

**Regrowth of pilchard (*Sardinops sagax*) stocks off southern
WA following the mass mortality event of 1998/99**

Final FRDC Report – Project 2000/135

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Fish for the future

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Researcher Tim Leary collects fresh pilchards from commercial fisher Graham Kennedy at Emu Point, July 2005. Photo: C. Anderson, Department of Fisheries.

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Non Technical Summary

Regrowth of pilchard (*Sardinops sagax*) stocks off southern WA following the mass mortality event of 1998/99

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Objectives

- 1 Document changes in the biomass of the four exploited adult assemblages of *S. sagax* in WA during the six year period following the 1998/99 mass mortality event.
- 2 Integrate results of biomass surveys into the age structured model for each assemblage of pilchards. Further develop the age-structured simulation models for *S. sagax* and investigate ongoing behaviour of these under conditions of different levels of information.
- 3 Determine if there is a measurable change in biological parameters of pilchards important to ongoing development of the simulation models, in particular growth rate, in response to the decrease in stock size.

Outcome achieved

This project produced time series of estimates of spawning biomass for pilchards in four purse seine management zones in Western Australia, three on the south coast and one on the west coast. The pilchard stocks in Western Australia have recovered strongly since the 1998/99 mass mortality.

This project has shown that the fishery independent and fishery dependent methods applied to Western Australia's pilchard fisheries cannot provide precise estimates of the size of the pilchard spawning biomass in each management region. Although the trends indicated by the age-structured simulation model are reliable and provide good evidence for a strong recovery of pilchard stocks, the magnitude of changes in stock size from year-to-year are not known with sufficient certainty to allow high rates of exploitation.

The demonstrated increases in pilchard spawning biomass has led to optimism for the future of the purse seine industry in southern WA, which in turn has encouraged this industry to be proactive in ensuring that exploitation rates were not set at levels that were too high to impede the recovery of the stock or put undue pressure on the long term viability of the stocks. Tracking the progress of estimating of pilchard biomass in each region over the duration of this project was a crucial factor that assisted industry members to better appreciate the need to examine factors additional to the point estimate of biomass generated in any one year.

The project also determined that the growth rates of pilchards in WA can be highly variable, sometimes changing markedly from year to year while at other times exhibiting more gradual changes over several years. The tendency for a negative relationship between annual growth rates and stock size suggests density dependence, which in turn supports the contention that pilchard stocks in this region are limited in size by the biological and oceanographic characteristics of the continental shelf waters of southern WA. The magnitude and variability of pilchard biomass observed during this project builds on the knowledge obtained during the 1990s, providing a strong basis against which the purse seine industry can expect future quota levels to be set.

In 1998/99 the population of pilchards off southern Australia suffered a 70% reduction in biomass due to a mass mortality caused by pilchard herpesvirus. Severe negative impacts on the Western Australian purse seine industry, which targets pilchards in three management regions on the south coast and one on the west coast, were immediate and there was an urgent need to ascertain whether there was a future for the industry. The importance of pilchards to the marine ecosystem off southern WA, to the communities for which purse seining was a key activity and to the broader communities of WA required that the pilchard stocks be monitored not only to ascertain whether they were recovering but to also increase our understanding of pilchard biology and ecology to assist with future management of the purse seine industry. This study used fishery-dependent and fishery-independent methods to examine changes in the pilchard biomass over a six-year period, plus developed methods to assess year-to-year changes in growth rates of pilchards.

The fishery independent method of calculating spawning biomass (the daily egg production method, DEPM) is based on surveying pilchard eggs using structured plankton surveys. These surveys were conducted at two-year intervals within each region, following on from series of similar surveys conducted in the 1990s. Each took place over a two week period at the peak

of the spawning season, with adult pilchards collected at the same time so that the output of eggs from an average sized pilchard could be estimated. Spawning biomass estimates were then derived through reconciling the egg abundance in the plankton within a region with the quantities of eggs produced by an “average” pilchard at that time.

For the south coast management regions the DEPM-based estimates were integrated into a simulation model, which included fishery dependent data. The simulation model utilizes information on factors that increase (e.g. recruitment, growth rate) and decrease (i.e. mortality due to fishing and natural causes) stock size to estimate changes in biomass over time. The fishery-dependent data used in the model were annual age-composition, which provides information of recruitment strength, and monthly catches. The age-composition of the stock in each region was determined from pilchard samples collected from commercial catches, continuing a sampling program that began in 1989.

The analyses completed in this project showed a strong recovery of pilchard stocks in WA from the very low levels that remained after the mass mortality event of 1998/99. Due to the nature of the pilchard’s biology and ecology, particularly their complex patterns of distribution and variable levels of recruitment of juveniles, it is not statistically appropriate to assume that the data collected will result in sufficiently precise estimates of the size of the pilchard stock in each management region to enable these point estimates to be used to set annual quotas. However, the time series of biomass estimates, along with crucial ancillary information such as variability in recruitment levels, obtained since the early 1990s has now enabled the WA Department of Fisheries to establish the likely range of the pilchard stock in each management region. The likely pilchard spawning biomass in each region if average to strong recruitment has been experienced over a number of years and environmental conditions favoured pilchards as the dominant baitfish species are:

West Coast	20,000 – 30,000 tonnes
Albany	15,000 – 30,000 tonnes
Bremer Bay	15,000 – 37,000 tonnes
Esperance	35,000 – 62,000 tonnes
Total	85,000 – 159,000 tonnes

The establishment by the various pilchard stakeholders during the late-1990s of an agreed “average” biomass level provided a benchmark against which management decisions could be made. This project has confirmed that the agreed levels were appropriate. Although estimates of pilchard biomass can occasionally exceed the above ranges, the recognition that the research results cannot provide precise estimates of biomass in any one year have helped the WA purse seine management system to adopt a medium- to long-term view. The requirement to plan several years ahead also recognizes that the pilchard biomasses can occasionally fall well below the above ranges. The pragmatic approach to management that has evolved during this project provides a superior alternative to a management approach that focuses solely on the year-to-year changes in the point estimates of spawning biomass.

Key words: pilchard (sardine), recruitment, age-structured model, spawning biomass, daily egg production method, stock recovery, fisheries management.

Acknowledgements

The purse seine industry has been very helpful in providing samples of fish from their catches - both fishers and processors have contributed to providing samples over the past 17 years.

Formal and informal discussions with industry members over the years has contributed substantially to the understanding of pilchards by fisheries scientists. Those authors who have had the opportunity worked directly with industry members in the latter's work place have invariably developed a better understanding of not only the fishery but also of the target species – the insights into spatial behaviour/patterns of pilchards gained through industry members has been invaluable for developing a better understanding for what the “hard data” can tell us.

We thank the many Departmental staff who participated in the plankton sampling, the most arduous aspect of the data collection. This entailed a particularly long and successful collaboration between Research and Compliance staff.

Norm Hall's advice on further development of the simulation model was invaluable.

1.0 Introduction

1.1 Background

1.1.1 Mass mortality

Between October 1998 and May 1999 a mass mortality of pilchards, *Sardinops sagax*, spread along the species' distribution around the temperate and subtropical coast of Australia (Figure 1). Pathological assessments indicated that a Herpesvirus was responsible, as in the similar mass mortality event of 1995 (Griffin et al., 1997; Hyatt et al., 1997; Whittington et al., 1997). The mortality rate caused by the virus in 1998/99 in South Australia (Ward and McLeay, 1999; Ward et al., 2001) and Western Australia (Gaughan et al., 2000), the regions with Australia's largest pilchard fisheries, were independently estimated to be around 70%, indicating the southern Australian stocks of pilchards had been seriously depleted over a very short period.

1.1.2 WA purse seine fisheries - industry affects

In south-western WA five purse seine management zones were established to reflect fishing, or proposed fishing, when the fishery became fully managed. There are currently four recognized functionally distinct adult assemblages (Edmonds and Fletcher, 1997; Gaughan et al., 2002), three on the south coast and one on the lower west coast (Figure 2) each of which requires estimates of biomass to provide a basis for setting total allowable catches (TACs). An age structured simulation model, which is tuned with fishery independent estimates of spawning biomass, has previously been developed for southern WA pilchards (Hall, 2000), and then subsequently modified (P. Stephenson unpublished data).

Spawning biomass of fish can be estimated using the daily egg production method (DEPM, e.g. Fletcher et al. 1996a). The Department of Fisheries has periodically undertaken DEPM surveys for pilchards in each management zone since the early 1990s. The strategy prior to the 1998/99 mass mortality was for one such DEPM survey each year, which alternated between the West Coast, Albany-Bremer Bay combined, and Esperance, thereby surveying each management zone once every three years. Previously, DEPM surveys have provided a basis for setting TACs; such estimates will now be integrated into the model for south coast pilchards, which is intended to provide a more formal means of setting TACs, as recommended by Cochrane (1999).

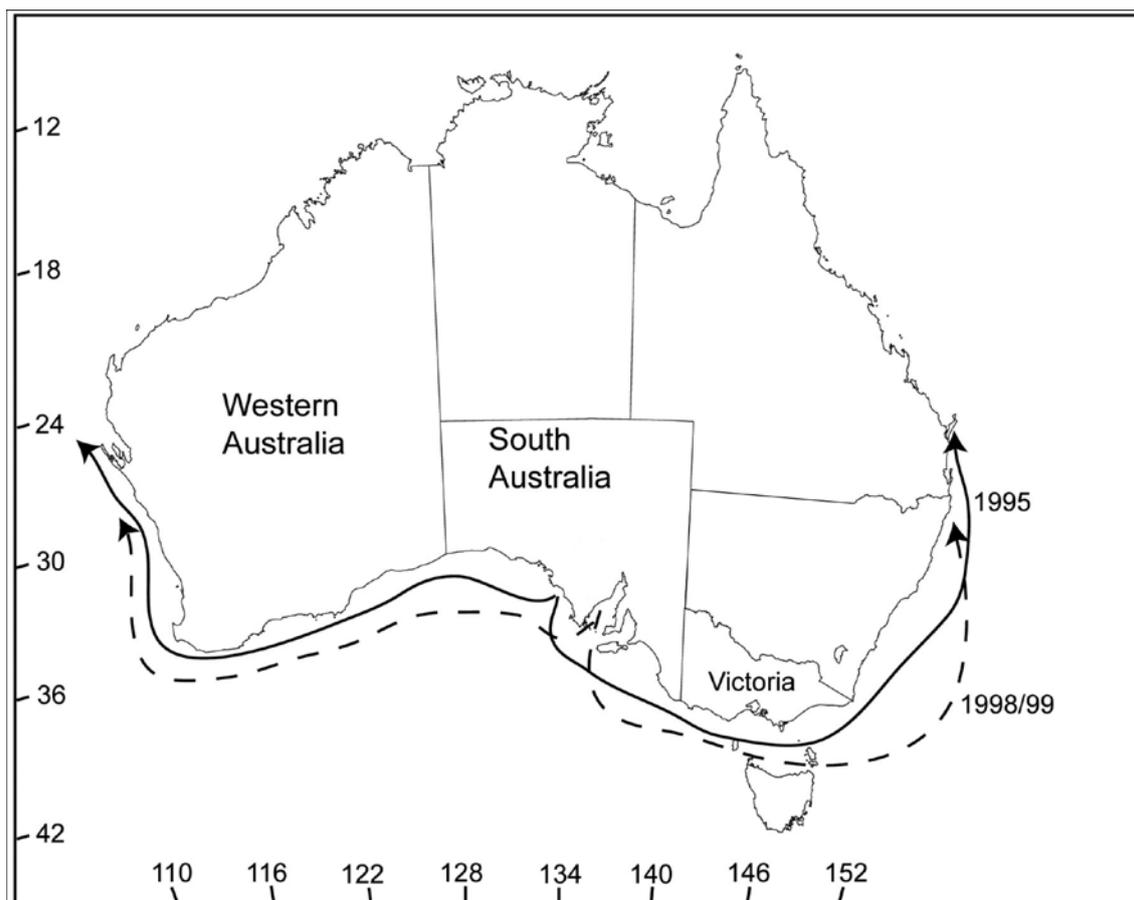


Figure 1. Extent of the 1995 and 1998/99 mass mortalities of *Sardinops sagax* around southern Australia from starting points along the central South Australian coast. Latitudes and longitudes are degrees south and east respectively. (From Gaughan, 2001).

At the start of this project the fishery for pilchards in southern WA had declined dramatically subsequent to the 1998/99 virus event. There was also a substantial decline in biomass of pilchards in the Albany zone after the mid-1990s, as was shown by marked decrease in catch rate in 1998 (D. Gaughan, unpublished data) that predated the epidemic mortality in this zone. However, after this epidemic, catches at Albany and Bremer Bay declined to negligible levels so vessel owners and crews left the area because of a lack of income. As a conservative measure, following the availability of preliminary estimates of the mortality rate of the Herpesvirus epidemic in WA, the TAC for each of the Albany, Bremer Bay and Esperance zones were substantially reduced and, for a period, were set to zero tonnes. The loss of pilchards was clearly sufficient to seriously impact on the commercial fishery, including processing establishments.

1.1.3 Management and research considerations

In order for WA's pilchard stocks to be successfully managed in the long term using the simulation models, it was considered imperative that regular time-series of both catch-at-age data and DEPM estimates of biomass continue to be obtained. In particular, because Cochrane (1999) has stressed the need to include formal risk analysis in management of WA pilchards and that elements of such analyses should be built into the simulation models, if these recommendations were to be followed then there was a pressing need to continue the process of fine-tuning the models. The mass mortality of pilchards was expected to negatively impact the ongoing development of the simulation model in two ways.

Firstly, the dramatic decrease in stock size may cause biological changes in the various pilchard assemblages (e.g. growth rate could increase, see below). The population of *S. sagax* in south-western Australia is postulated to be severely limited in size compared to elsewhere due to the poor nutrient levels of shelf waters in the region (Gaughan et al., 2001a, 2004). As such, large changes in stock size may result in relatively rapid density-dependent changes in growth rate of magnitudes sufficient to influence outputs of stock assessments. The potential for a density-dependent effect on growth rates on *S. sagax* off south-western Australia was exacerbated by the mass mortality. Relatively large changes in spawning biomass are typical for small pelagic fish such as *S. sagax*, but a 70% change within one year, and indeed within 1-2 months, for this species off south-western Australia is atypical for marine teleost populations and provided conditions suitable for determining the extent to which growth rates may respond to variable stock abundance. “Density dependence is a fundamental concept in the study of fish population dynamics” (Rose et al., 2001), with growth and survival generally acknowledged to be density dependent (Post et al. 1999) but there has yet to be an investigation into this phenomenon for pilchards in WA. Given the limited capacity for growth of pilchard populations in WA (Lenanton et al., 1991; Gaughan et al., 2001b), density dependent effects could be considerable.

Secondly, the decrease in catches at some regions flows through to fewer samples from which to estimate annual catch-at-age, a crucial data requirement of the age-structured simulation models. Further DEPM surveys were planned for each pilchard fishing region in WA to circumvent the anticipated gap in required data so as to allow continued development of the simulation models and to assess the level of risk associated with different levels of catch.

1.1.4 Ecosystem effects

Besides the negative impacts on the purse seine fisheries in WA, the potentially disastrous ecosystem effects of losing a huge quantity of a dominant clupeoid (e.g. Berdnikov et al., 1999, to take an extreme example) also need to be considered. The loss of an estimated 70% of the pilchard stock over a very short period is cause for concern since pilchards occupy a pivotal position of energy transfer in food webs in which they occur (e.g. Cole and McGlade, 1998). In regions where they occur, pilchards constitute a major conduit between primary production (i.e. phytoplankton) and a variety of large predators from the fish, bird and mammal groups (e.g. Cole and McGlade, 1998).

In a modelling exercise aimed at assisting in the development of management principles of exploited pelagic ecosystems, Mackinson et al. (1997) suggested that a large decrease in stock of small pelagic fish was likely to result in increased biomass of their food (plankton) and competitors (other small pelagic fish), and a decrease in the population of their predators. Furthermore, the higher trophic levels took the longest time to recover.

The importance of pilchards in southern Australian pelagic ecosystems has recently been summarized by Ward and Jones (1998). There are likely to be a variety of effects following the recent massive stock loss of pilchards from southern Australia. Other commercially/recreationally important species which may be impacted by the loss of pilchard stock include many of the larger pelagic predators which occur in southern Australia, including western Australian salmon, various species of sharks, southern bluefin tuna (SBT), other tuna species and billfish species.

