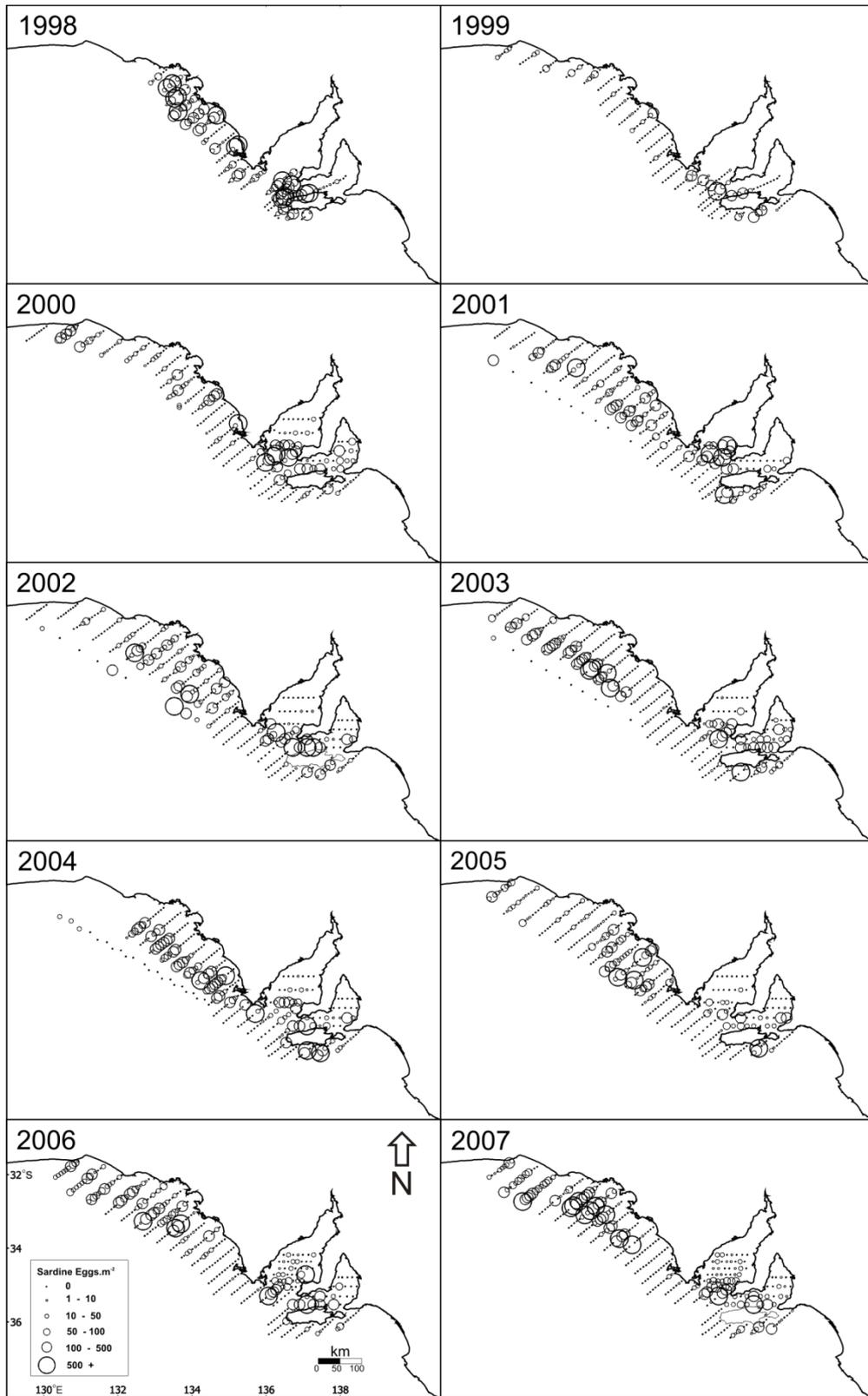


Figure 4-3. Distribution and abundance of eggs collected during surveys between 1998 and 2007.

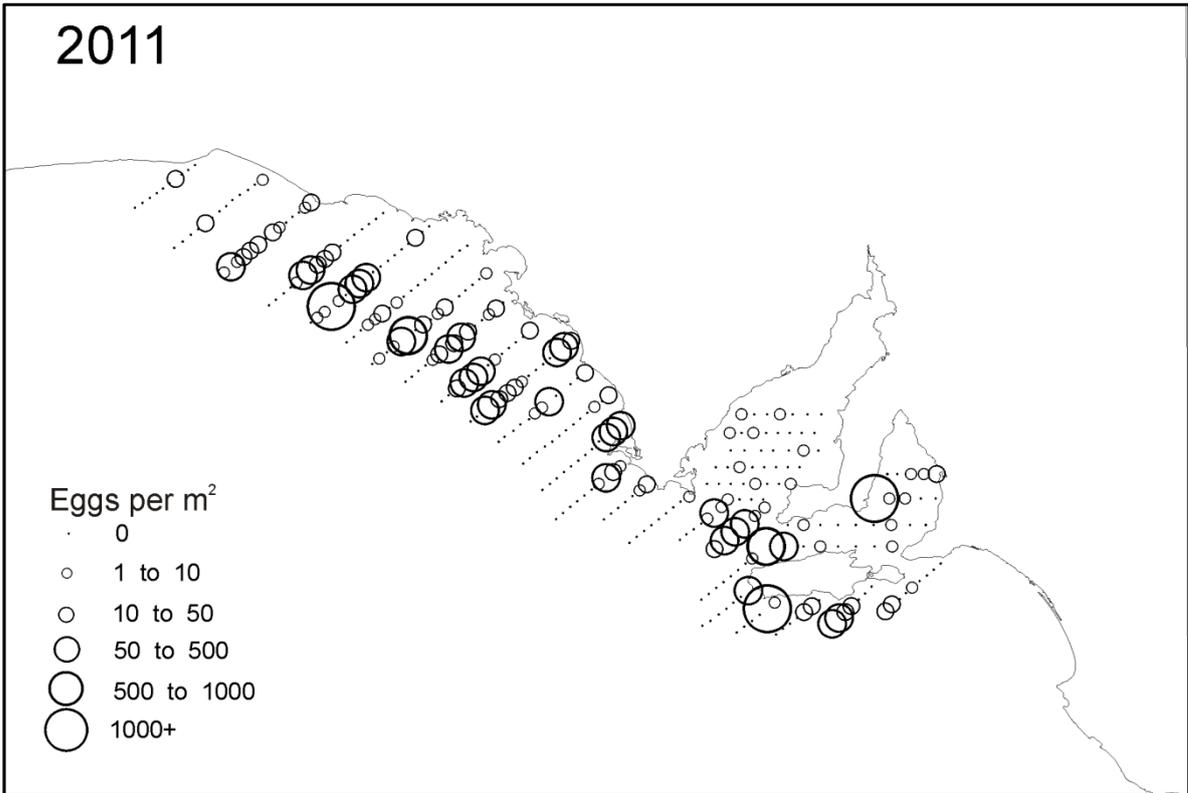
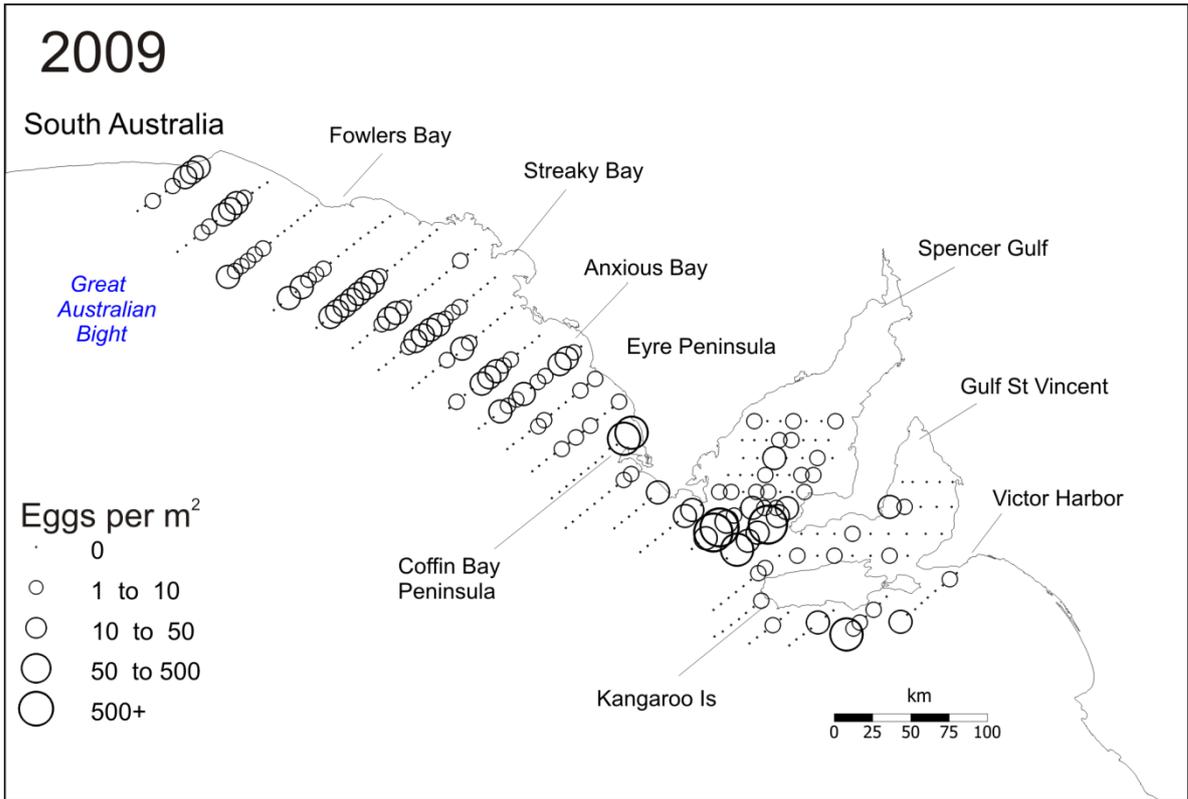