SBT juveniles migrate south along the western Australian coast and then east to subsequently arrive in the Great Australian Bight (GAB). Tagging data suggests that the GAB region constitutes an important feeding ground and that SBT grow substantially while resident in this region. As *S. sagax* are recognized, although infrequently quantified (Serventy, 1956), as an

important prey item of SBT, the 70% loss of the *S. sagax* stock off southern Australia could have a significant impact on the SBT stock. For example, if the migration to the GAB is to access 'good growing conditions' and *S. sagax* stocks in that region constitute an important food source, then a major loss of *S. sagax* could cause a large decrease in fitness and potentially an increase in mortality of SBT. Furthermore, a long term reduction in stock size of pilchards and the presumed decreased fitness of SBT could affect their migration patterns, similar to the change in the proportions of northern bluefin tuna crossing the northern Pacific when the Japanese pilchard stock decreased (Polovina, 1996). This very brief example of altered migration is only one of many possible consequences for SBT and a suite of other species.

Some immediate impacts of the 1998/99 mass mortality of pilchards were obvious, albeit not quantified in most cases. Following the mass mortality there was concern that other negative impacts, potentially longer-term, would occur but be difficult to identify. Because the magnitude of such effects may be large and wide ranging, the regrowth of the pilchard stocks must be documented. The best way of doing this is to use age-structured models tuned using DEPM surveys of spawning biomass (Cochrane, 1999).

1.2 Need

This project was undertaken to assess the regrowth of pilchard stocks in southern Australia because of (1) the socioeconomic problems associated with fishery declines and the associated need to provide sufficiently detailed information so that management could proceed with the appropriate balance between sustaining the fishery and allowing recovery of the stocks, and (2) the potential for causes of wide ranging ecosystem affects to go unrecognized if there is no baseline data (i.e. abundance) for this key pelagic species.

In order to provide a time-series of the regrowth of pilchard stocks in WA the biomass size of each adult assemblage was assessed over a period of six years. Although this is a relatively short period of time over which to examine the recovery of a pelagic stock (e.g. see Beckley and van der Lingen, 1999; Schwartzlose et al., 1999), such a routine will provide a basis for assessing the regrowth of pilchard stocks.

The integration of spawning biomass, age composition and catch data in an age-structured simulation model should enable prediction of the impacts of further mortality events involving pilchards in southern Australia and facilitate the decision-making process in the ongoing management of pilchards in WA.

1.3 Objectives

1. Document changes in the biomass of the four exploited adult assemblages of *S. sagax* in WA during the six year period following the 1998/99 mass mortality event.
2. Integrate results of biomass surveys into the age-structured model for each assemblage of pilchards. Further develop the age-structured simulation models for *S. sagax* and investigate ongoing behaviour of these under conditions of different levels of information.
3. Determine if there is a measurable change in biological parameters of pilchards important to ongoing development of the simulation models, in particular growth rate, in response to the decrease in stock size.

2.0 Methods

2.1 The daily egg production method (DEPM)

2.1.1 Overview of fieldwork and sampling strategy

There are currently four recognized assemblages of adult pilchards in southern WA, with three of these on the south coast (Esperance, Bremer Bay and Albany) and one on the lower west coast (Figure 2). Since the mid- 1990s, pilchard spawning biomass has typically been estimated, using DEPM surveys, at one of these assemblages once every three to four years (see Gaughan et al., 2004). Following the mass mortality in 1998/99, the high levels of concern for the pilchard stocks and the industries that relied on them lead to the development of this project so that a higher frequency of DEPM surveys could be undertaken. However, for the purposes of undertaking biomass estimates on a manageable scale, DEPM surveys could not be undertaken at each purse seine fishery in WA in each year. Rather, the south coast and west coast regions were surveyed in alternate years, noting that despite the occurrence of distinct adult assemblages along the south coast (Gaughan et al., 2002), each contributes to a single breeding stock while the west coast pilchards constituted a second breeding stock.

DEPM surveys followed the techniques originally provided by Lasker et al. (1985) and used by Fletcher et al. (1996a) off southern WA. DEPM surveys of spawning biomass of each adult assemblage were undertaken as shown in Table 1. The aim for both the south and west coast breeding stocks was to conduct a DEPM survey every two years over the duration of the project, resulting in a total of three surveys for each region. However, because of healthy biomass levels recorded at Esperance in 2004, the final survey for the Esperance region was instead replaced with an exploratory search for pilchards outside of the traditional fishing grounds aboard a commercial purse seine vessel and using industry-standard searching methods (see Section 2.3.2).

The surveys were timed to coincide with the period of maximum spawning activity in each region (Gaughan et al., 2002). While it would have been preferable to undertake the entire south coast survey (Albany zone to Esperance zone) in a continuous manner, this was precluded by the differences in spawning periods between the Esperance and Albany/Bremer Bay zones.

Table 1. Sequence of DEPM surveys.

Sample location	First survey	Second Survey	Third survey
West Coast	2000 – July	2002 – August	2004 – July
Albany	2001 – July	2003 – July	2005 – July
Bremer Bay	2001 – July	2003 – July	2005 – July
Esperance	2002 – Jan.	2004 – March	Replaced with industry acoustic survey ¹

1. See text and Section 2.3.2

2.1.2 DEPM model

The daily egg production method (DEPM) employs the following equation to relate the numbers of eggs found in the plankton samples to the numbers of eggs produced by spawning

females in the region at the same time. This is combined with data on the average weight and sex ratio of adult pilchards to give an estimate of spawning biomass in tonnes. The DEPM thus requires two distinct sampling programs, one for the eggs (plankton sampling) and one for the adults (from commercial catches).

$$\text{Biomass} = \frac{(\text{Spawning area} \times \text{Egg production} \times \text{Weight} \times k)}{(\text{Spawning fraction} \times \text{Fecundity} \times \text{Sex ratio})}$$

where

Spawning area (km²) is the area that contained pilchard eggs up to 1 day old (i.e. Day 1 eggs);

Egg production (nos./0.05 m²) is the initial numbers of eggs spawned;

Weight (grams) is the average weight of an adult female;

k (= 20) is conversion factor to bring the various units to a value in tonnes;

Spawning fraction is the proportion of mature females that spawn per day;

Fecundity is the average number of eggs produced by an average sized mature female;

Sex ratio is the ratio of females to total adults by weight

2.1.3 Adult sampling

For each survey, samples of adult pilchards were collected from the purse seine fleet to estimate the adult parameters of spawning fraction, weight, and sex ratio. This involved obtaining samples as they are caught, typically at sea, so as to maximize the quality of preserved ovaries. It was assumed that the adult fish obtained from the commercial catches used in DEPM surveys provided an unbiased sample for the estimation of the weight, sex ratio and fecundity of the entire spawning population in each region. Samples of ovaries were prepared for microscopic examination using standard histological techniques; spawning fraction was estimated from the proportions of female pilchards that had recently spawned, as evidenced by the presence of post-ovulatory follicles. Samples of whole ovaries were retained so that fecundity could be estimated.

Segregation of sexes at certain, but poorly understood, periods during the peak spawning periods can cause biases in the estimation of sex ratio when numbers of adult samples were low. The maximum allowable proportion of females permitted for the DEPM surveys was therefore set at 65%, which approximates the annual sex ratio determined from routine biological sampling.

2.1.4 Plankton sampling

Plankton sampling for any one survey were conducted aboard one of the Department of Fisheries' patrol vessels plus two or three chartered vessels. The patrol boat undertook the bulk of the plankton survey, with chartered vessels concentrating on the areas adjacent to port, or areas between ports which could be covered by day-trips. Plankton sampling followed inshore – offshore transects spaced across the zone being surveyed, resulting in a grid of sampling stations sufficient to adequately represent the distribution of eggs (Fletcher and Sumner, 1999). Plankton samples were taken across the continental shelf to ensure coverage of the spawning habitat of pilchards (Fletcher et al., 1996a; Gaughan et al., 2004). For consistency, it was attempted to cover the same regions sampled for DEPM surveys undertaken during the 1990s (Gaughan et al., 2002, Figure 3).

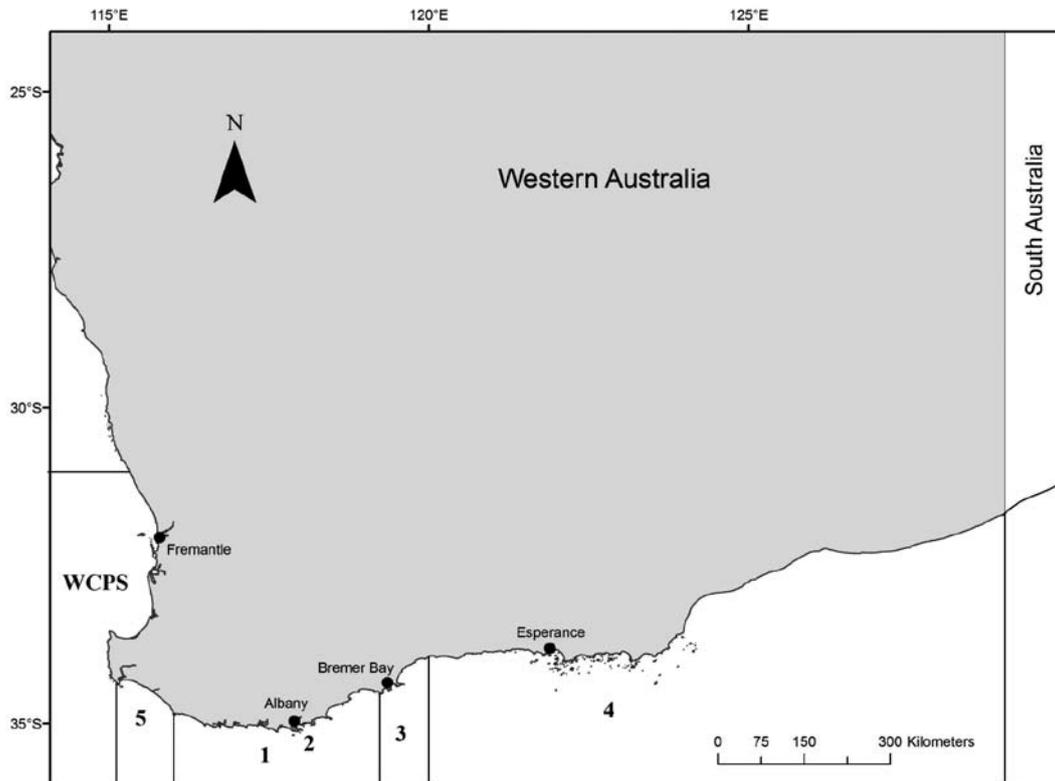


Figure 2. Purse seine management zones on the south (Zones 1–5) and lower west coasts (West Coast Purse Seine) of Western Australia. Zone 2 is a small inshore zone that covers King George Sound, the original location for purse seining at Albany. This zone was enacted in recognition of the need to ensure that the Albany purse seine fleet did not overexploit the inshore aggregations of pilchard.

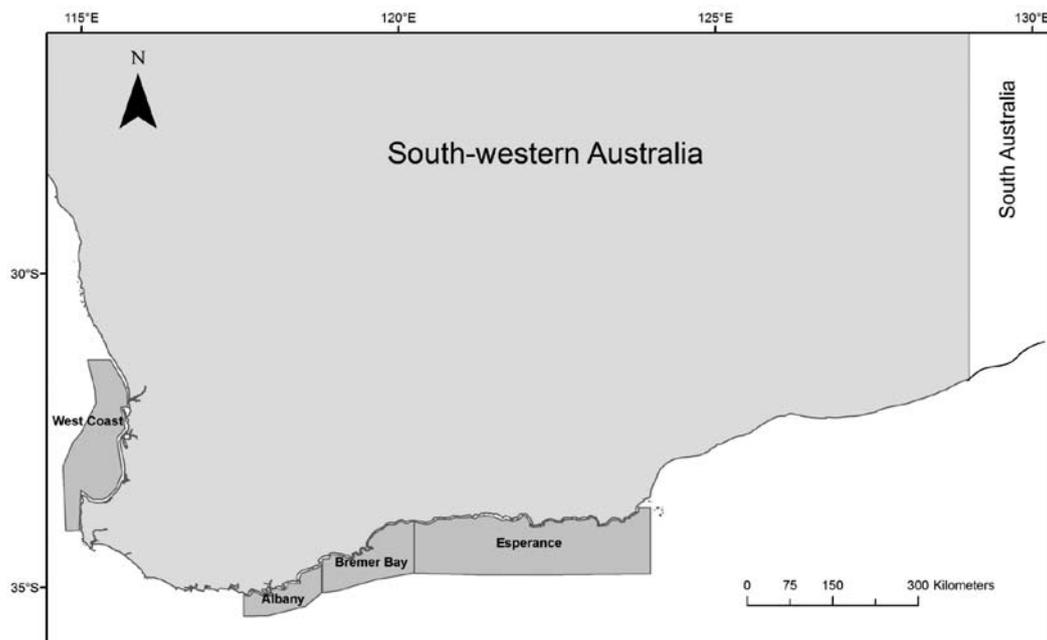


Figure 3. The extent of DEPM surveys carried out in purse seine management zone in south-western Australia from 2000–2006.

2.1.5 Estimation of egg production

For the purpose of DEPM, “egg production” refers to the estimated number eggs spawned by the adult population on a given day. The method for applying the regression model of egg abundance versus time to estimate egg production (and the coefficient of variation, CV) is the same as that used by Fletcher et al. (1996a, b) and that has been routinely applied to the Californian sardine fishery (e.g. Lo et al., 1996). However, rather than having a prescriptive, and hence inflexible, method of estimating P_0 from the regression model, we employed three alternatives aimed at obtaining the best fit of the regression to the data. Lo et al. (1996) suggested that because stage 12 eggs are the last prior to hatching, their abundance will be underestimated because it is not usually known how many of the eggs have already hatched into larvae. Consequently, eggs of up to stage 11 only were used for the base-model initially employed in this study. If the fit to the regression model was unacceptable, stage 12 eggs were then included and the model refitted. If this model was unacceptable, the yolk-sac larvae were also included as an additional developmental stage. For the duration of this project, each of these three alternatives were fitted, regardless of whether the first appeared to be acceptable or not. The choice of the three alternatives was based on which fit gave the lowest CV.

In some cases the generation of P_0 estimates produced counter-intuitive results whereby the slope of the regression relationship was positive; while mortality of eggs dictates that the slope of the relationship must be negative, a positive slope is indicative of a sampling regime that cannot account for short-term variability in spawning and or patchiness of eggs. Considering that the egg data collected in any one survey contained some level of information regarding the actual abundance of eggs in the water we therefore undertook to use this data using the following method. The spawning area of the survey was iteratively increased by included further “zero” stations than were contained in the original estimate of spawning area. This then changed the estimation of P_0 through inclusion of more zero values in the egg-count data. The approach is intuitively correct within the context of the DEPM because including more “zero” stations will decrease the estimate of egg abundance but concomitantly increase the estimate of spawning area.

2.2 Integrated fishery model

The pilchard population model for each south coast region, originally developed by Hall (2000), and revised during this project, integrated estimates of spawning biomass, age-composition and catch data for each of the south coast pilchard fisheries. The model outputs included estimates of spawning biomass, annual recruitment and age composition. The model is described in Appendix 3.

2.3 Acoustic and visual detection of pilchard schools

2.3.1 DEPM surveys

Cochrane (1999) recommended the use of acoustic techniques to help determine biomass of pilchards in WA. Because it was our intention to conduct much of the plankton sampling aboard one of the Department of Fisheries’ patrol vessels, it was anticipated that there would be a good opportunity to concomitantly undertake acoustic sampling. The ability to fit the sonar unit from the Department’s research vessel to a patrol vessel was investigated, but

the fitting of hull-mounted sonar transducers to the patrol vessel was found to represent an unacceptable compromise to the performance of the patrol vessel, which has a planing hull. Likewise, fitting of retractable transducer pods was not an option due to space restrictions in the engine room of the patrol vessel. Systematic acoustic surveys intended during the plankton sampling were therefore not performed as originally planned. However, evidence of pilchard schools from either visual observations (e.g. feeding activity by seabirds or dolphins) or echo sounder readings was routinely recorded during each plankton survey and marked on the plots of pilchard egg distribution for these surveys.

2.3.2 Specific exploratory survey

Rather than undertake the final DEPM survey for the Esperance region in 2006, an exploratory survey for adult pilchards east of Esperance was undertaken using an industry vessel, including an experienced skipper and crew. This survey was undertaken in response to industry concerns that despite the DEPM surveys indicating recovery of the spawning biomass, the industry itself had not recovered. This failure of the fishery to recover has primarily been a result of a preponderance of small pilchards, often too small to market, in the traditional fishing grounds. Market-sized pilchards have been only irregularly present in the inshore fishing grounds of Esperance. Thus, 1- and 2-year old pilchards have dominated the catch at Esperance since fishing resumed after the mass mortality. The DEPM surveys have shown, through the presence of eggs, that there are mature/spawning pilchards offshore and east of Esperance, but it is not known how big these fish are, nor how old. The aim of the survey was to determine if there were appreciably more pilchard schools outside of the fishing grounds and to determine the size and age of any such pilchards.

The survey was conducted over a week in May 2006 (03 – 10 May), extending eastwards from Esperance for approximately 240 km and traversing both through both nearshore and offshore waters. The vessel's sonar (colour sonar with industry standard features) was used to check for sub-surface schools forward of the vessel, while a watch in all directions for surface activity indicative of pilchards was maintained from the vessel. When conditions were appropriate, gillnets with mesh sizes of 13, 19 and 25 mm, which had previously been used to catch pilchards, were variously deployed to catch fish that had been detected.

2.4 Growth rates

2.4.1 Sampling

Samples of pilchards have been routinely collected from commercial catches since 1988, providing biological data for the entire year, but weighted to reflect the seasonality of catches. Typically 25 pilchards are sampled from each of 30 – 100 catches per year from each management zone (see Gaughan et al., 2002 for details). Sex, fork length (FL), whole body weight (BW), gonad weight and otolith weight (OW) were recorded. The OW – age key developed by Fletcher (1991, 1995) has allowed annual age composition to be determined for each region using large numbers of samples. Although the samples were collected from only a small proportion of the area occupied by the stock, the OW method of ageing pilchards appears satisfactory as the age compositions show the presence of strong cohorts moving through the fishery over periods of several years (Gaughan et al., 2002).

2.4.2 Preliminary analyses

The time series of biological measurements, and retention of otoliths, from regular catch sampling provided an opportunity to assess growth rates of pilchards from before and after the 1998/99 mass mortality event. Historically, only pilchards from Albany and the West Coast had been aged using the annuli method. In this study we intended to extend this to all regions, including retrospective ageing of pilchards using archived otoliths collected during the 1990s. Age determination using otolith increments were completed for Esperance pilchards from 1998 and 2000, i.e. immediately before and after the 1998/99 mass mortality event. Counting increments on whole pilchard otoliths proved difficult: agreement between consecutive readings was only 28%, less than the 45% expected (Fletcher and Blight, 1996). This may have been due to a regional difference; Fletcher and Blight (1996) aged otoliths only from Albany, which has a different annual water temperature pattern than Esperance, and likely also experiences a different pattern of annual primary productivity (L. Twomey, Murdoch University, pers. comm.). Regardless of the cause for the lower interpretability of internal structures of otoliths from Esperance pilchards, there were insufficient data points to fit a reliable growth curve. Therefore, a more stringent protocol for counting otolith increments was subsequently developed.

Consistency in interpreting otolith increments for pilchards improved once the protocol had been developed. Repeatability in counting annuli improved to 39% from the 28% obtained previously. However, although samples of ~200 pilchards from each of 1998 and 2000 were aged, with only 39% repeatability the numbers of data points remained insufficient for reliably estimating growth using the k parameter of the von Bertalanffy growth model. Because processing and reading of otoliths was a resource-intensive technique, which provided only low numbers of age estimates, this method was discontinued and as such variability in growth rates could not be directly quantified. The focus of this study changed to determining whether an indirect method would be suitable for assessing if there had been relative changes in growth rates.

Given the large data set for OWs that was available for pilchards in WA, this parameter provided the basis for assessing relative changes in growth rates. OW is closely related to age for a wide variety of fish species (e.g. Pawson, 1990; Worthington et al., 1995; Lou et al., 2005), but for the purposes of this study, and given the above difficulties, we were not interested in the ages of pilchards. Instead, because we wished to ascertain if there were predictable variations in relative growth rate, we compared FL directly against OW. Fletcher (1991) previously established that FL was more variable than OW for a given age and that a heavier otolith for a given body size is indicative of slower growth, and *visa versa*. That is, slower growing fish have proportionally larger otoliths, while faster growing fish have proportionally smaller otoliths. We therefore used the relationship between OW and FL as an indicator of relative growth rate.

2.4.3 Modelling of otolith weight versus fork length

Plots of OW against FL show that between 1996 (medium stock level) and 2002 (very low stock levels) there was a decrease in OW for a given FL (Figure 4). Under the assumption that faster growth results in comparatively smaller otoliths for a given body size, the change in the OW-FL relationship between 1996 and 2002 is indicative of an increase in growth rate between the two periods. However, plots of fitted curves for all years (Figure 5) showed considerable variability, which may have been due to “true” variation in growth rates but could also have reflected the interannual variability in the sizes (and ages) of pilchards that occurred in the “stocks” within each region.

Preliminary modelling confirmed that year-year comparisons were not always valid when the OW-FL relationships were examined across all OWs because of the significant gaps in some age-classes in some years that resulted from highly variable recruitment (see Appendix 4). If there is density-dependence, because pilchards can live to 8+ years, estimates of a growth coefficient will be increasingly confounded by interannual changes in stock size as the fish's age increases. This problem of examining stock-wide variability in growth for a short-lived species with variable age structure represents a considerable statistical challenge that will continue to be pursued, with the aim of seeking independent peer-review.

For the purpose of this project, analyses subsequently focused on a restricted OW range, specifically that corresponding to pilchards in the 1+ and 2+ year classes. The samples for years 1989 to 2005 were thus restricted to otolith weights between 80 mg and 138 mg (i.e. approximating the 1+ and 2+ year classes). We recognize the circularity in using a single OW – age key developed in one region to examine variability in growth rates at specific ages, thus acknowledging that the OW at any one age would change in response to a change in growth rate.

While this restricted OW range encompasses recruit-aged fish rather than the (sometimes) more abundant year-classes available to the fishery, it was chosen so that the period of growth more closely reflected the time at which the spawning biomass was estimated. Furthermore, the growth rates of pilchards at these young ages are relatively fast (e.g. Fletcher and Blight, 1996), and respiration rates of fish typically decrease with increasing size (e.g. Brett and Groves, 1979). Consequently, any density dependent response of growth rates would be expected to be more detectable in young pilchards and would be acting in the period leading up to full recruitment to the fishery.

For each year, a linear regression model with zero intercept was fitted for FL (mm) against OW (mg). Since the OW range was restricted, the linear regression was a simple and reliable approximation of the power curve relationship over the full domain of otolith weights (Appendix 4). The slope was interpreted as an average growth rate coefficient in the sense that a higher (or lower) slope indicates an expected longer (or shorter) fish, given a specified otolith weight.

Growth coefficients developed from these fish were then compared to stock size in (a) the same year as the fish were caught and (b) the previous year (i.e. a one-year lag). Comparing average growth rates of fish from the contemporaneous year and the previous year was undertaken to reflect the biomass levels, and hence densities, experienced by the 1+ - 2+ age groups of pilchards. We recognize that this indirect method will not provide a cause-effect explanation for variability in growth rate, which is inherently difficult to determine (e.g. Sass and Kitchell, 2006), but may nonetheless indicate if it is likely that pilchards in WA waters can be subjected to density-dependent limitations on growth.

The null hypotheses for this study was that there is no predictable difference in the OW-FL relationship at different stock sizes, which will be tested against the alternative hypothesis that there is a predictable change in the OW-FL relationship at different stock sizes.

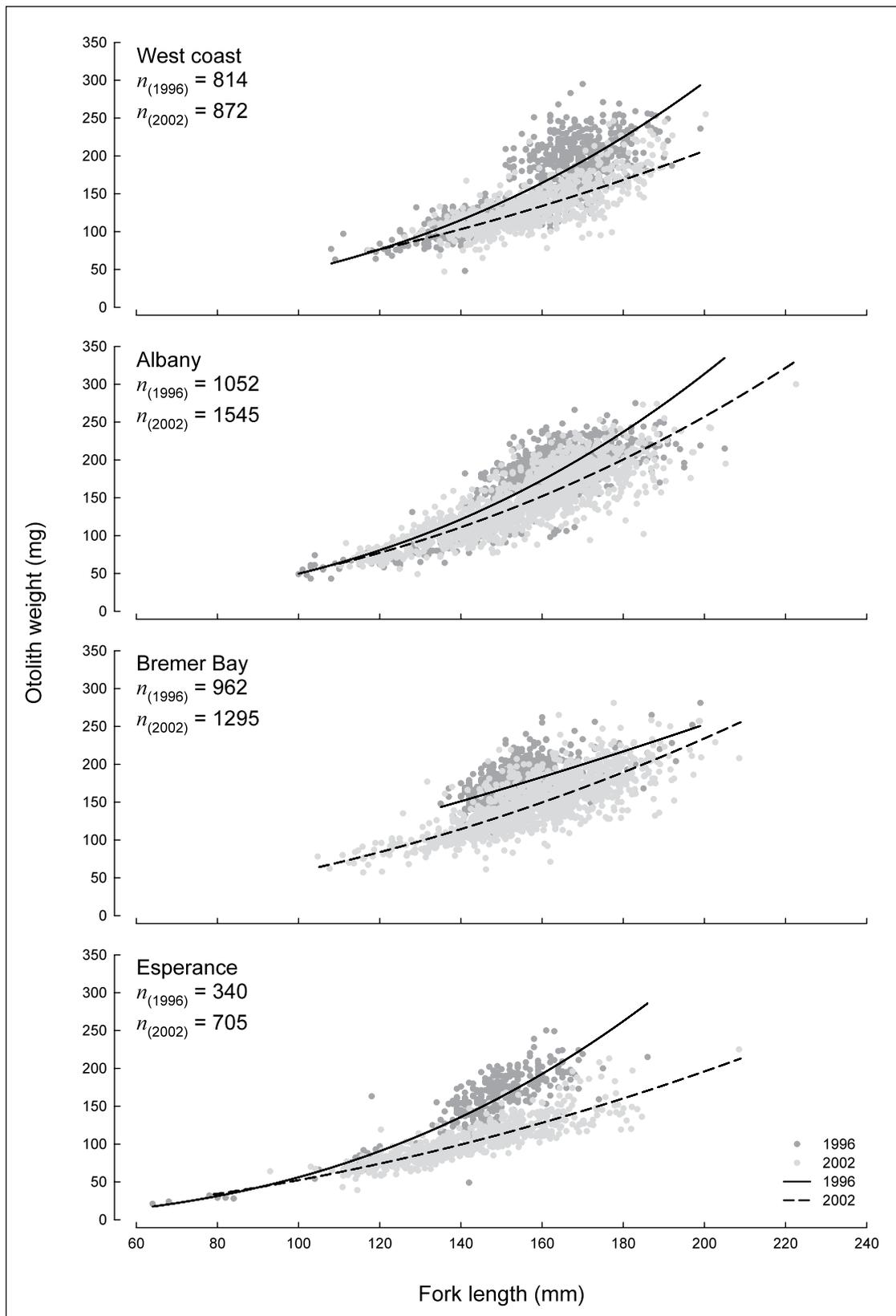


Figure 4. Relationship between otolith weight and fork length for pilchards from four locations in Western Australia (West Coast, Albany, Bremer Bay, Esperance) in 1996 and 2002.

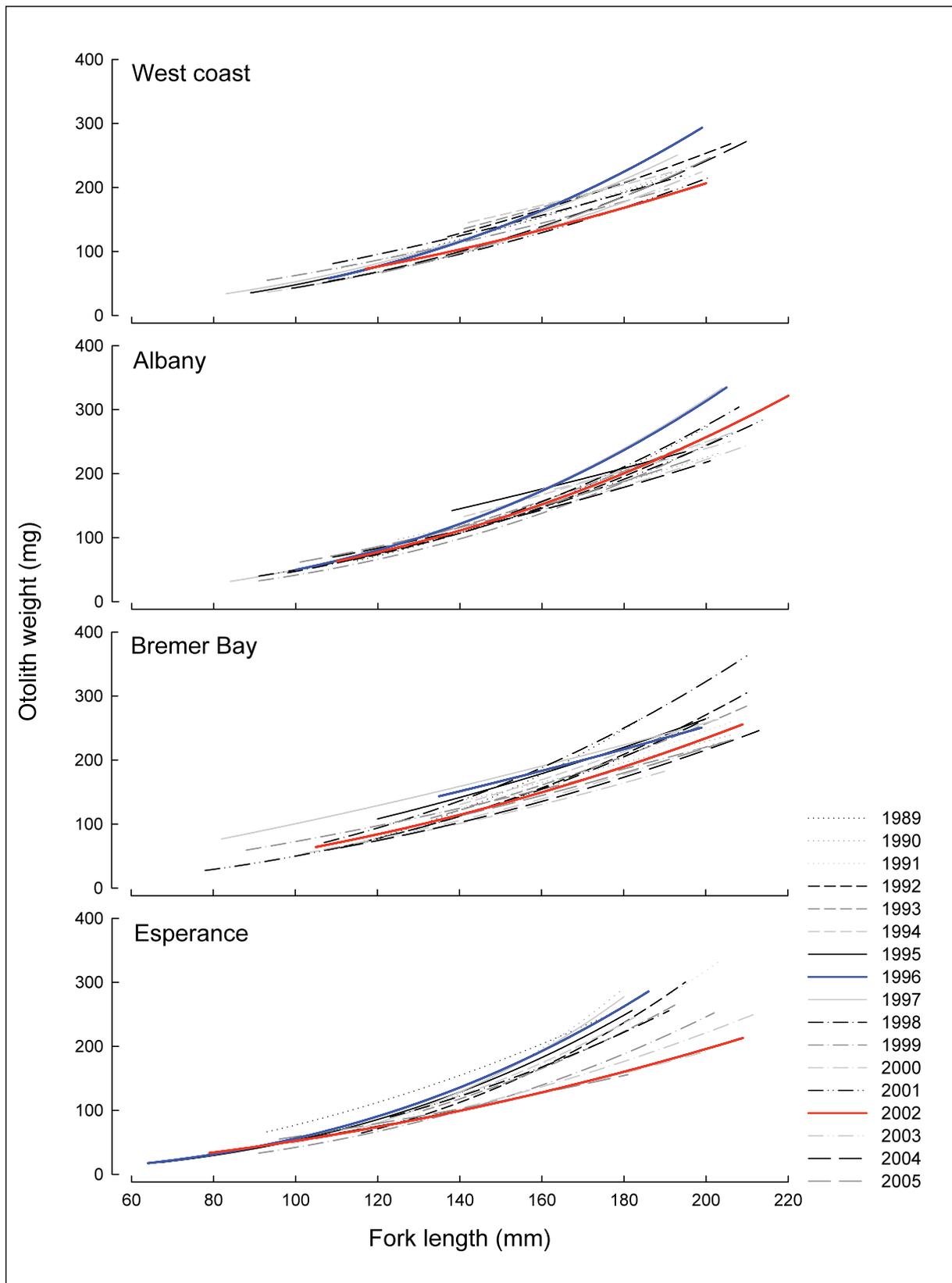


Figure 5. Annual (1990 – 2005) relationships between otolith weight and fork length for pilchards from four locations in Western Australia (West Coast, Albany, Bremer Bay, Esperance).

2.4.4 Interannual variability in body condition

There can be considerable variability in the weight of a fish for a given length. If there are density-dependent effects on growth rate of pilchard, such effects may be expressed in the condition of the fish (e.g. van der Lingen and Durholtz, 2003), with well-conditioned fish having a larger body weight (BW) for a given length. We therefore also assessed the BW – FL relationship, as an indicator of condition, and compare these against spawning biomass estimates to further investigate density-dependent effects. The growth coefficient in this case was the cubic coefficient of the BW-FL model

$$FL = a.BW^3,$$

Annual coefficients were estimated for the available pilchard data in each year up to 2005 for each of the four management zones. As for the OW – FL relationships, we restricted these estimates to the 1+ and 2+ year classes. We had anticipated that the gonadosomatic index (gonad weight as a proportion of body weight) might vary in response to stock size, but preliminary examination of the data revealed such a high degree of variability that further analyses were not undertaken. As with the length-at-age analyses, we compared the BW-based growth coefficient against levels of spawning biomass in the same year that the fish were captured and in the previous year.