Figure 4-4. Distribution and abundance of eggs collected in 2009 and 2011.

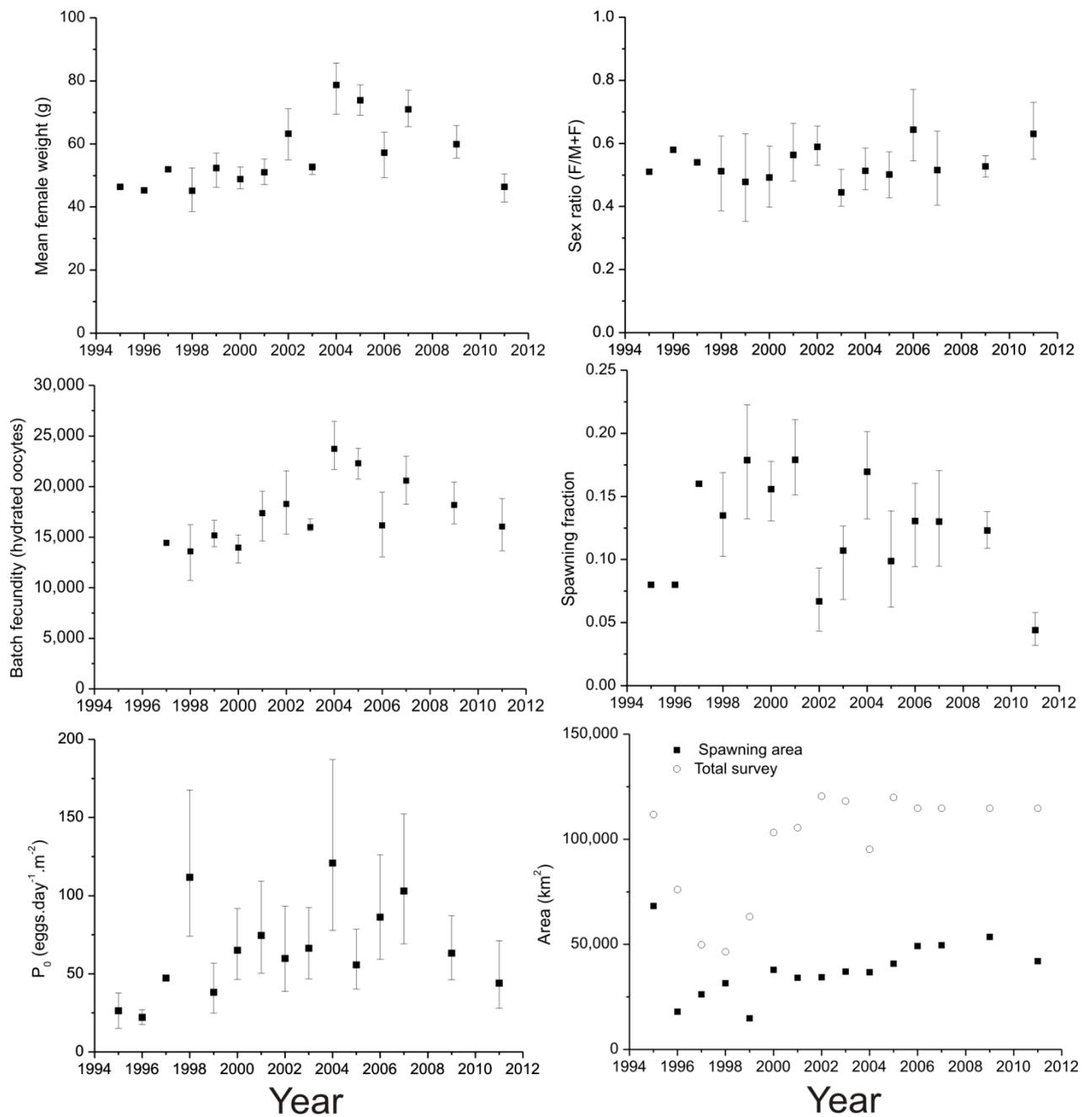


Figure 4-5. Spawning biomass parameters ( $\pm 95\%$  CI) used from 1995-2011. Values from 1998-2009 are from Ward et al. (2011b), earlier values from Ward et al. (2001c).