3.0 Results/Discussion

3.1 Spawning biomass

This section addresses Objectives 1 and 2 of the project.

Objective 1. Document changes in the biomass of the four exploited adult assemblages of *S. sagax* in WA during the six year period following the 1998/99 mass mortality event.

Objective 2. Integrate results of biomass surveys into the age-structured model for each assemblage of pilchards. Further develop the age-structured simulation models for *S. sagax* and investigate ongoing behaviour of these under conditions of different levels of information.

3.1.1 DEPM surveys

The DEPM surveys in this project were conducted at each of the four purse seine management zones (West Coast, Albany, Bremer Bay, Esperance) as shown in Table 1 (Section 2.1.1). The sampling areas for each plankton survey are shown in Figure 6, along with the distribution of pilchard eggs < 24 hours old (Day 0 eggs). The numbers of samples collected, estimates of the DEPM parameters and estimates of spawning biomass are shown in Table 2. The DEPM is designed to account for changes in values of the parameters from one survey to the next. Thus, while it is desirable for the survey to be undertaken at the time of peak spawning, there is inter-annual variability in level of spawning activity that cannot be predicted hence estimates for single parameters are not always comparable between years. For example, variations in estimates of either spawning fraction or egg production between years, when considered in isolation of the other DEPM parameters, does not imply that spawning biomass has changed.

Due to inherent variability in the spatial distribution of pilchard eggs at any given time, a successful plankton sampling program is no guarantee of obtaining a robust estimate of egg production. Thus, for some surveys CVs were very high (e.g. West Coast 2002, Esperance 2002). The egg mortality relationship, from which the egg production parameter is estimated, is typically the greatest source of variation in DEPM estimates of spawning biomass. It is typical for CVs for egg production to be in the order of 0.3 or higher (e.g. Stratoudakis et al., 2006).

While the plankton sampling component of DEPM surveys was generally successful, collection of adult samples was less successful. The numbers of adult samples collected was less than 10 on six occasions, with none obtained for the survey in Bremer Bay in 2003. This scarcity of samples of adult pilchards was typically due to low availability of mature-age pilchards in the fishing grounds at the time of the survey. The low numbers of adult samples can contribute to wide confidence limits for the estimates of spawning biomass, but even in those cases where a limited number of adult samples have resulted in parameters estimates with low CVs, the representativeness of the samples of the entire stock remains suspect.

Estimates of spawning biomass were obtained from each of the DEPM surveys undertaken for West Coast, Albany and Esperance, but for only one (i.e. 2001) of the three surveys at Bremer Bay. The Albany and Esperance regions showed consistent increases in spawning biomass after the 1998/99 mass mortality, with the 2005 estimates of 63,452 and 113,044 tonnes, respectively, far exceeding historical levels (Table 2, Figure 7). The West Coast pilchard stock increased from 9669 tonnes in 1999 to 12,819 tonnes in 2002, followed by a considerable increase to 31,828 tonnes in 2004.

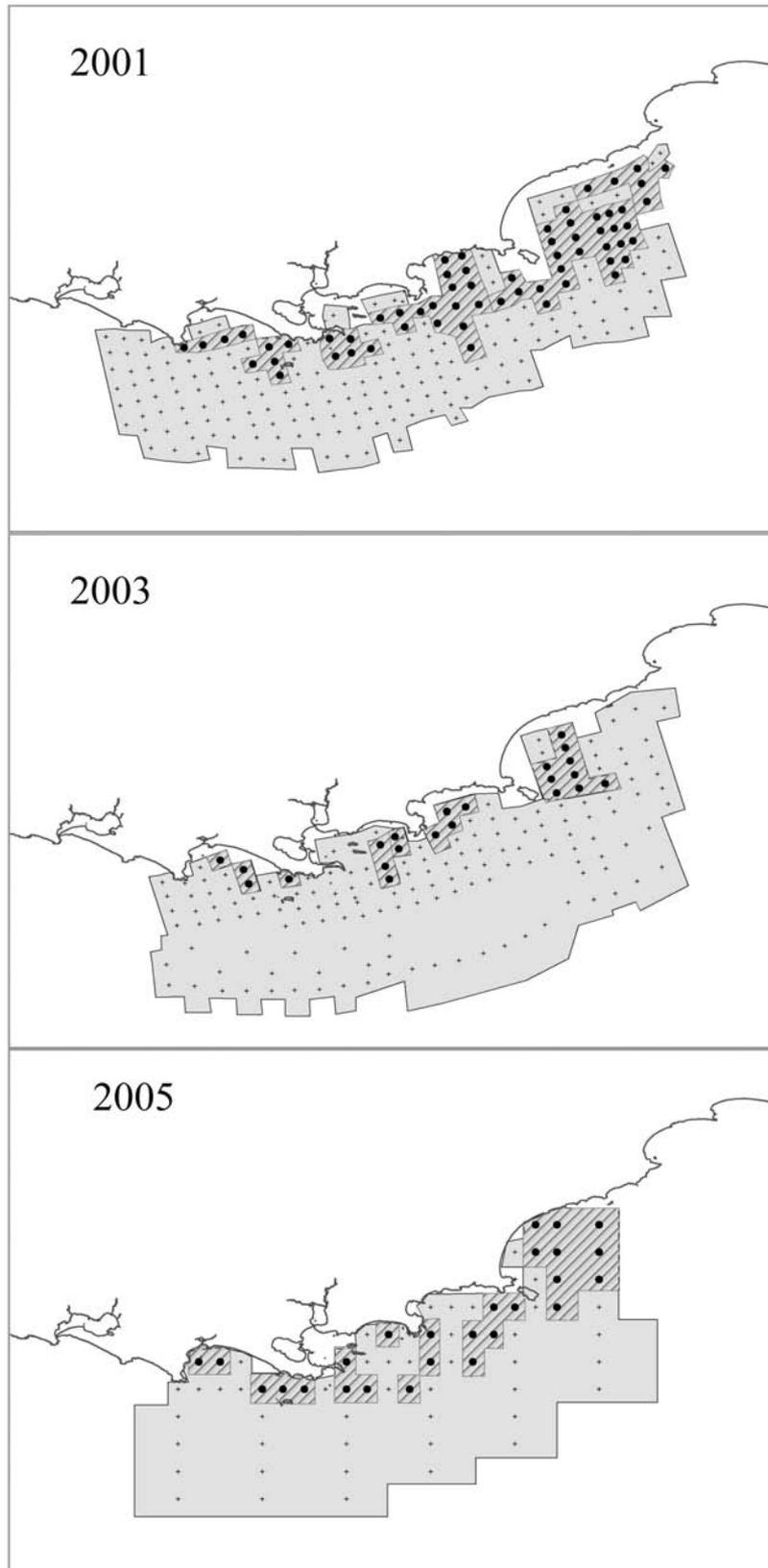


Figure 6a. Plankton survey area (grey) and pilchard spawning areas (hatched) for DEPM surveys at the Albany region of south-western Australia in 2001, 2003 and 2005. The larger dots depict sampling stations at which Day 0 pilchard eggs were found.

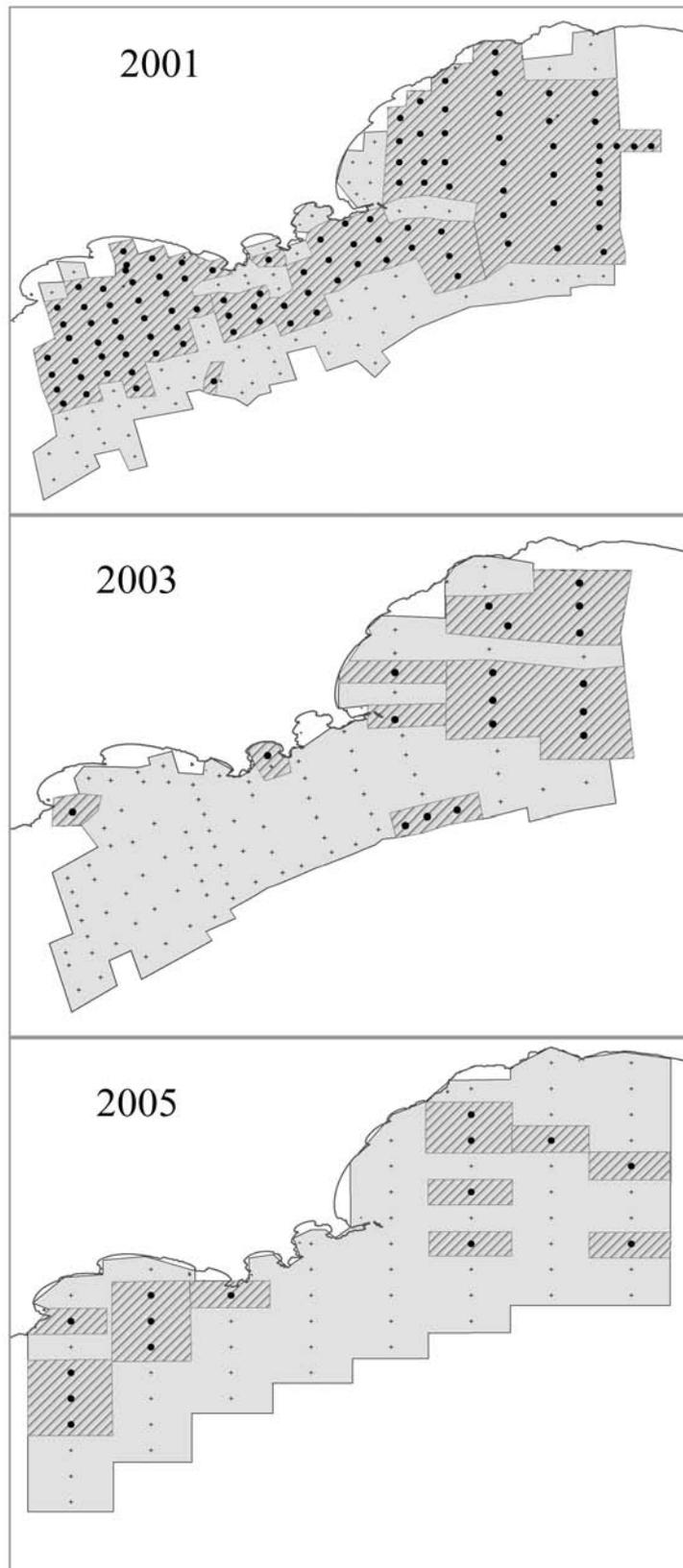


Figure 6b. Plankton survey area (grey) and pilchard spawning areas (hatched) for DEPM surveys at the Bremer Bay region of south-western Australia in 2001, 2003 and 2005. The larger dots depict sampling stations at which Day 0 pilchard eggs were found.

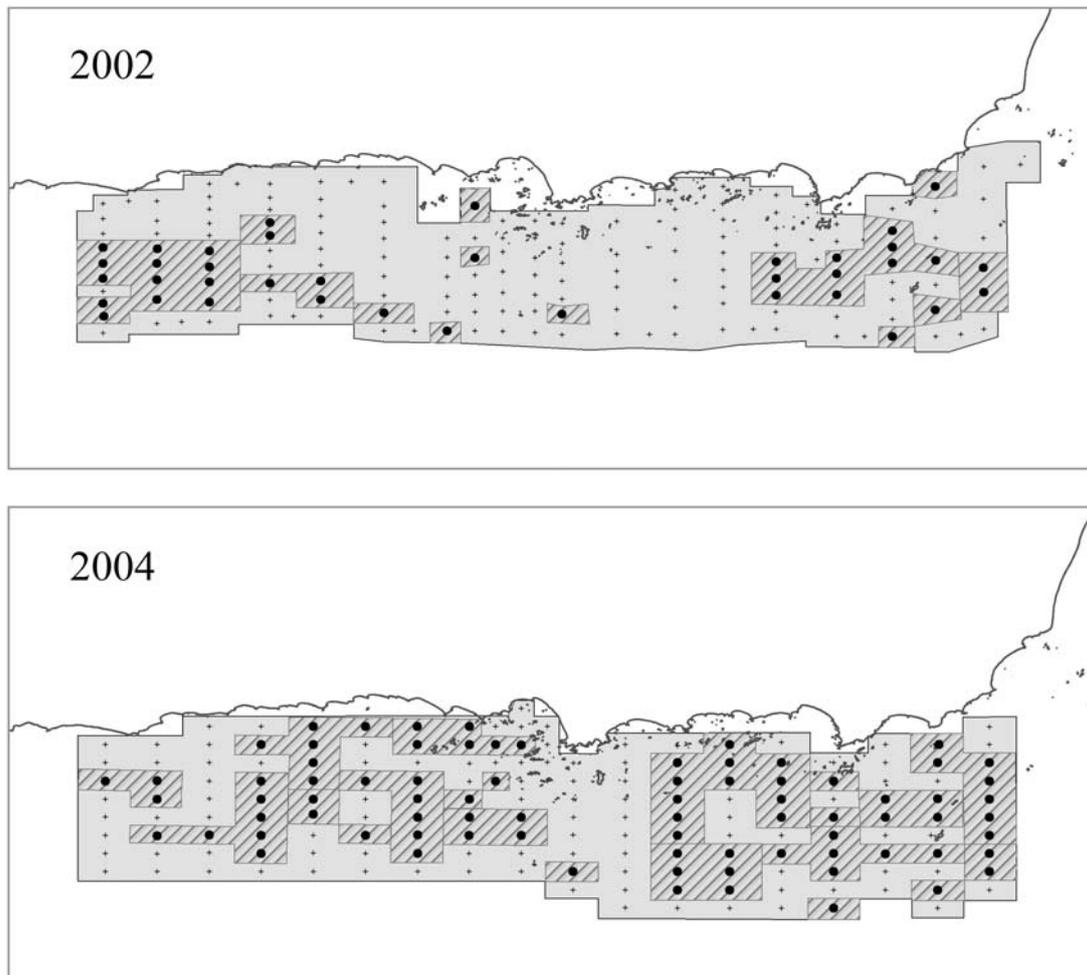


Figure 6c. Plankton survey area (grey) and pilchard spawning areas (hatched) for DEPM surveys at the Esperance region of south-western Australia in 2002 and 2004. The larger dots depict sampling stations at which Day 0 pilchard eggs were found.