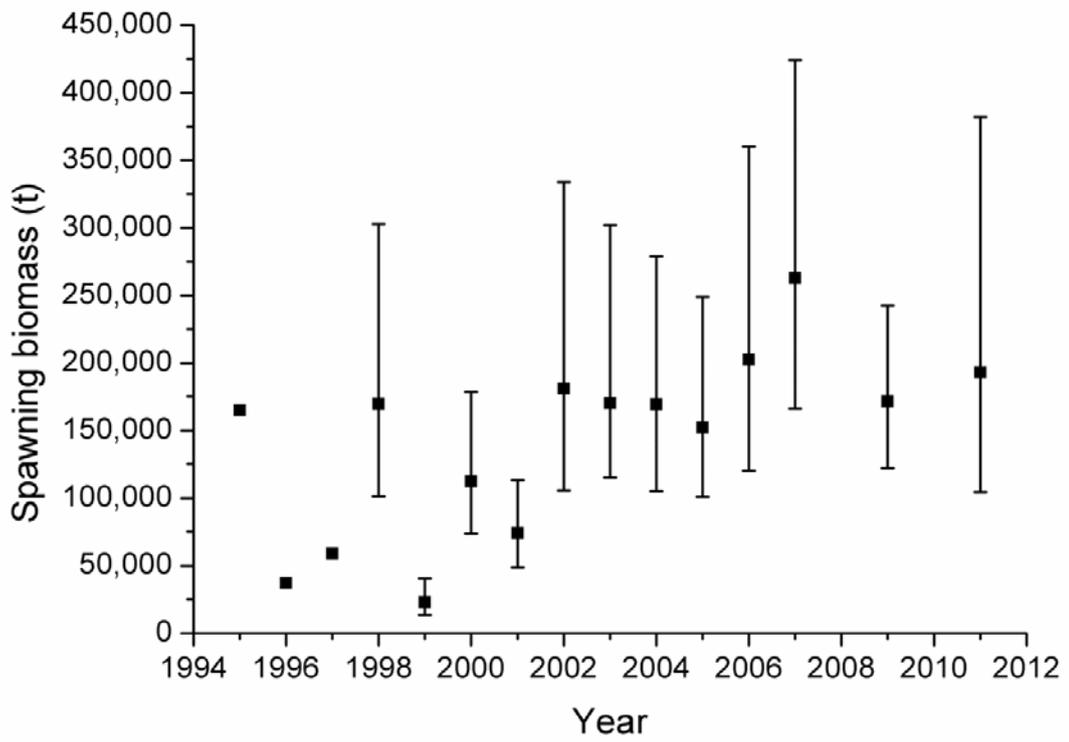


Figure 4-6. Daily Egg Production Method (DEPM) estimates of spawning biomass ( $\pm 95\%$  CI) in South Australian waters from 1995 - 2011.

#### 4.4 Discussion

Two of the findings for 2011 differ from those obtained in previous years. Firstly, egg densities in Spencer Gulf recorded during 2011 were unusually low. This finding, combined with the reduced proportion of large adult fish in catches from southern Spencer Gulf in recent years (Chapter 2), suggests that fishing pressure may have reduced adult abundance in this area. However, the low spawning frequencies recorded in 2011, combined with the reduced sardine growth rates (Chapter 3), suggest that environmental factors may also have played a role in these variations.

The DEPM has been integral to the rapid and sustainable development of the SASF. The main weakness of the method is that estimates of spawning biomass obtained using the DEPM are relatively imprecise (Cochrane 1999). This has mainly been attributed to patchiness in egg distribution and abundance (McGarvey and Kinloch 2001; Gaughan et al. 2004; Stratoudakis et al. 2006), and the high variance levels associated with estimates of mean daily egg production. Hence, variances for estimates of spawning biomass are large, e.g. CVs are commonly >35% of the mean (Fletcher et al. 1996; Ward et al. 2001a; Gaughan et al. 2004; Stratoudakis et al. 2006).

The confidence intervals around the estimates of spawning biomass are also wide (coefficients of variation of 0.35-0.63). To address this uncertainty a conservative approach (i.e. the log linear-model) has been adopted to estimating mean daily egg production. Ward et al. (2011b) demonstrated that the log-linear model provides more conservative estimates of egg production and spawning biomass than other methods (exponential model, various GLMs). Importantly, the log-linear model does not provide unrealistically high estimates of egg production in years when a few samples contain very large numbers of young eggs (Ward et al. 2011b). A research proposal to develop improved methods that will provide more precise estimates of spawning biomass is currently being developed for submission to FRDC in late 2012.

In 2007, the management framework for the SASF was amended to address the imprecision of estimates of spawning biomass and provide increased inter-annual stability in catch levels. Rather than setting the TACC as a percentage of the estimate of spawning biomass, as was done previously (Shanks 2005), an indicative TACC of 30,000 t was established for the fishery (PIRSA 2007). This indicative TACC is established as the effective TACC when the latest estimate of spawning biomass remains above 150,000, which corresponds to an exploitation rate (catch/spawning biomass) of 20%. Since 2010, an additional 4,000 t has been taken outside the traditional fishing grounds – this was a

first step towards establishing the potential for a spatial management framework for the SASF. As discussed in previous chapters, the recent reductions in the age/size of sardine taken from southern Spencer Gulf, combined with the increasing proportion of immature fish taken from this region, suggests that additional spatial management may be required in the fishery.

## 5.0 STOCK ASSESSMENT MODELLING

### 5.1 Introduction

This chapter integrates the time series of spawning biomass estimates for the SASF with other information for the fishery (especially age structure and growth data) using the population model developed in Ward et al. (2005a). The model is used to estimate mature biomass and project forwards two years into the future under three hypothetical TACCs (i.e. 30,000 t, 37,500 t and 45,000 t).

### 5.2 Methods

#### 5.2.1 Stock assessment model

The stock assessment model used in this study was originally written by Dr Peter Stephenson as part of an FRDC funded project to implement an age-structured stock assessment model for the SASF (Ward et al. 2005a). The model is both age and sex dependent with different von Bertalanffy growth parameters for males and females. Fish recruit to the model at age 2 and suffer monthly depletion from natural and fishing mortality. Each year they progress through to the next age class and until they reach 8 years of age where they remain until they are fished or die from natural mortality.

Before 1992 the sardine population was assumed to be in a state of unfished equilibrium. Initial recruitment was determined from the Beverton and Holt stock-recruitment relationship (Hilborn et al. 1994) with the parameters determined from the biomass in the initial state using a steepness parameter fixed at  $h = 0.7$ . Annual recruitment variation for years where age data were available, were multiplied by a log-normally distributed recruitment deviation factor, which was estimated in the model. Where no age data were available, annual recruitment deviations were randomly sampled from a log-normal distribution with a mean of zero and variance equal to the historical annual recruitment deviations.