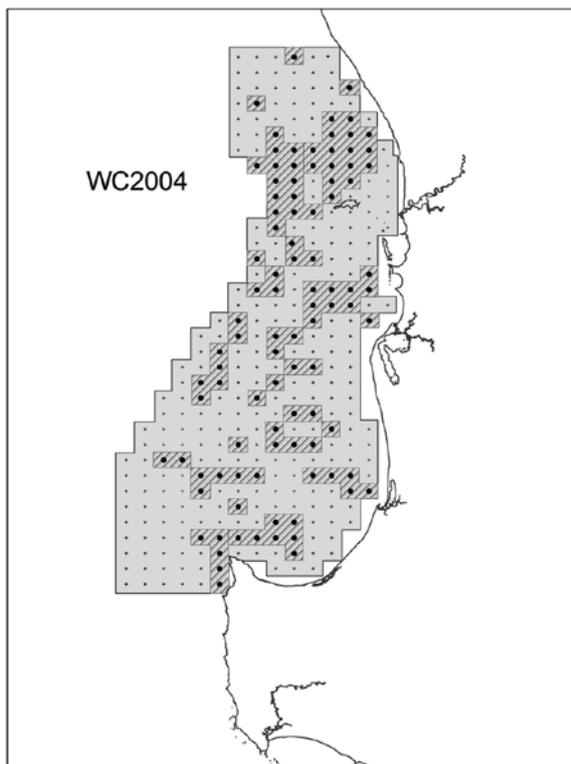
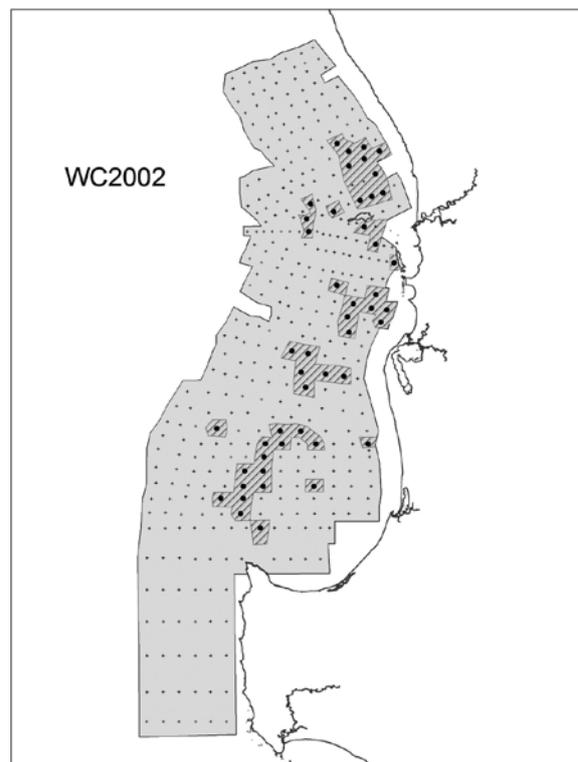
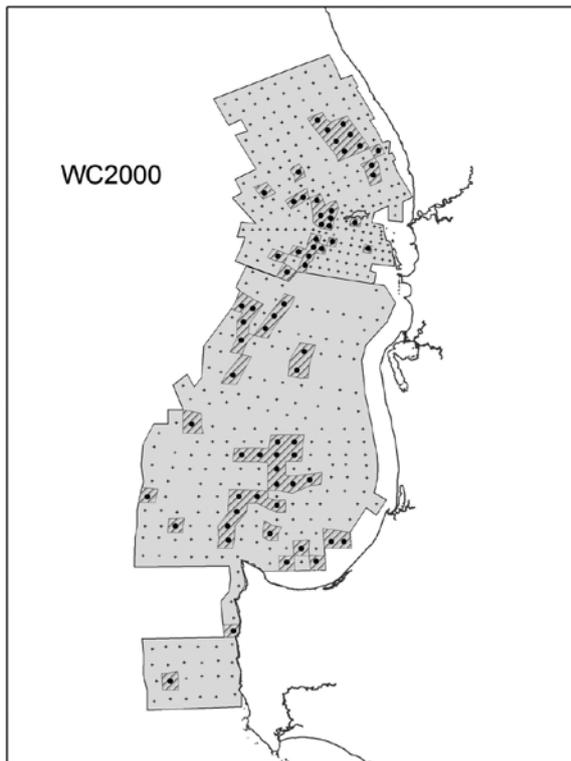


Figure 6d. Plankton survey area (grey) and pilchard spawning areas (hatched) for DEPM surveys at the west coast region of south-western Australia in 2000, 2002 and 2004. The larger dots depict sampling stations at which Day 0 pilchard eggs were found.

Table 2. Summary of parameter estimates and spawning biomass estimates for each of the DEPM surveys. (CV, coefficients of variation are shown in parentheses). Estimates of spawning biomass for Bremer Bay in 2003 and 2005 could not be determined using data only from that region. Available data were therefore combined with that for Albany. While the estimated spawning biomass for both regions in 2005 is less than that for Albany alone, and thus appears to be counterintuitive, this result reflects the imprecision of the DEPM and in particular the difficulty in estimating egg production. Note that the spawning area estimates provided here include some that have been weighted up as part of the process to generate meaningful estimates of P_0 (See Section 2.1.5) so will not always match the areas depicted in figures elsewhere in this report.

Survey	No. of plankton samples	Egg production (Eggs/0.05 m ²)	Spawning area (km ²)	No. of adult samples	Mean weight (g)	Spawning fraction	Fecundity (eggs / average weight female)	Sex ratio	Spawning biomass (tonnes)
West Coast 2000	458	4.96 (0.22)	3,804	9	53.7 (0.008)	0.20 (0.11)	16,039 (0.05)	0.73 (0.19)	9,669 (0.21)
West Coast 2002	394	2.43 (0.52)	2,997	9	44.3 (0.05)	0.065 (0.145)	12,515 (0.04)	0.67 (0.09)	12,819 (0.59)
West Coast 2004	328	4.33 (0.04)	5,366	8	63.0 (0.03)	0.080 (0.035)	17,906 (0.05)	0.64 (0.03)	31,828 (0.20)
Albany 2001	241	5.50 (0.16)	1,716	15	45.1 (0.04)	0.21 (0.01)	14,674 (0.04)	0.57 (0.09)	4,822 (0.19)
Albany 2003	217	5.80 (0.17)	1,360	13	28.8 (0.15)	0.059 (0.016)	6,540 (0.16)	0.53 (0.04)	23,145 (0.31)
Albany 2005	85	7.88 (0.19)	2,399	21	22.1 (0.05)	0.045 (0.012)	6,267 (0.07)	0.48 (0.10)	63,452 (0.30)
Bremer Bay 2001	172	8.97 (0.16)	4,549	12	45.6 (0.08)	0.12 (0.051)	12,392 (0.04)	0.86 (0.10)	37,603 (0.23)
Bremer Bay ¹ and Albany 2003	275	4.90 (0.14)	2,797	13	28.82 (0.15)	0.059 (0.016)	6,540 (0.16)	0.53 (0.04)	40,751 (0.34)
Bremer Bay ¹ and Albany 2005	151	4.04 (0.12)	4,076	26	24.4 (0.06)	0.040 (0.010)	6,684 (0.07)	0.71 (0.24)	47,516 (0.12)
Esperance 2002	167	4.35 (0.70)	5,868	9	29.2 (0.12)	0.044 (0.076)	12,444 (0.05)	0.76 (0.04)	41,996 (0.70)
Esperance 2004	181	3.74 (0.24)	11,241	6	31.2 (0.04)	0.048 (0.052)	7,516 (0.06)	0.69 (0.65)	113,044 (0.21)

1. Separate estimates could not be made for Bremer Bay so combined estimates including the Albany region are provided for 2003 and 2005.

The lack of adult samples in the 2003 survey of Bremer Bay necessitated that the pilchard spawning biomass be estimated in combination with data from Albany, under the assumption that the samples of adult pilchards from Albany were representative of those from Bremer Bay. Similarly, despite four samples of adults being obtained from the Bremer Bay region in 2005, no post-ovulatory follicles were found in the histological sections of 166 ovaries and hence an estimate of spawning fraction could not be made. Furthermore, egg production could not be estimated for the Bremer Bay in 2005 because the egg data collected resulted in a positive mortality rate. We therefore again estimated a combined spawning biomass for Albany and Bremer Bay. The combined estimates for 2003 and 2005 are compared against the summed DEPM derived estimates of spawning biomass from previous years in Figure 7, noting that only those years for which surveys were undertaken in both Albany and Bremer Bay are shown. Combining data from Albany and Bremer Bay to overcome data problems in the latter is not an ideal solution, and highlights the problem of using DEPM to estimate spawning biomass. Thus, the 2005 survey of Albany and Bremer Bay resulted in an estimate for Albany alone of 63,452 tonnes, but when the data for Albany and Bremer Bay were combined the estimated spawning biomass was only 47,516 tonnes (Table 2). It is important to recognize that neither of these estimates is better than the other and their use could result in very different management outcomes if DEPM-based estimates of spawning biomass were the sole source of stock assessment advice. Nonetheless, the time-series does suggest that pilchard spawning biomass at Albany/Bremer Bay increased significantly between 1999 and 2001. The trend for a significant increase in spawning biomass in 2001 and then similar levels in 2003 and 2005 is reflected by the spawning area during the DEPM surveys in each of these years (Figure 6c). These spawning area estimates are considerably higher than found in 1999, several months after the mass mortality, and provide evidence that the abundance of pilchards in the Bremer Bay and Albany regions had increased substantially.

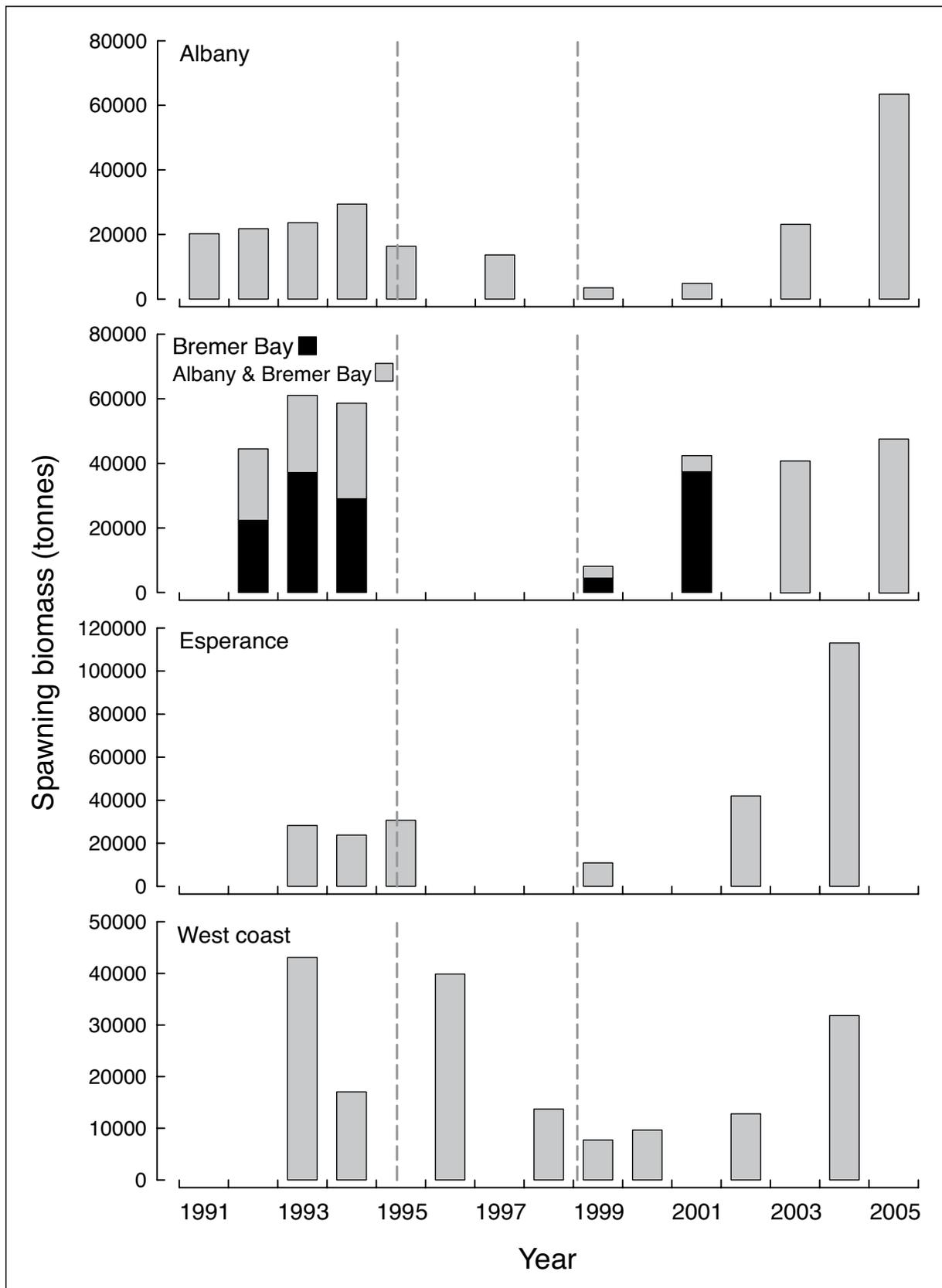


Figure 7. DEPM estimates of spawning biomass of pilchards for Albany, Bremer Bay, Esperance and west coast regions between 1991 to 2005. Results are only shown for those years in which surveys were conducted in each region. The two PHV epidemics occurred in May 1995 and January 1999 (dashed lines).

3.1.2 Integrated fishery model

3.1.2.1 Biomass estimates

The estimates of spawning biomass from the DEPM were used in the integrated model for each south coast zone (Figure 8). In each zone there has been a substantial increase in spawning biomass since 1999. The estimated spawning biomass (and 95% confidence intervals, CIs) for each south coast region in 2005 were:

Albany: 27,061 tonnes (5973 – 48,149)

Bremer Bay: 13,390 tonnes (0 – 42,966)

Esperance: 60,834 tonnes (3778 – 117,890)

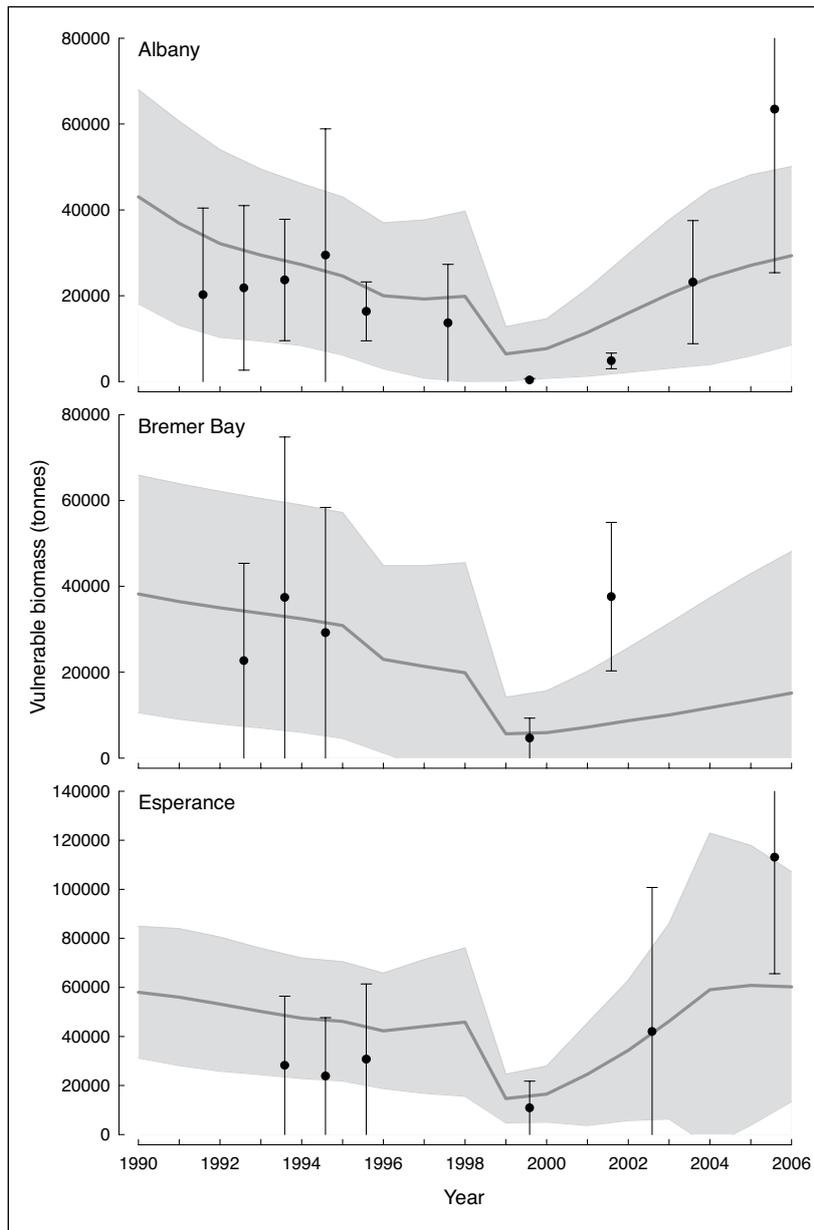


Figure 8. Spawning (vulnerable) biomass for *Sardinopas sagax* from 1990 to 2006 from the Albany, Bremer Bay and Esperance regions of southern Western Australia, estimated using an age structured model that integrates DEPM-based estimates of spawning biomass (dots with ± 2 s.e.). The greyed area represents the 95% confidence interval.

The estimates of spawning biomass provided by the integrated model indicate that the pilchard stock in each purse seine management region of southern WA has increased since the 1998/99 epidemic, but the magnitude of the 95% CIs in any one year also indicates that there remains considerable uncertainty in the actual sizes of the stocks.

The pilchard model outputs do not always appear to reflect the DEPM-based estimates. DEPM-based estimates of spawning biomass far exceeded those from the model at Bremer Bay in 2001, Esperance in 2004 and Albany in 2005. In simple terms, this results from the integrated model's inability to fully reconcile the age-data collected from the fishery against the DEPM-based estimates of spawning biomass. Thus, some high DEPM-based estimates of spawning biomass exceeded what would be possible given the fishery-dependent data (i.e. age composition and recruitment strength, fishing mortality) and known biological capabilities of pilchards (i.e. capacity for population growth through reproduction and growth of individuals). The DEPM-based estimates can also lie well below the model-based estimates, as occurred in Esperance in the early 1990s.