The uncertainties around estimates of mature biomass were determined using Bayesian methods. A joint posterior probability distribution was generated using the Markov Chain Monte Carlo (MCMC) procedure. A subset of 1300 values from the joint posterior distribution was obtained by saving every 1000<sup>th</sup> from 1,300,000 cycles of the MCMC. The first 300 estimates generated during the “burn in” period were rejected and the mature biomass determined using the remaining 1000 values from the joint posterior distribution. The model was implemented using the software package, AD Model Builder (ADMB Project 2009). Full details of the model are given in Ward et al. (2005a).

The 2010 assessment identified natural mortality and the steepness of the stock recruitment relationship were poorly estimated by the model. Preliminary model runs of this (2012) assessment also found natural mortality and steepness to be unreliable, tending towards the upper and lower bounds set in the model. As with the 2010 analysis these parameters were fixed and a number of sensitivities undertaken to determine their effect on estimates of mature biomass.

### 5.2.2 Input data and sensitivity analysis

The following section describes the data used in the current and updated implementation of the age-structured stock assessment model. Sensitivity analyses undertaken on parameters and data used are also discussed.

#### *Commercial catch*

Monthly commercial catch data from the SASF were available from January 1992 to December 2011 (Chapter 2).

#### *Fishery-independent spawning biomass estimation*

Spawning biomass obtained from annual DEPM surveys between 1995 and 2007 in 2009 and 2011 were used as an input to the model (Chapter 4).

#### *Vulnerable biomass*

The original implementation of the model estimated the vulnerability to capture of fish aged 2+, 3+, and 4+ years, with older fish assumed to be fully vulnerable. As sardine aged 2+ are likely fully vulnerable to fishing gear the vulnerability of all age classes was fixed in the model. Sensitivity analyses were undertaken with vulnerability rates of between  $V = 1$  (fully vulnerable),  $V = 0.5$  and  $V = 0.3$  for all fish to determine the effects of this parameter on estimates of spawning biomass.

#### *Age data*

Age-composition data from commercial catches were available from 1995 to 2011 and in most years from fishery-independent research surveys between 1998 and 2006 (Chapter 3). Fishery-independent samples of age structure have not been available in recent years. The three methods used to estimate age are described in Chapter 3. Age was estimated from (1) annuli counts from otoliths with readabilities 1 and 2 alone, (2) annuli counts from otoliths with readabilities 1 and 2 and from the otolith weight relationship for remaining fish with an otolith weight, and (3) the age-length key (Table 3-3).

### *Mass mortality events*

The original implementation of the model assumed mass mortality rates ( $K$ ) = 0.7 for all fish age  $\geq 3$  after the first mass mortality event in 1995 and  $K = 0.7$  of all fish aged  $\geq 1$  after the second mass mortality event in 1998 (Ward et al. 2001a). To determine the influence of  $K$ , sensitivity runs of  $K = 0.3$  and  $K = 0.5$  were undertaken.

### *Natural mortality*

Natural mortality was fixed at  $M = 0.6$  in the model. Sensitivity analyses were undertaken with  $M = 0.4$  and  $M = 0.8$  to investigate how this parameter affected biomass estimates.

### *Recruitment steepness*

The steepness parameter,  $h$ , is used to determine the relationship between stock size and recruitment in the model. This parameter was fixed at  $h = 0.7$  in both the original model and the 2010 assessment. To investigate how  $h$  affects estimates of mature biomass sensitivity analyses with  $h = 0.6$  and  $h = 0.8$  were compared to a baseline of  $h = 0.7$  in this assessment.

### *Objective function weightings*

The three data sources fitted in the model must be assigned weightings to be used in the fitting process. In the original implementation of the model these weightings were 0.01, 1 and 1 for the age data, DEPM and recruitment deviations, respectively. The weightings on each of these data sources were varied separately to determine their influence on the model.

### *Base-case model for sensitivity analysis*

Each sensitivity analysis varied one parameter or data input to determine how biomass estimates varied with changing model inputs compared with the base-case model. The base-case model used age data from both sectors obtained from otolith weight relationships, a natural mortality of  $M = 0.6$  and a mass mortality rate of 0.7 applied to 3+ fish in 1995 and 1+ fish in 1998. All fish were assumed fully vulnerable to fishing and the objective function weightings of the age data, DEPM estimates and recruitment deviations were 0.01, 1 and 1, respectively, corresponding to those used in the original implementation of the model (Ward et al. 2005a) and the 2010 assessment (Ward et al. 2010).

### 5.2.3 Final stock assessment model outputs

The sensitivity analyses were used to inform the data sources and values of the fixed parameters in the final stock assessment model runs. Sensitivity analyses determined the weightings on the objective function should remain unchanged from the 2010 assessment and the original implementation of the model of 0.01, 1 and 1 for the age data, DEPM and recruitment deviations, respectively.

#### *Age data*

The age data were found to strongly influence estimates of mature biomass from the stock assessment model. Recognising this uncertainty, the final implementation of the model used age data from method 2 (otolith weight relationship). Two scenarios were considered, all age data (both commercial and fishery-independent) and age data from commercial samples alone.

#### *Mass mortality rate*

The model performed poorly with respect to the mass mortality events. A mass mortality rate of  $K = 0.7$  was used for both mass mortality events.

#### *Vulnerability*

The fishery currently operates primarily in Spencer Gulf while both the stock assessment model and the DEPM are applied over a broader spatial region. To account for the likelihood that a component of the population is not available to fishing, vulnerabilities of  $V = 1$  and  $V = 0.5$  were considered.

#### *Recruitment steepness*

The steepness parameter  $h$  is used to determine the relationship between recruitment and stock size. Values of  $h = 0.7$  and  $h = 0.8$  were assigned based on the assumption that recruitment is not highly reliant on stock size.

#### *Projected biomass*

In the original model (Ward et al. 2005a) catch in the projected years was determined by the level of the vulnerable biomass and the harvest decision rules. For the 2010 assessment, the harvest decision rules were replaced with an explicit TACC that was harvested if there was sufficient vulnerable biomass. The current implementation of the model sets explicit TACCs of 30,000, 37,500 and 45,000 t for the projected years 2012 and 2013.

## 5.3 Results

### 5.3.1 Model sensitivity analysis

#### *Base-case model*

Under the base-case model unfished biomass in 1992 was about 282,000 t, falling to 234,000 t in 1995 after the first mass mortality event (Figure 5-1a-i). Biomass then recovered to reach a peak of 410,000 t in 1998 before declining to 132,000 t in 1999, after the second mass mortality. Biomass declined slightly in 2000 and 2001, then increased rapidly to 203,000 t in 2004. Between 2005 and 2011, biomass was estimated to be between 186,000 and 206,000 t. Estimates of biomass from the base-case model were higher than the DEPM estimates in most years; except in 2002, 2006 and 2007.

#### *Ageing methods*

Biomass estimates from the age-length key were the highest in all years with a maximum biomass ~635,000 t in 1998 immediately before the second mass mortality event (Figure 5-1a). The unfished biomass from this method was also highest at ~408,000 t. The other two methods for estimating age (annuli data alone and otolith weight) produced similar estimates of biomass in most years. The exception was the period between the two mass mortality events (1995 – 1999) where biomass from the annuli method was ~50% lower than that estimated from the otolith weight method (Figure 5-1a).

#### *Age input data*

Varying the data sources, using commercial age only or fishery-independent age produced few differences in estimated biomass, except for 1998, prior to the second mass mortality event where biomass when all fish were included was ~410,000 t, fishery-independent samples only ~325,000 t and commercial samples only ~220,000 t (Figure 5-1b).

#### *Mass mortality rate*

Varying the mortality rate from the two mass mortality events resulted in differences in estimated biomass in most years, not just those around the years of the mortality events (Figure 5-1c). The unfished biomass was highest under the  $K = 0.7$  scenario at ~280,000 t, falling to 230,000 t after the first mortality event then rising to ~410,000 t in 1998. Under the  $K = 0.5$  and  $K = 0.3$  scenarios, unfished biomass was ~247,000 t and 231,000 t, respectively, rising to 330,000 t and 295,000 t in 1998. In 1999, estimates of biomass were ~131,000 t, ~175,000 t and ~213,000 t under the  $K = 0.7$ , 0.5 and 0.3 scenarios respectively. All three methods predicted similar biomass between 2004 and 2009.

Estimated biomass in 2011 was ~190,000 t under the  $K = 0.7$  scenario, ~172,000 t under the  $K = 0.5$  scenario and ~164,000 t under the  $K = 0.3$  scenario.

#### *Natural mortality rates*

Varying the natural mortality rate produced small changes in biomass estimates. Unfished biomass was predicted to be highest when  $M = 0.4$  at ~320,000 t, compared to ~280,000 t when natural mortality was 0.6 and 270,000 t where  $M = 0.6$  (Figure 5-1d). Between the two mass mortality events (1995 – 1999), biomass was 10 – 20% lower than the baseline under the  $M = 0.4$  scenario and 6 – 15% higher than the baseline under the  $M = 0.8$  scenario. In recent years (2005 onwards), biomass was similar among all three scenarios being 2 – 5% lower than the base-case when  $M = 0.4$  and 2 – 5% higher than the base-case when  $M = 0.8$ .

#### *Vulnerability*

Varying the vulnerability of fish in the model resulted in small changes to the biomass series (Figure 5-1e). Under the  $V = 1$  scenario, biomass was highest in all years except 2005 and 2006, when the  $V = 0.5$  scenario was highest. The greatest differences of biomass estimates among scenarios was between 1995 and 1998 when the  $V = 1$  scenario was 12 – 18% higher than the  $V = 0.5$  scenario and 22 – 30% higher than the  $V = 0.3$  scenario. Trends in biomass were identical under all scenarios.

#### *Recruitment steepness*

The recruitment steepness parameter was varied above ( $h = 0.8$ ) and below ( $h = 0.6$ ) the level of  $h = 0.7$  used in the base-case model. Trends in biomass were similar under all scenarios. The greatest differences among methods occurred between 1992 and 1995 when biomass was ~5% higher than the base-case when  $h = 0.6$  and ~4% lower than the baseline when  $h = 0.8$  (Figure 5-1f). This parameter had minimal impact on estimates of mature biomass.

#### *Objective function weightings*

##### *Age weighting*

The model was sensitive to changes in the weighting on the age data, with weightings outside of those tested failing to converge to a solution. The weightings that are presented (0.0125, 0.01 and 0.005) resulted in differences in biomass compared to the base-case of up to 26% between 1995 and 2000 (Figure 5-1g). In other years, the greatest difference from the base-case was ~5%.

### *DEPM weighting*

The model was very sensitive to changes in the weighting of the DEPM estimates. Decreasing the DEPM weighting to 0.25 resulted in biomass estimates between 35% and 46% higher than the base-case (Figure 5-1h). When the weighting on the DEPM was increased to 10, biomass estimates were lower than the base-case in all years, except 2006 and 2007.

### *Recruitment deviations weighting*

Increasing the weighting on the recruitment deviations up to 2 resulted in slightly higher biomass estimates compared with the base-case model in most years (Figure 5-1i). Increasing the weighting on the recruitment deviations above 2 or below 0.75 resulted in the failure of the model to converge.

## 5.3.2 Final stock assessment model outputs

### *Biomass estimates to 2011*

Estimates of mature biomass from the base-case model differed from those obtained using the DEPM prior to 2001. In 1995 and 1999, immediately after the mass mortality events, biomass estimates from the base-case model were much higher than those from the DEPM (Figure 5-2a-h). However, model estimates lay within the DEPM confidence intervals from 2002 onwards. Vulnerability and recruitment steepness had only small effects on biomass estimates. While the biomass estimates prior to 2000 differed between the scenarios that used both commercial and fishery-independent age data and those using commercial age data alone, trends after 2002 were similar.

In recent years (since 2002) the scenarios incorporating both commercial and fishery-independent age data estimated biomass to peak at ~205,000 t in 2007 or 2008. Biomass then declined slightly to between 183,000 and 190,000 t in 2011 (Figure 5-2a-d). Under the scenarios using the commercial age data alone the biomass in recent years peaked in 2007 or 2008 at between 202,000 and 205,000 t. Biomass in 2011 under these scenarios ranged between 167,000 and 171,000 t (Figure 5-2e-h).

The levels of uncertainty around model estimates of biomass were high, with 95% confidence intervals on 2011 estimates of biomass ranging from 112,000 to 325,000 t. The confidence intervals from the DEPM were between 104,000 and 382,000 t (Figure 5-2a-h).

### *Projected biomass*

The differences in projected biomass estimates for 2013 under the 30,000 t TACC scenarios were predominately due to the use of different age data, with vulnerability and recruitment steepness having little influence on biomass estimates (Figure 5-2a-h). Estimates of biomass in 2013 under a TACC of 30,000 t were between 188,000 and 199,000 t when all the age data were used and between 165,000 and 172,000 t when the age data from commercial catch alone were used. In scenarios where the TACC was increased to 37,500 t, estimates of biomass in 2013 were between 4% and 6% less than those where the TACC was 30,000 t. Scenarios where the TACC was increased to 45,000 t had biomass estimates 8–12% lower in 2013 than those with a TACC of 30,000 t.

Under a 30,000 t TACC, biomass in 2013 was predicted to remain similar to the 2011 estimate under all scenarios. Increasing the TACC to 37,500 t or 45,000 t resulted in biomass estimates declining slightly from those in 2011. Importantly the projected biomass was above the 150,000 t limit reference point (PIRSA 2007) in all scenarios, except where the TACC was 45,000 t and  $V = 0.5$ ,  $H = 0.8$  and commercial age data alone were used (148,624 t). The precision of these estimates was poor, with 95% confidence intervals for spawning biomass between 100,000 t and 325,000 t under the majority of scenarios.