The large discrepancies that sometimes occurred in the integrated model used in this study confirm that sampling is not always representative of the actual population, whether this be for adults or eggs. Due to the inherent problems with collecting representative fisheries data, some level of discord in reconciling fishery- and fishery-independent data is to be expected in integrated models. Many complex models, including age-structured models, fail as fisheries management tools because of the quality of the available data, or because the models require a certain (i.e. considerable) amount of contrast in the data (Kelly and Codling, 2006). Although the pilchard integrated models have successfully tracked the relative trends in spawning biomass, the levels of uncertainty with these estimates impacts on the use of the models for providing definitive or "stand-alone" scientific advice.

In addition to the statistical aspects of the model (that attempt to reconcile the different data), violation of the model's underlying assumptions can also impact on the efficacy of the outputs. For example, age-structured models typically assume, i.e. implement, a predictable stock – recruitment relationship but the form of this relationship must be imposed in most cases without evidence of the form of the real relationship (e.g. Michielsens and McAllister, 2004). Furthermore, age-structured models often fail to account for recruitment levels that respond to fluctuations in environmental conditions rather than to variations in stock size, due in part to the difficulty of establishing cause and effect relationships (e.g. Patterson et al., 2001). The lack of definitive evidence for stock-recruitment relationships has been overcome in some cases by formally incorporating expert opinion to provide intuitively realistic bounds or conditions, via Bayesian techniques, on outputs from calculations performed within models (e.g. Michielsens and McAllister, 2004).

Some aspects of the pilchard models' behaviour are further described below to explain why ancillary biological and ecological information must be considered alongside the biomass estimates when undertaking management of the WA purse seine fisheries.

3.1.2.2 Model stability

The ongoing development of the pilchard integrated model, which was first developed in 1998 (Hall, 2000), has resulted in fits of various reliability stemming from problems with inconsistency of data inputs. The challenges posed through addressing these problems lead to an improved understanding of the uncertainties in both the model and the pilchard stocks. However, subsequent to the relative stability of the model when first completed, it has

since behaved irregularly in response to the annually updated fishery-dependent data collected since that time. The addition of “new” data to a fishery model results in a refit of the complete model, so when new data are added each year the biomass trajectories change in response to the updated fit. This is particularly the case if the model finds the new data difficult to fit, resulting in updated outputs that can be quite different to those obtained previously. Potential causes of instability for the south coast pilchard models are described below.

The changes in pilchard spawning biomass at Albany and Bremer Bay between 1999 and 2001, and the potential mechanisms for this increase, have been described by Gaughan (2003). These are summarized here to indicate how inconsistency in spatial/behavioural dynamics may be contributing to instability in the integrated models.

Following the mass mortality event of 1998/99, pilchards were difficult to find off Albany and Bremer Bay. Research-fishing for pilchards in the Albany and Bremer Bay regions was undertaken between November 2000 and February 2001 using experienced purse seine fishers (there was no commercial fishing at this time); while the presence of several schools of pilchards in Bremer Bay were found during this period, none were found off Albany. Subsequently, the DEPM survey off Albany and Bremer Bay found a widespread distribution of spawning adults several months later in July of 2001 (Figure 9). Samples of these fish consisted predominantly of two to five year olds. Given the very low levels of residual stock in Albany and Bremer Bay after the 1998/99 mass mortality and the sudden appearance of several cohorts of mature pilchards, it is possible that these fish were entering the Albany region as migrants rather than being local recruits.

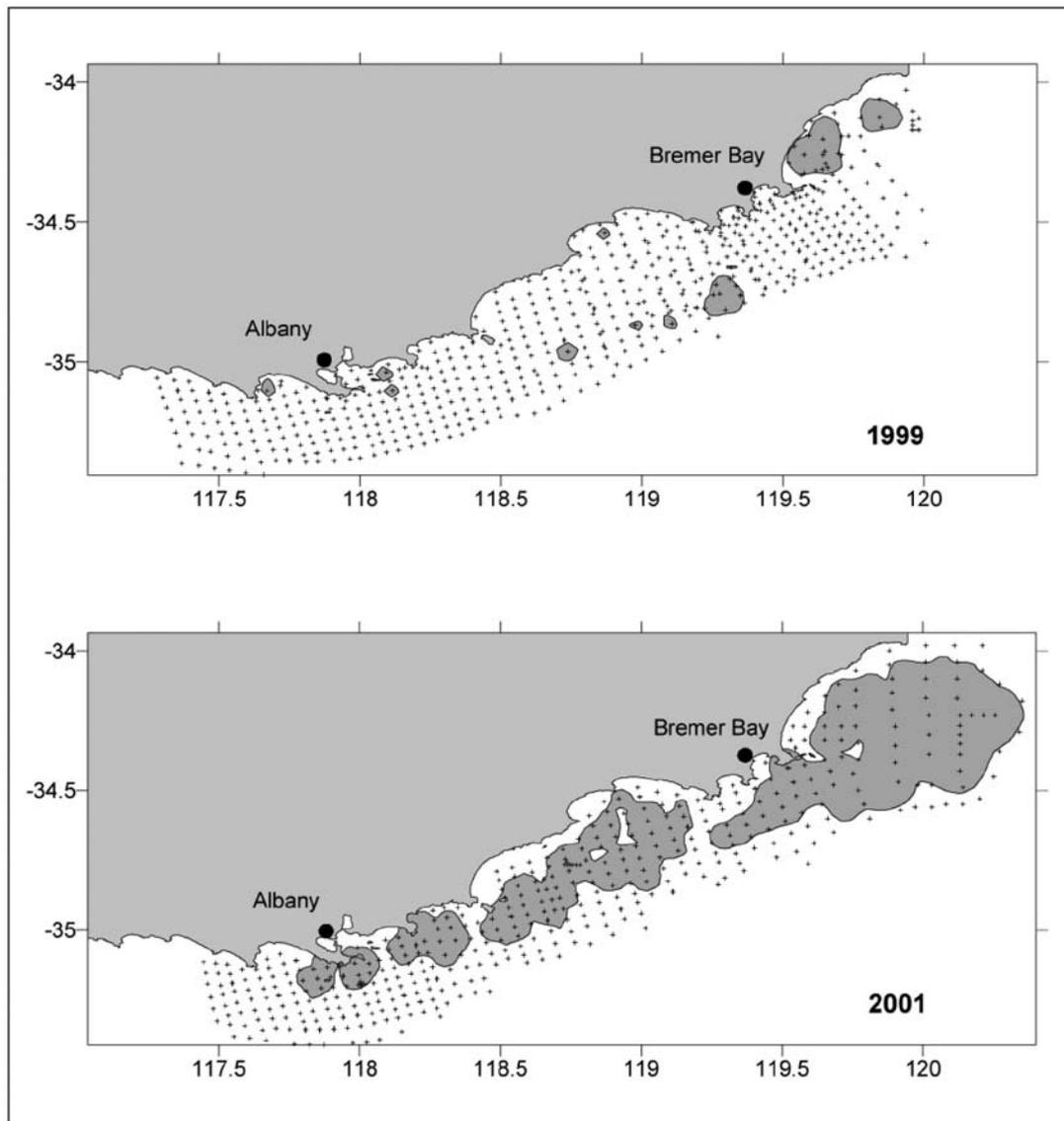


Figure 9. Spawning area of *Sardinops sagax* at Albany and Bremer Bay as determined by the presence of eggs < 24 hours old collected using plankton nets. Crosses denote plankton sample stations. (a) June – July 1999; (b) June – July 2001 (from Gaughan, 2003).

The migration of large quantities of mature pilchards into the eastern management zones is contrary to the spatial population dynamics during the 1990s when large-scale migration of adults along the coast was not evident (Edmonds and Fletcher, 1997; Gaughan et al., 2002). Furthermore, the increase in pilchard biomass in the Albany and Bremer Bay regions in 2001 may not have been due to a concomitant increase in the overall size of the south coast pilchard population but rather from a redistribution of the stock that had been located to the east of Bremer Bay. Some of the inconsistencies in model outputs over time may be attributable to the appearance of four- and five-year old pilchards in regions for which these cohorts were not well represented in the previous years.

Irregular or non-linear change in spatial dynamics (Bakun, 2001), as opposed to consistent/predictable (e.g. seasonal) variability, is difficult to identify in the short to medium term and typically not accounted for in age-structured models. Indeed, the possibility of irregular changes in spatial dynamics has not been explicitly recognized for many exploited fish stocks.

In the case of pilchards off southern WA, the large-scale eastward migration may have been a density-dependent response to a severely depleted stock, hence may no longer occur, i.e. may only be of benefit when the overall stock is reduced in size and patchily distributed. Alternatively, a significant change in patterns of behaviour, with associated influences on age-composition of pilchards in any one region, may persist despite that stock size in each region is now at healthy levels. Such changes have been documented for small pelagic fish elsewhere and have been attributed to recruiting year classes needing to learn their spatial behaviour (e.g. seasonal migration) from older fish - “normal” spatial behaviour of recruits can change following a significant impact on the older age classes within a stock (Corten, 2002).

Further evidence for changes in behaviour of pilchards is provided by the ongoing scarcity of mature aged fish in the traditional inshore fishing grounds, despite the DEPM surveys indicating the presence of spawning pilchards (i.e. adults) in the offshore waters of each fishing zone. Whereas, schools of large, adult pilchards once resided for many months each year within the inshore waters near Albany, Bremer Bay and Esperance, this is no longer the case. Anecdotal information from fishers and evidence from their catch histories indicate that influxes of adult pilchards to the inshore fishing grounds have been sporadic and of a lower magnitude than during the 1980s and 1990s; because of this, many fishers consider that the fishery has not recovered. The availability of predominantly juvenile/young pilchards, combined with the fishers’ economic need to catch small fish that would have traditionally been avoided, confounds the robustness of the annual age composition data. These linked factors have resulted in large proportions of young (2 – 3 year old) pilchards in the catches, but without the flow-through of these cohorts, i.e. as progressively older fish, over subsequent years (D. Gaughan, unpublished data). This contrasts the strong cohort signals evident in the pilchard fisheries during the 1990s (Gaughan et al., 2002) and which apparently contributed to the initial stability of the pilchard model developed by Hall (2000).

3.1.2.3 Future fisheries management

The need to invoke arbitrary assumptions on fishery models to make them tractable and the fact that models typically cannot capture the true complexities operating within a fishery means that models must always be viewed with scepticism (Schnute and Richards, 2001). The conclusion by these authors that “*models must serve as tools for thought rather than rigid prescriptions*” is supported by the experience with the pilchard simulation model used in WA. This equally applies to the DEPM-based estimates when considered independently of the simulation model. The concurrent synthesis and reconciliation of different data sources in the integrated model provides a more transparent basis from which to consider the future of the purse seine fisheries off southern WA. However, in any one year neither DEPM-based estimates of spawning biomass nor the outputs from integrated model can be known with sufficient certainty to be considered as the sole source of scientific advice.

The considerable quantities of data and resources that have already been applied to researching pilchards in WA, including the development of simulation models, now make it clear that there will likely always be considerable uncertainty in the biomass estimates. Therefore, rather than relying each year on point estimates against which an agreed exploitation rate can be applied, a holistic view of the biomass trends over the past ~15 years have been used to provide a longer-term prognosis of the likely stock size of pilchards in each management region. The development of this prognosis consisted of examining the complete series of biomass estimates for each region to determine the typical range that the biomass fell within, and follows Gaughan et al. (2004) in which the upper end of this range was similarly defined. Although the

selection of the range is not based on a formal assessment, it is nonetheless pragmatic and when previously presented to industry members was seen to have merit. The extended time-series of biomass estimates presented in this report permit the following update on the likely pilchard biomass in each region, which will be discussed with fisheries managers, including the purse seine industry and conservation stakeholders.

The likely pilchard biomass in each region if average to strong recruitment has been experienced over a number of years and environmental conditions favoured pilchards as the dominant baitfish species are:

West Coast	20,000 – 30,000 tonnes
Albany	15,000 – 30,000 tonnes
Bremer Bay	15,000 – 37,000 tonnes
Esperance	35,000 – 62,000 tonnes
Total	85,000 – 159,000 tonnes

3.2 Acoustic and visual detection of pilchard schools

3.2.1 DEPM surveys

Schools of pilchards were observed only infrequently during the plankton surveys, and on some surveys none were detected (Figure 10a-d), only those surveys for which schools were detected are shown. No consistent trends in the number of location of pilchard schools relative to the presence of pilchard eggs < 24 hours old were discernible. The numbers of pilchard schools seen during the plankton surveys thus provided no data that could be used to indicate the health of the stock in each region.

3.2.2 Specific exploratory survey

The search for pilchard schools in the Esperance zone in early 2006, undertaken in lieu of the DEPM survey, extended ~240 km east of the traditional fishing grounds near Esperance. This survey detected about 20 schools of pilchards (Figure 11) and many smaller traces of baitfish that were likely to have been pilchards. Most schools of pilchards were small, with only two of a size (~3 – 5 tonnes) that would normally be considered a worthwhile target for commercial fishing. Most pilchards observed were exhibiting fine-scale behaviour that is normally described by commercial fishers as “patchy”, which indicates that there are fish in the vicinity but that they are not “schooled-up” sufficiently to present a worthwhile target for a purse seine shot.

The locations of schools recorded during the survey indicate a wide distribution of pilchards throughout the zone, as has previously been indicated by the presence of pilchards eggs in the 2002 and 2004 DEPM surveys (see Figure 6c). However, as has frequently been experienced by Esperance purse seine fishers over recent years, no large schools of pilchards were detected in the vicinity of Esperance.

The gillnets were deployed daily but failed to catch any pilchards. The sizes of pilchards that occurred outside of the traditional fishing grounds around Esperance was therefore not able to be determined. Many of the pilchard schools were distributed too far offshore to constitute targets that would be viable to fish by currently available vessels. Furthermore, there is no infrastructure east of Esperance that would be suitable for landing large quantities of pilchards.

While pilchards could be landed on some beaches, using barges to ferry the catch to shore, the extra handling time combined with the distance that catch would then have to be trucked back to Esperance constrains the viability of this option.

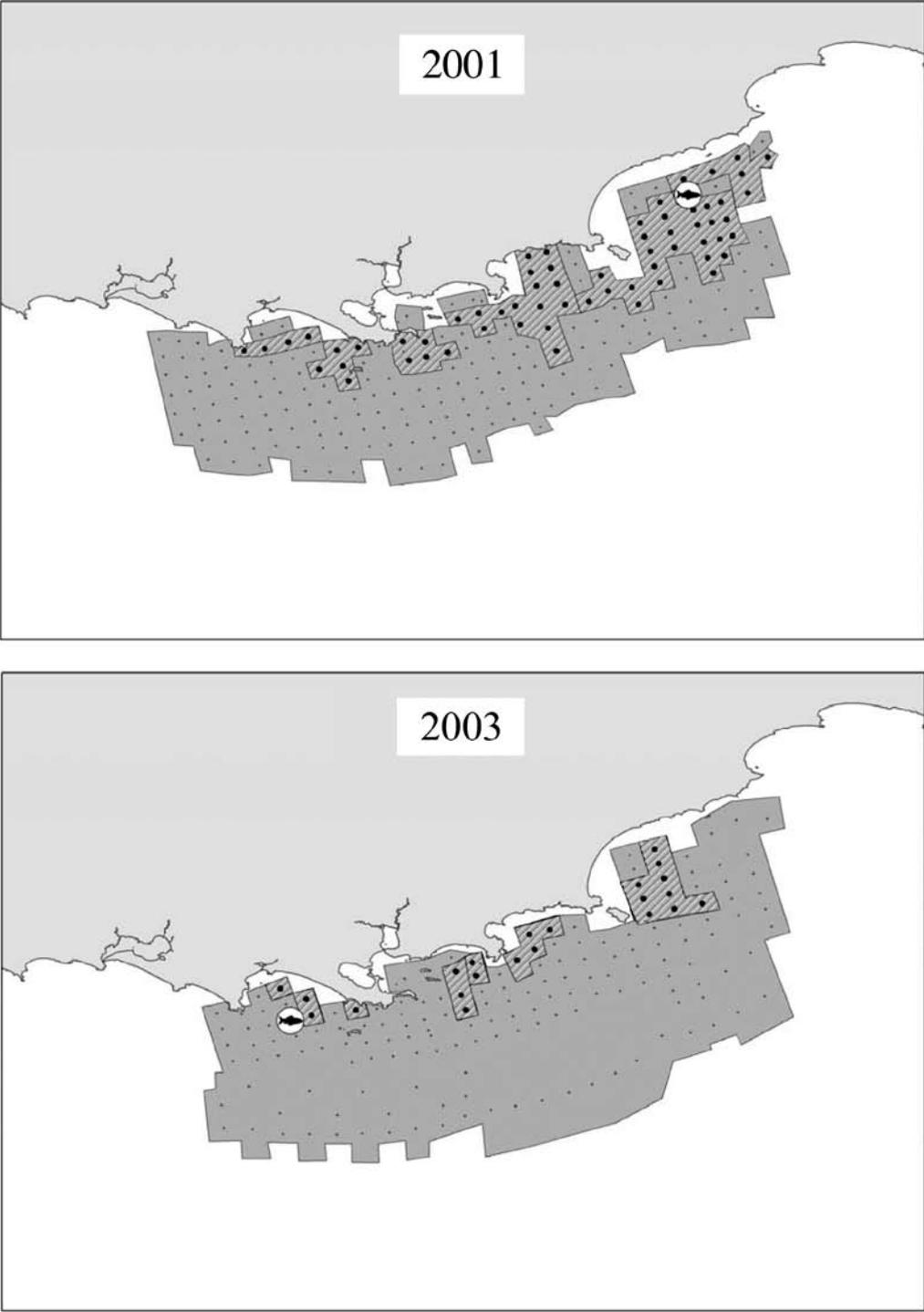


Figure 10a. Locations of pilchard schools (fish symbol) observed during plankton surveys at the Albany region in 2001 and 2003 (none were observed in the 2005 survey). Pilchard spawning area (hatched) represents that estimated for the DEPM calculations.