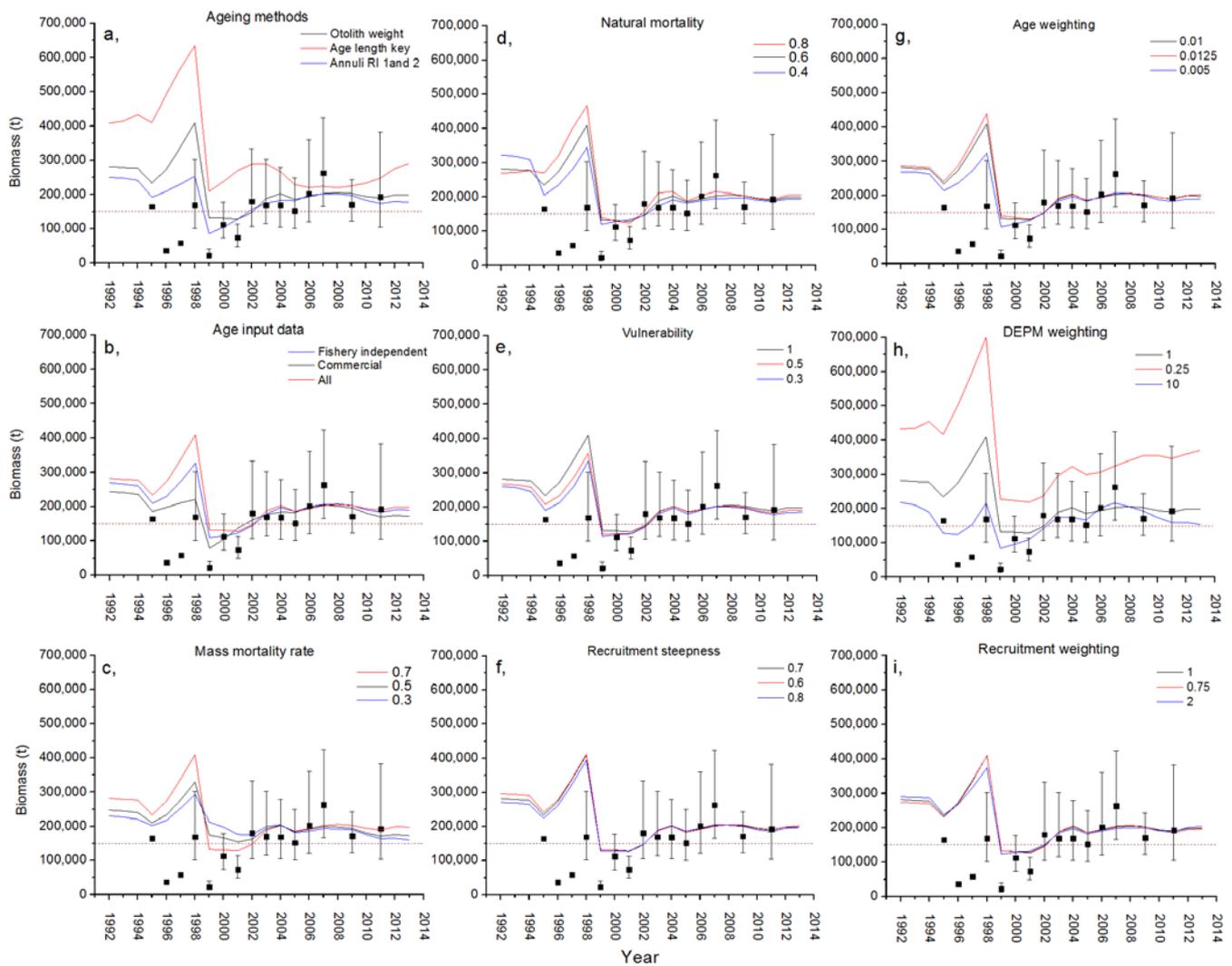


Figure 5-1. Estimates of model derived biomass between 1992 and 2011 from sensitivity analyses. The base-case model was implemented using all age data obtained from an otolith weight relationship,  $M = 0.6$ , a mass mortality rate of 0.7, a vulnerability of 1 and objective function weightings of 0.01, 1 and 1 for the age data, DEPM estimates and recruitment deviations, respectively. Three sources of age data were tested, annuli data with readabilities of 1 and 2 alone, age from an otolith weight relationship and age from an age-length key. (a) Commercial and fishery dependent age data were used separately. (b) Mass mortality rates of 0.7, 0.5 and 0.3 were investigated. (c) Natural mortality rates of  $M = 0.8, 0.6$  and  $0.4$  were investigated (d). Vulnerabilities of the stock of 1, 0.5, 0.3 were tested. (e) The relationship between stock size and recruitment was varied. (f) The weightings of the age data. (g) DEPM (h) and recruitment deviations (i) in the objective function were varied. DEPM estimates (black squares and lines) are shown for reference.