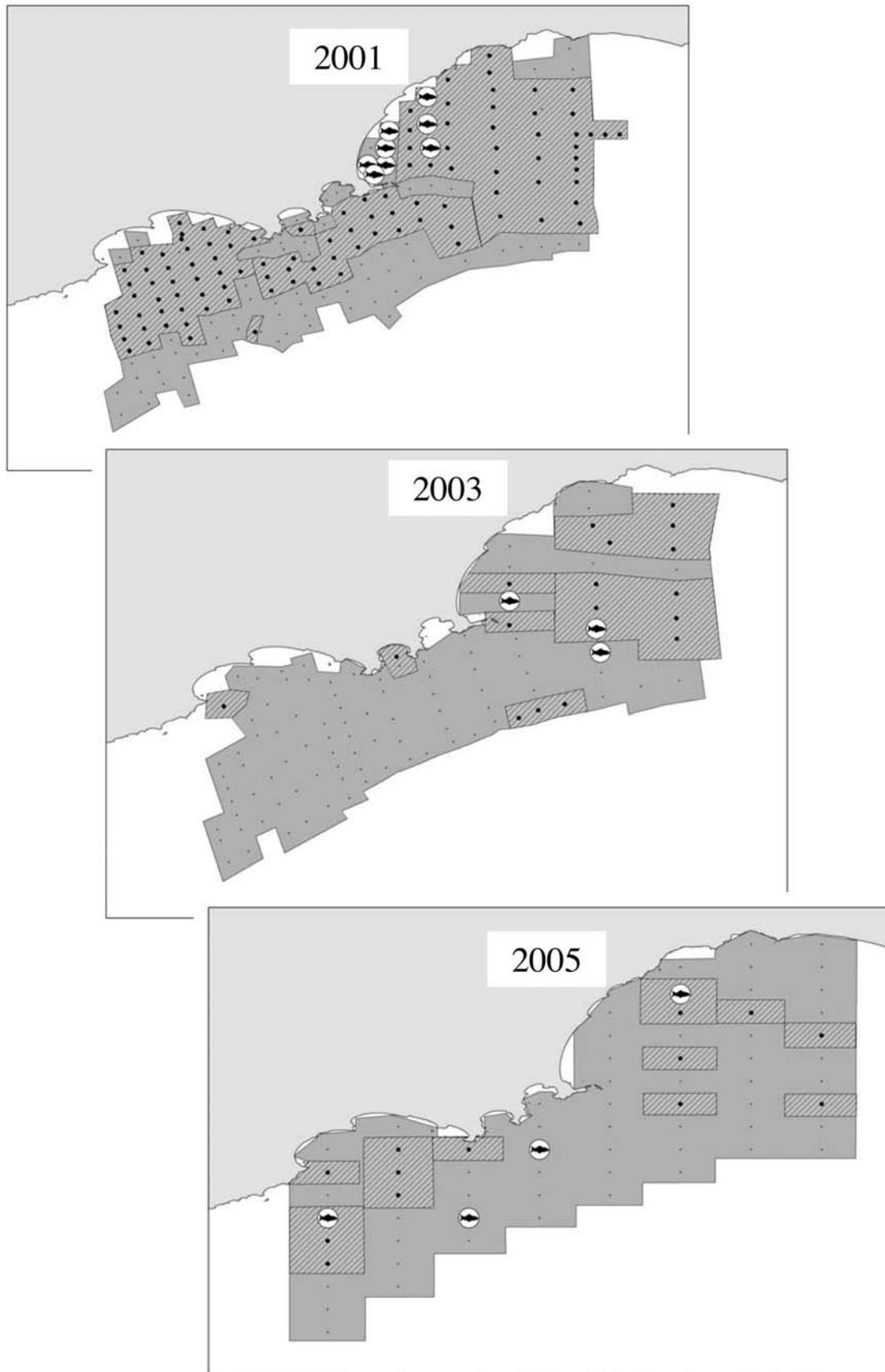


Figure 10b. Locations of pilchard schools (fish symbol) observed during plankton surveys at the Bremer Bay region in 2001, 2003 and 2005. Pilchard spawning area (hatched) represents that estimated for the DEPM calculations.

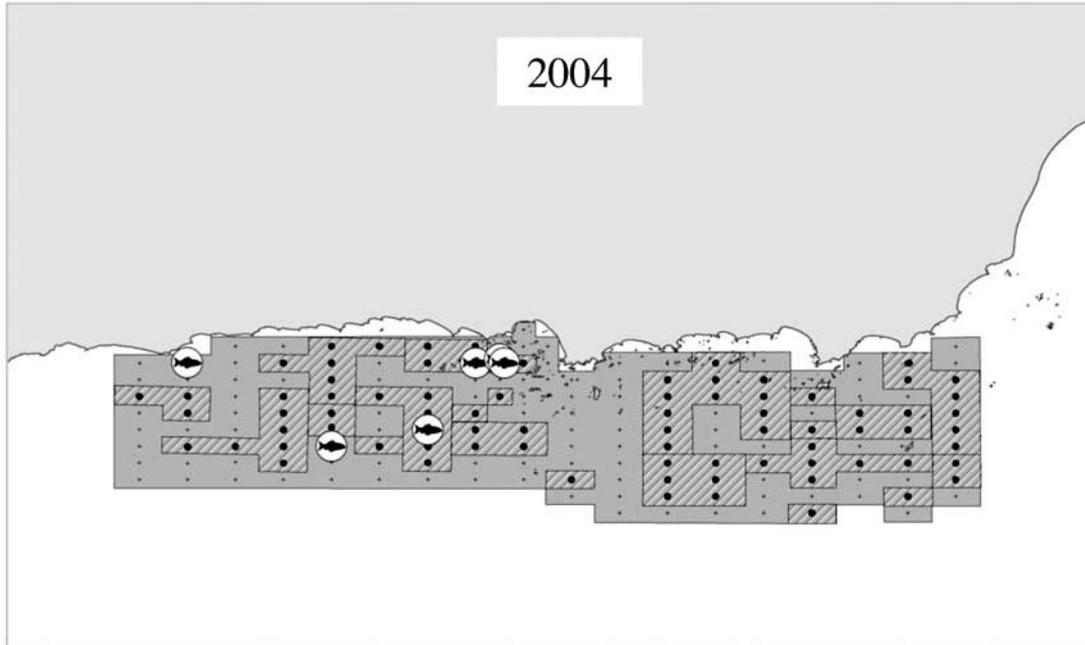


Figure 10c. Locations of pilchard schools (fish symbol) observed during plankton surveys at the Esperance region in 2004 (none were observed in the 2002 survey). Pilchard spawning area (hatched) represents that estimated for the DEPM calculations.

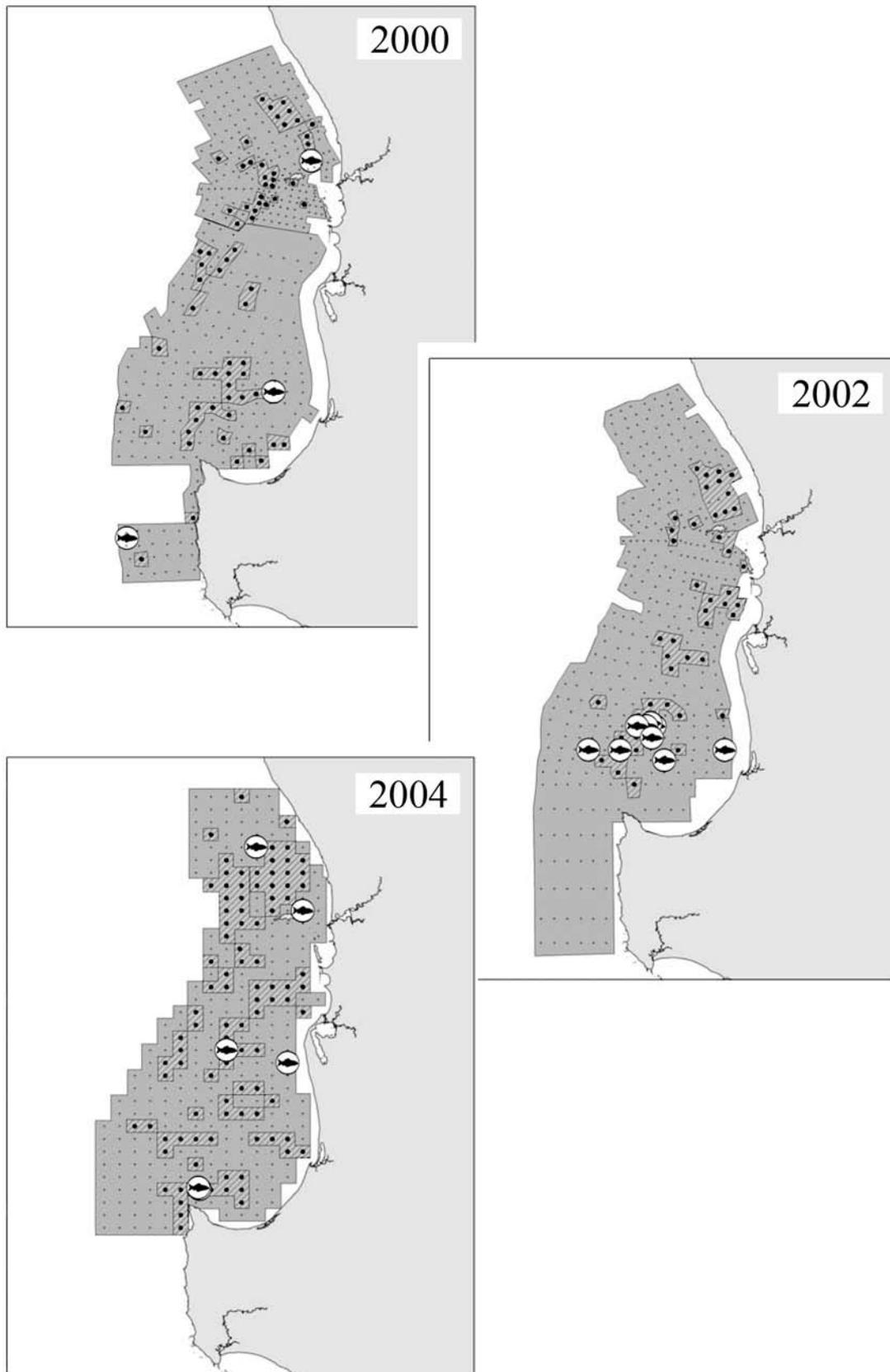


Figure 10d. Locations of pilchard schools (fish symbol) observed during plankton surveys at the West Coast region in 2000, 2002 and 2004 surveys. Pilchard spawning area (hatched) represents that estimated for the DEPM calculations.

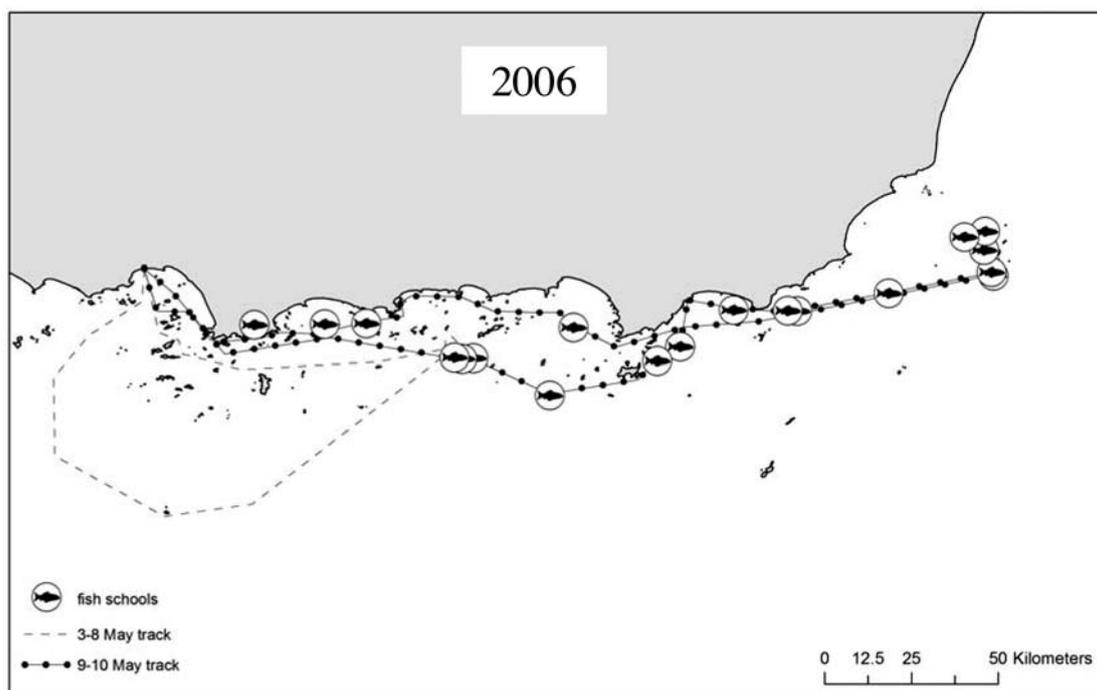


Figure 11. Acoustic sounding survey track and fish schools encountered in the Esperance region May 2006.

3.3 Comparison of growth rates

Objective 3. Determine if there is a measurable change in biological parameters of pilchards important to ongoing development of the simulation models, in particular growth rate, in response to the decrease in stock size.

3.3.1 Annual variability in growth rate

3.3.1.1 Fork length - Otolith weight

The slope of the FL-OW linear relationship, for pilchards with otolith weights restricted to 80–132 mg, is used here as an indicator of growth rate. The slope was interpreted as an average growth rate coefficient in the sense that a higher slope indicates a longer fish for a given otolith weight. The growth rate coefficients based on FL-OW relationships exhibited considerable temporal variability, with variation around the long-term mean of $\pm 10 - 12\%$ for the south coast regions and $\sim \pm 8\%$ for west coast (Figure 12). On the south coast the trends in annual means for the three regions showed broad similarities, with minima in 1994–95 and maxima in 1999. The Albany pilchards experienced a substantial increase in growth from 1995 to 1996 and another from 1998 to 1999, but with a large decrease from 1997 to 1998. Although growth at Albany was highest in 1999, the increase from 1998 (i.e. post epidemic) was of no greater magnitude than had previously been experienced. At Bremer Bay and Esperance the largest change between consecutive years occurred between 1998 and 1999, but as with Albany the magnitude of these changes was not markedly larger than occurred at other times. On the west coast the increase from 1998 to 1999 was relatively small and continued an upward trend that had begun in 1997–98 and continued until 2001.

3.3.1.2 Bodyweight – fork length

The cubic coefficient of the BW-FL regressions is used here as an indicator of body condition; a higher (lower) coefficient indicates heavier (lighter) fish for a given length. Considerable interannual variability in body condition has occurred for pilchards on the west coast of WA and at each south coast region, but no regular patterns were discernible (Figure 13). Interannual variability was particularly evident off the west coast, with marked increases from 1994 to 1995 and again from 1999 to 2000. Pilchards on the west coast were also consistently heavier than those from the south coast. At the other regions the changes were typically more gradual, particularly off Esperance where regular peaks and troughs, with an apparent cycle-period of approximately five years, were evident.