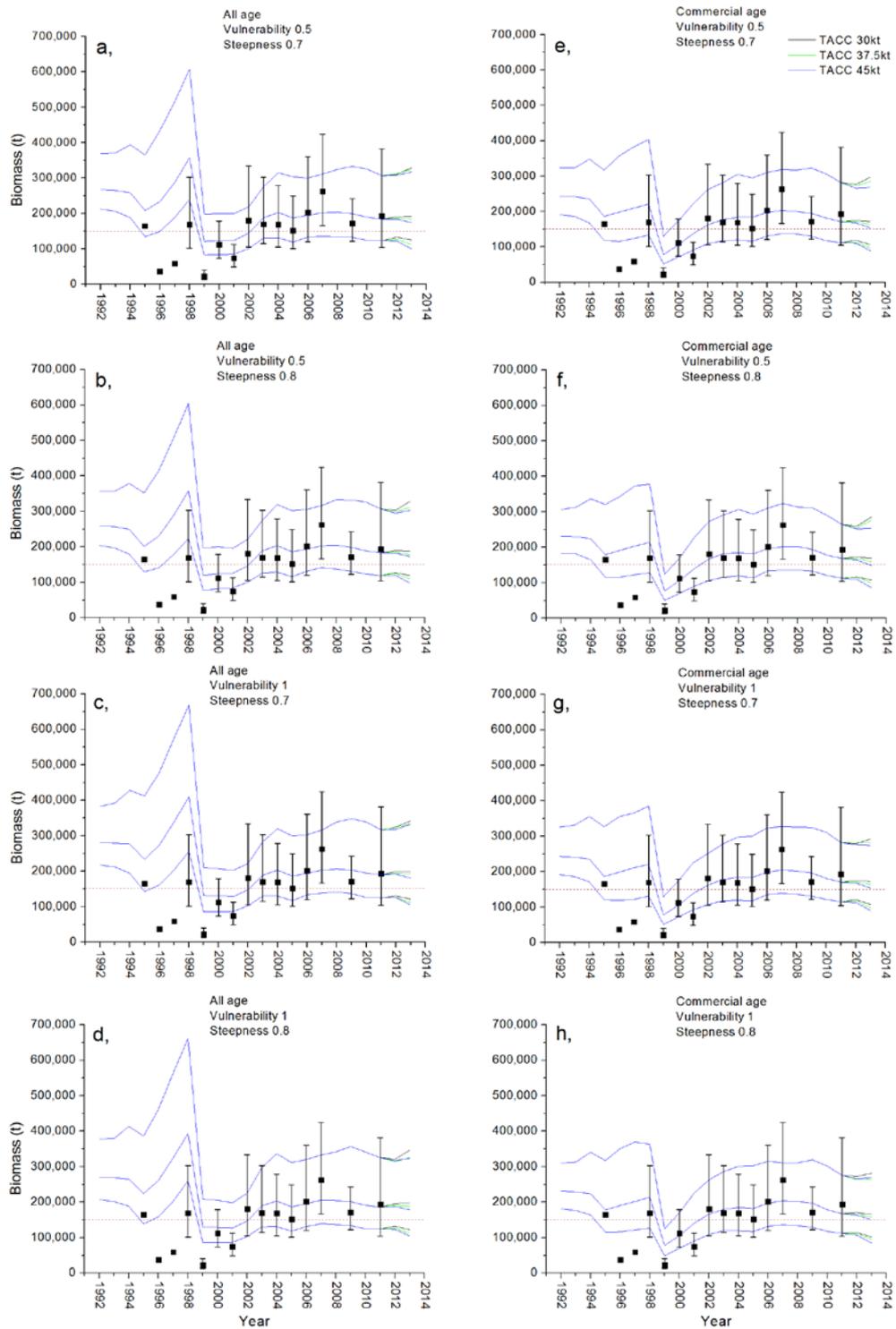


Figure 5-2. Model derived estimates of mature sardine biomass between 1992 and 2011 with 95% confidence intervals. Estimates were obtained using age structures derived from an otolith weight relationship, a mass mortality rate of 0.7 and a natural mortality rate of  $M = 0.6$ . Results from models using both commercial and fishery-independent age data (a-d) and those using commercial age structure alone (e-i) are presented under the assumption of vulnerabilities 1 and 0.5 and recruitment steepness 0.7 and 0.8. DEPM estimates (black squares and lines) are shown for reference. The model projects forwards (2012 and 2013) to estimate mature biomass under TACCs of 30,000, 37,500 and 45,000 t.

## 5.4 Discussion

One of the main advantages of using a population model to estimate spawning biomass is that this approach has the potential to reduce year to year fluctuations that are caused by the low precision of the DEPM rather than actual changes in abundance. Conversely, there is potential for this approach to mask real fluctuations in abundance by fitting to the available data. The model outputs suggest that spawning biomass has been relatively stable at around 170,000 to 200,000 t over the last decade, whereas annual estimates of spawning biomass obtained using the DEPM since 2002 have ranged between approximately 152,000 and 263,000 t (Chapter 4).

The assessment conducted by Ward et al. (2010) identified several sources of uncertainty that affect model outputs. For example, the method used to estimate age had a considerable impact on model outputs. Spawning biomass estimates obtained using age estimates from the age-length key were higher and more variable than those estimated from annuli or otolith weight data. These differences were greatest prior to and immediately after the second mass mortality event. It has previously been shown that age-length keys produce imprecise and biased estimates of age, especially for species with highly variable growth rates (Westrheim and Ricker 1978; Fletcher 1994).

The source of age data used (commercial fishing or fishery-independent samples) also had significant effects on the estimates of spawning biomass, especially in the period following the first mass mortality event in 1995. This is because fishery-independent samples, especially those collected after 1998, are comprised of older fish than the commercial catch samples. Neither the commercial or fishery-independent samples were collected randomly: the fishery independent samples were obtained for use in the DEPM and targeted mature fish and the commercial catch was taken from an area where relatively young fish usually predominate. These differences in age structure explain the discrepancy in the model estimates of spawning biomass between the 1995 and 1998 mass mortality events that are obtained using these two data sources.

The model fitted poorly to the fluctuations in biomass associated with the mass mortality events, particularly the second event, under all scenarios tested. This is because the stock recovered very rapidly from these events, perhaps due to density-dependent effects (i.e. higher rates of growth, survival and reproduction at reduced abundances) that are not accounted for in the model (Ward et al. 2001a). To allow for this rapid recovery, model-generated estimates of the 1998 spawning biomass were between 300,000 and 450,000 t, allowing for spawning biomass estimates of between ~130,000 and ~160,000 t in 1999,

which were both much higher than the corresponding DEPM estimates. The differences between model-generated and DEPM estimates of spawning biomass were most pronounced before 2001 for all scenarios tested.

From 2003 onwards, trends in model-generated spawning biomass estimates were comparable to, although generally slightly higher than, the trends in the DEPM estimates under a wide range of scenarios. This finding provides confidence that model-generated outputs are robust. The spawning biomass was stable from 2003 to 2011 under all modelled scenarios, typically at levels of approximately 200,000 t. Under the majority of scenarios tested, spawning biomass is predicted to remain above the limit reference point of 150,000 t for TACCs below 45,000 t.

## **6.0 GENERAL DISCUSSION**

### **6.1 Stock status and uncertainty**

This assessment of the status of the South Australian sardine stock analyses a large quantity and wide range of high-quality fishery-dependent and fishery-independent data and synthesises these data using a dynamic age-structured population model. The implications of uncertainties in the data, assumptions and estimates are explicitly tested by conducting sensitivity analyses using the model. We consider that the information available and approach taken in this report provides a sound basis for assessing the status of the South Australian sardine stock. The key limitations of the assessment are associated with uncertainties in 1) estimates of spawning biomass resulting from both the inherent imprecision of the DEPM and from its biennial (c.f. annual) application in recent years and 2) estimates of fish age resulting from difficulties associated with reading annuli in sardine otoliths.

Data and analyses suggest that the spawning stock on which the SASF is based is in a relatively strong position. Most importantly, the estimate of spawning biomass that was obtained in the DEPM surveys undertaken in February-March 2011 (~193,000 t) is well above the limit reference point of 150,000 t for this key biological performance Indicator for the SASF that was specified in the harvest strategy (PIRSA 2007). The outputs from the age structured model developed for the fishery suggest that the spawning biomass has been stable at approximately 170,000 to 200,000 t over the last decade. Model outputs suggest that catch levels below 45,000 t are unlikely to cause the spawning biomass to decline below the limit reference point over the next two years.

Despite the strong position of the sardine spawning stock, there is some evidence to suggest that recent fishing patterns may need to be refined. Most importantly, in the last few years both the size and age of sardine taken in commercial catches from southern Spencer Gulf have declined. The modal size class for the fishery has typically been ~150 mm FL but declined to 140 and 130 mm FL in 2011 and 2012 (data for January-June only), respectively. This decline in fish size may be partially explained by the decline in growth rates that has occurred in recent years (Chapter 3). However, the reduced numbers of fish older than three years of age and the increased proportion of immature fish (>80%) in catches taken since 2010 suggests that the concentration of fishing effort in southern Spencer Gulf may also be a contributing factor. The reduction in the density of eggs in southern Spencer Gulf observed during the 2011 DEPM survey also suggests that current fishing pressure in this area may be too high.

## 6.2 Future management and research

A new management plan is currently being developed for the SASF which will include a revised harvest strategy that builds on the two previous harvest strategies (Shanks 2005, PIRSA 2007). In this section of the report we provide information to assist the development of the new harvest strategy, especially in relation to the identification of suitable biological performance indicators and limit/target reference points.

### *Spawning biomass as a key biological performance indicator*

It is widely acknowledged that estimates of spawning biomass obtained using the DEPM, which is the key biological performance indicator for the SASF (Shanks 2005, PIRSA 2007) are an accurate but relatively imprecise measure, with coefficients of variation commonly exceeding 35% (e.g. Stratoudakis 2006; Ward et al. 2011b). The confidence intervals around the estimates of spawning biomass are also wide (coefficients of variation of 35-63%).

The harvest strategy for the SASF addressed the uncertainty in the estimates of this biological performance indicator by establishing a conservative TACC (30,000 t) that was maintained while the estimate of spawning biomass remained above the lower limit [target] reference point (150,000 t). This approach was specifically designed to reduce inter-annual variability in TACCs associated with year to year fluctuations in estimates of spawning biomass caused by the low precision of the method rather than actual changes in adult abundance. When the previous harvest strategy was adopted, it was acknowledged that the benefit of inter-annual stability in catches was traded off against the cost of a reduction in potential yield. This was, in part, because the lower limit reference point for spawning biomass of 150,000 t for a 30,000 t TACC equates to an exploitation rate of 20%, which is a conservative level for productive species such as sardine (Patterson 1992). The conservative nature of the 20% as a limit reference point for exploitation level for the South Australia sardine stock is evidenced by the rapid recovery of the spawning biomass off South Australia following the mass mortality events in 1995 and 1998, which each killed over 70% of the adult biomass (Ward et al. 2001c, 2011b).

Maximum sustainable yields for low trophic level (forage) species, such as sardine, are typically achieved at depletion levels of approximately 60 per cent, equivalent to 40 per cent of unfished biomass (Smith et al. 2011). However, harvest strategies for fisheries for these species also need to consider potential impacts on biodiversity and ecosystem health (Goldsworthy et al. 2011). Biomass levels above 75% of the unfished level have been identified as a global average for achieving a balance between protecting ecosystem

function and biodiversity and providing for food production and economic development of low trophic level species (Smith et al. 2011). However, Australian ecosystems have been shown to be considerably less sensitive to harvesting of low trophic level species than other systems worldwide (e.g. Smith et al. 2011). Information presented in this report indicating that (after a decade of catches above 25,000 t per annum) the current spawning biomass is approximately 170,000-190,000 t suggest that the lower reference point for spawning biomass of 150,000 t is likely to be sufficiently conservative to take into account the importance of sardine in the ecosystem. Intensive ecosystem monitoring and modelling by Goldsworthy et al. (2011) also provide empirical evidence that levels of fishing effort in the SASF that have maintained the spawning biomass above that level have not impacted negatively on ecosystem function.

The decreased risk to the stock provided by the previous harvest strategy also provided an opportunity to reduce the level (cost) of stock assessment. A critical element of the previous harvest strategy was to move from an annual to biennial application of DEPM and the production of a stock assessment report every second year (alternative year to DEPM reports) instead of annually. As noted above, the previous harvest strategy acknowledged that the benefits of stable TACCs and reduced assessment levels were explicitly traded off against a reduction in yield from the fishery.

The establishment of an additional TACC of 4,000 t outside the traditional fishing grounds in 2010-12, combined with the industry request to take 2,000 t of the 2013 TACC during 2012, suggests that the next harvest strategy for the SASF may need to include options for increasing yield (within ecological sustainable limits) beyond the conservative level identified in the previous harvest strategy. However, the next harvest strategy should also include options for maintaining the stable and conservative TACCs provided by the previous approach. Information presented in the current report suggests that the next harvest strategy for the SASF should maintain 150,000 t as the lower target reference point for spawning biomass because this level takes into account the ecological importance of sardine (Goldsworthy et al. 2011; Smith et al. 2011). Model outputs also suggest that the spawning biomass will remain above 150,000 t over the next two years for catches below 45,000 t. However, if TACCs are increased significantly above the current base level of 30,000 t this would increase the level of risk to the stock/ecosystem and may require an increase in the frequency with which DEPM surveys are conducted and/or stock assessment reports are produced.

The previous harvest strategy for SASF did not include a lower limit reference point below which fishing would cease. The suggestion of Smith et al. (2011) that the yields from productive species such as sardine are typically reduced at depletion levels below 60% implies that setting the lower limit reference point for spawning biomass at approximately half the lower target reference point (i.e. 75,000 t) would be a conservative approach for the SASF. The rapid recovery of the SASF from the mass mortality events in 1995 and 1998, when the spawning biomass was reduced to that level or below also provides evidence that this limit is above the level at which future recruitment would be reduced and for this reason would be as a suitably conservative lower limit reference point for an ecologically important species such as sardine. If the next harvest strategy for the SASF were to adopt these reference points, reduced TACCs could be applied if the spawning biomass fell below 150,000 t and fishing may cease if the spawning biomass fell below 75,000 t. The exploitation rates used to guide TACC decisions could be varied to reflect the level of stock assessment undertaken. For example, exploitation rates up to 20% could be applied if DEPM surveys and stock assessment reports were undertaken biennially (as was the case in the previous harvest strategy). Alternatively, higher exploitation rates could be applied if DEPM surveys and stock assessment reports were undertaken annually.

Consideration also needs to be given to the method that will be used to estimate spawning biomass in future assessment undertaken under the next harvest strategy. Previously, the implications of the uncertainty associated with the estimation of spawning biomass discussed above have been addressed by using a demonstrably conservative approach (i.e. the log-linear model) to estimate the most critical and imprecise DEPM parameter, i.e. mean daily egg production (Ward et al. 2011b). Improving the precision of estimates of egg production has been identified as a priority for the SASF because of the current interest in exploring opportunities to increase the yield from the fishery. A research proposal is currently being developed to address this issue in both the SASF and Commonwealth Small Pelagic Fishery. Another option for addressing the imprecision of spawning biomass obtained using the DEPM would be to use model-generated estimates of spawning biomass as the key biological performance indicator in the next harvest strategy. However, there are also limitations associated with the use of model-based indicators.

#### *Age as a key biological performance indicator*

As identified above, there is significant uncertainty associated with estimating the age of sardine (e.g. Fletcher 1994; Rogers and Ward 2007). This uncertainty is significant

because age has been used previously as a secondary biological performance indicator for the SASF (Shanks 2005) and is a key input to the population model. Given the recent reductions in the age of sardine taken from southern Spencer Gulf, there may also be a need to include a measure of fish age (e.g. mean age of catch) as a key biological performance indicator in the next harvest strategy. Currently, uncertainty in the estimation of age is managed using the approach to age-determination described by Rogers and Ward (2007) which involves annual determination of the relationship between annuli-based age estimates and otolith weight. However, there is also potential to further improve accuracy and precision of estimates of population age structure by adopting the methods proposed by Francis and Campana (2004) and discussed in Chapter 3. Other methods should also be investigated for estimating a suitable age-based indicator for inclusion in the next harvest strategy.

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