Pilchards from all locations were heaviest soon after the 1998/99 mass mortality and then exhibited rapid declines in weight. This was particularly evident at Albany and Bremer Bay where declined well below the levels observed during the 1990s.

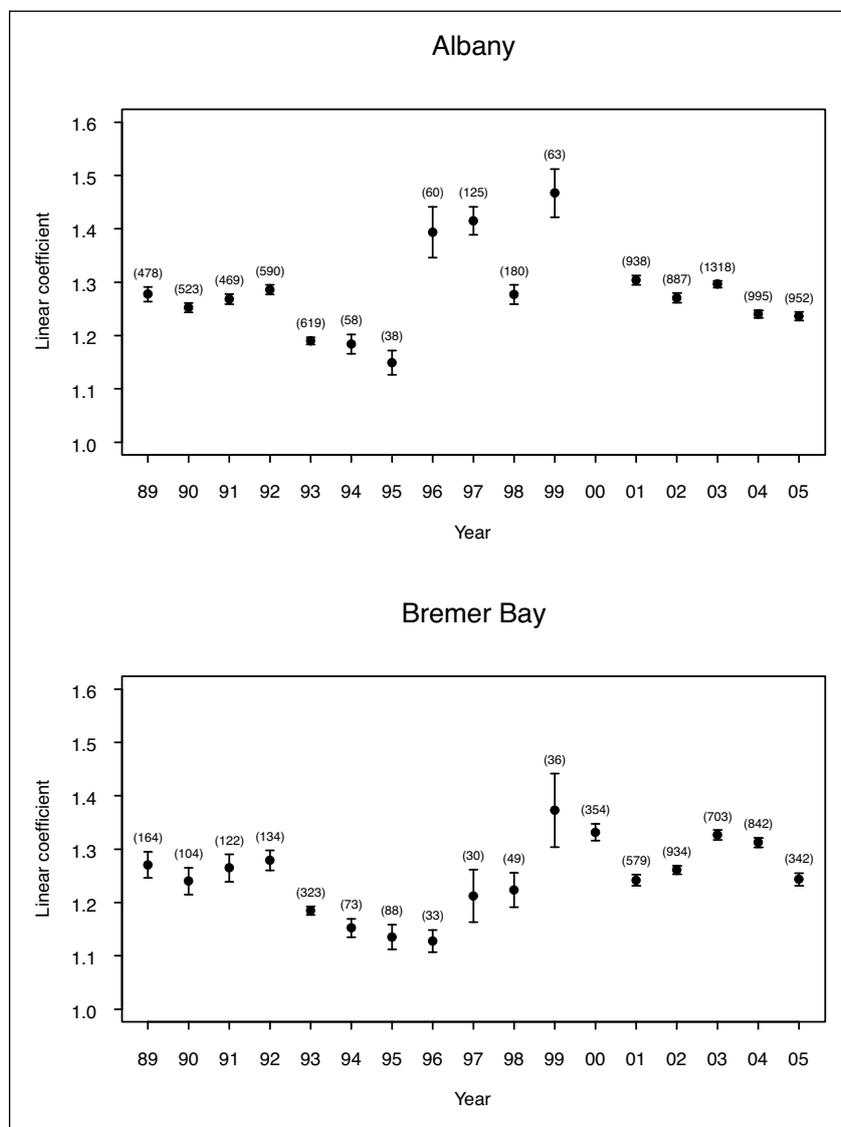


Figure 12. Annual mean (± 2 s.e.) growth coefficients for *Sardinops sagax* at four locations in Western Australia. The growth coefficient was estimated from the slope of the linear fork length (mm) – otolith weight (mg) relationship for the 1+ - 2+ age group. The sample size for each year is shown. Continued over.

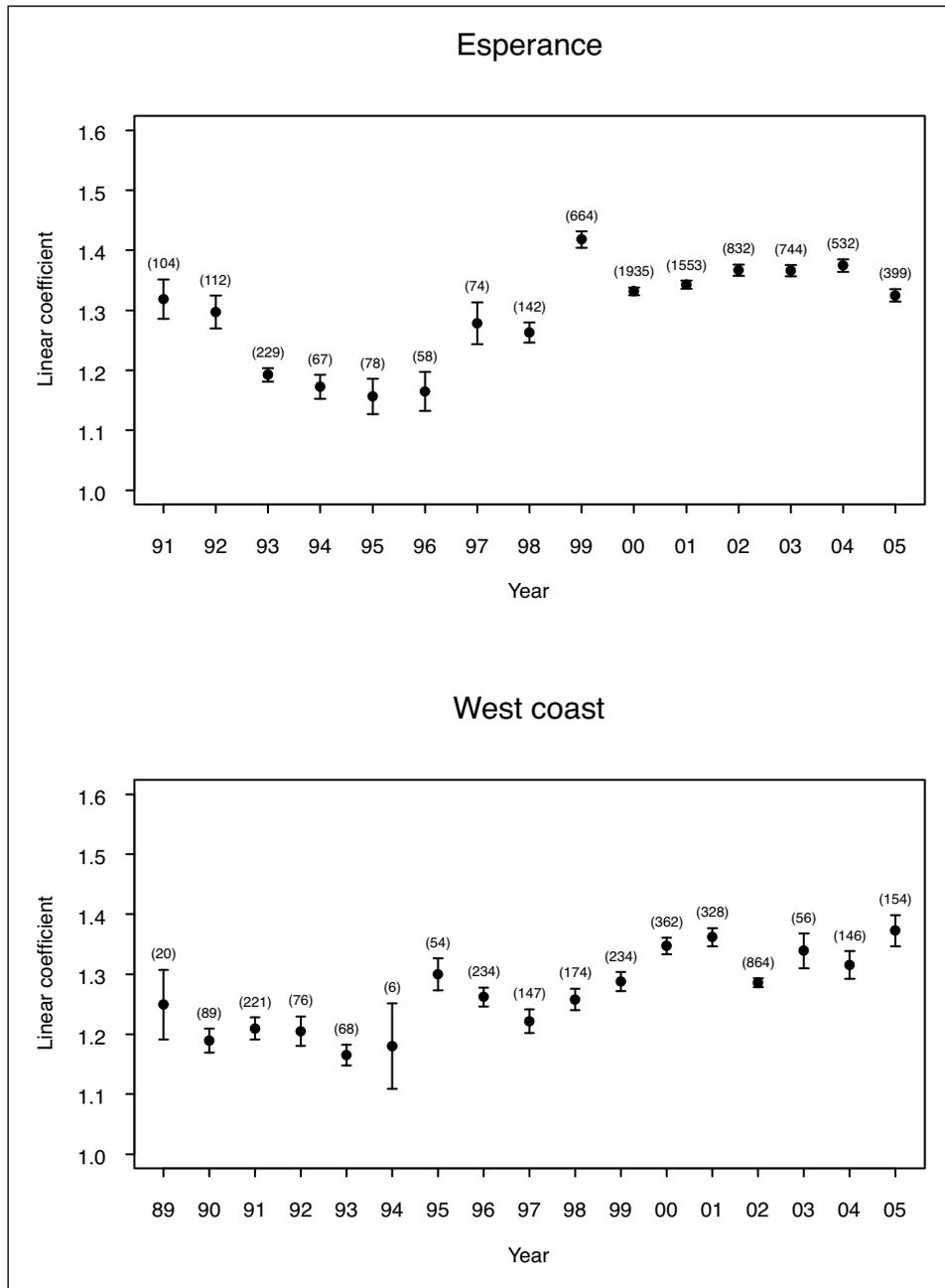


Figure 12 continued. Annual mean (± 2 s.e.) growth coefficients for *Sardinops sagax* at four locations in Western Australia. The growth coefficient was estimated from the slope of the linear fork length (mm) – otolith weight (mg) relationship for the 1+ - 2+ age group. The sample size for each year is shown.

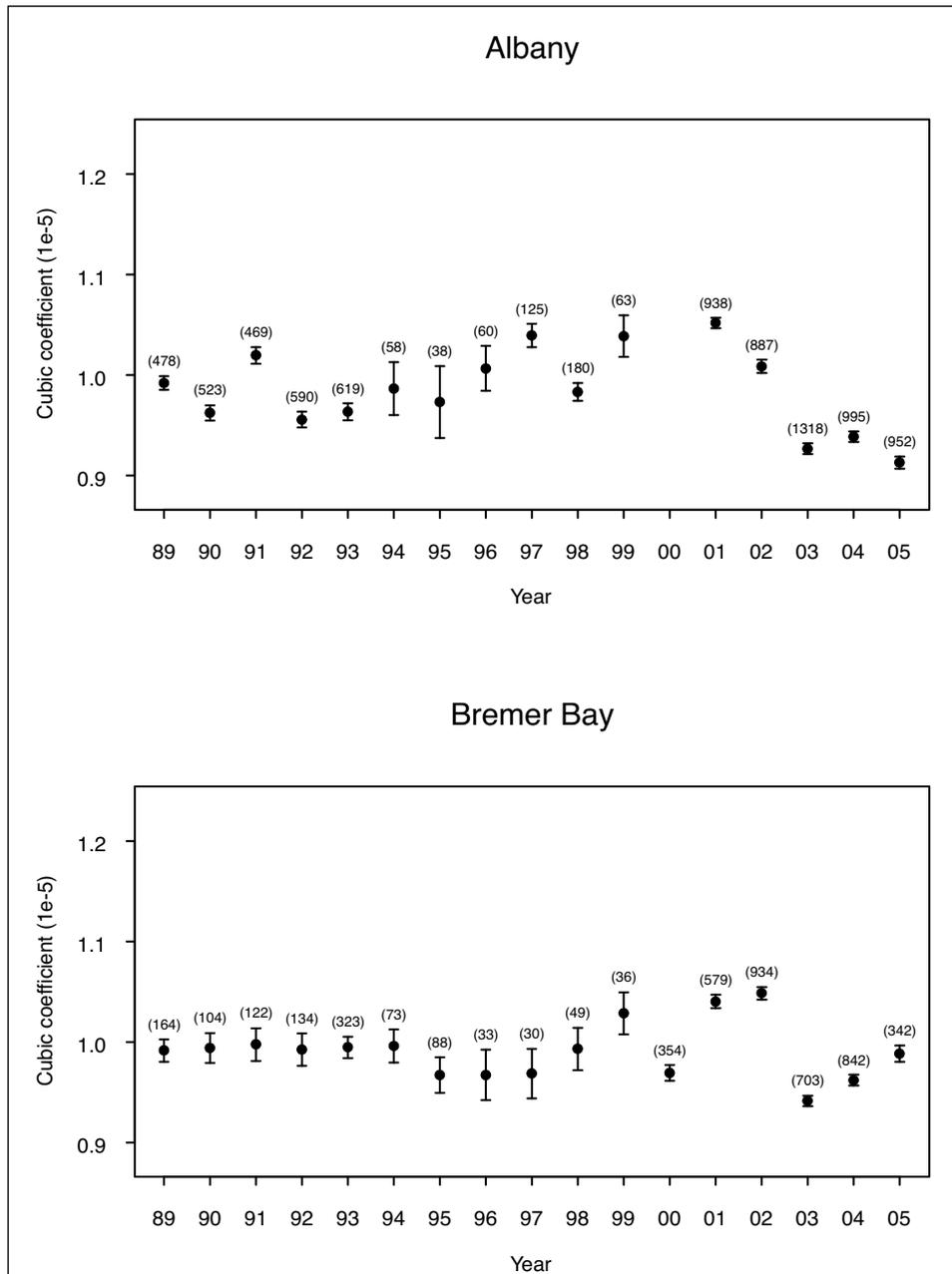


Figure 13. Annual mean (± 2 s.e.) cubic coefficients of the body weight (g) – fork length (mm) relationships for *Sardinops sagax* at four locations in Western Australia. These relationships were fitted using the same 1+ - 2+ age group of fish as in Figure 12. The sample size for each year is shown. Continued over.

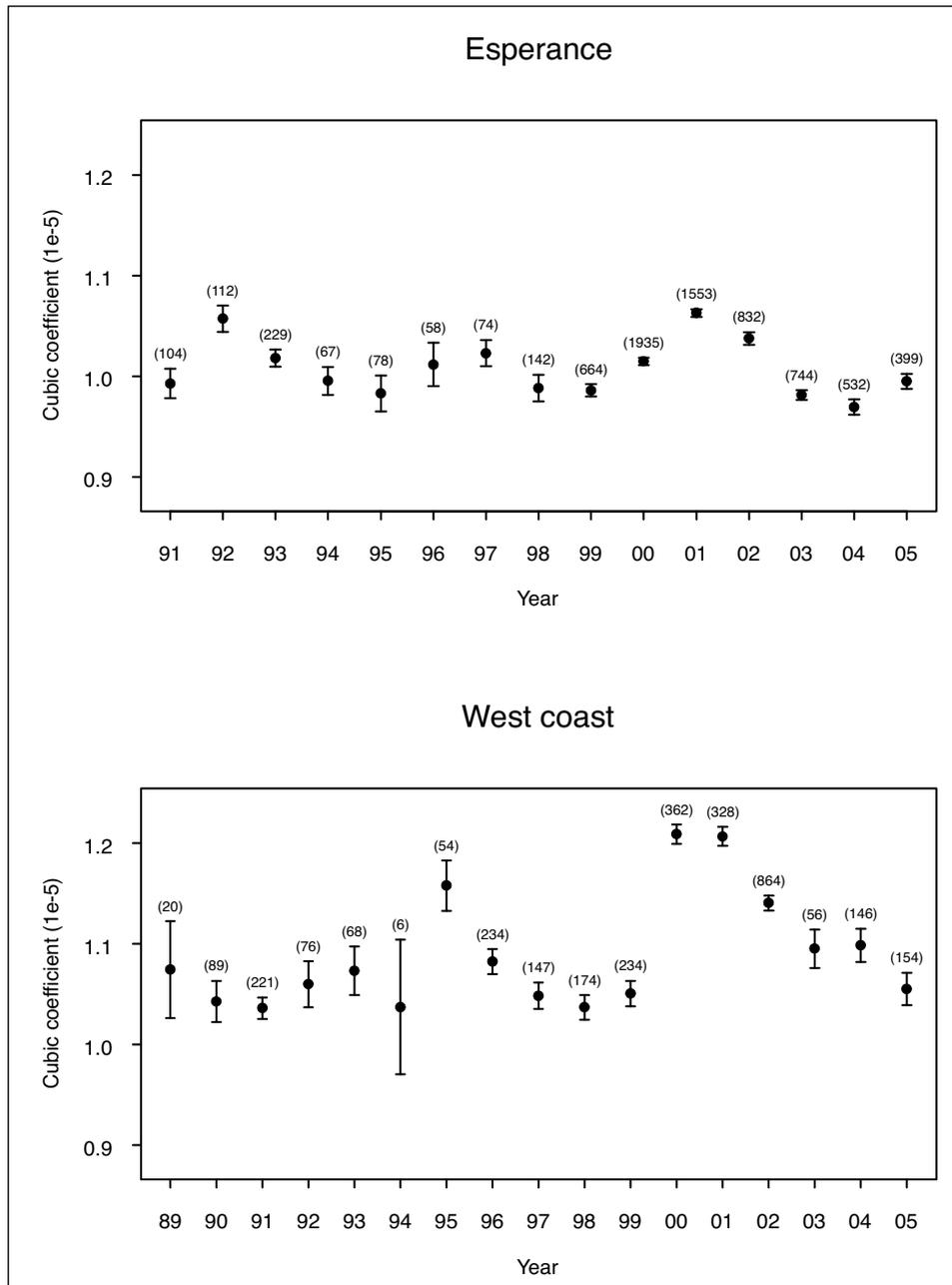


Figure 13 continued. Annual mean (± 2 s.e.) cubic coefficients of the body weight (g) – fork length (mm) relationships for *Sardinops sagax* at four locations in Western Australia. These relationships were fitted using the same 1+ - 2+ age group of fish as in Figure 12. The sample size for each year is shown.

3.3.2 Influence of stock size on growth rate

The annual means of the OW-based growth coefficients and the BW cubic coefficients at each region were regressed against stock size in the contemporaneous year and with a lag of one year. There were insufficient estimates of stock size at Bremer Bay to perform meaningful regressions so only those for the other three regions (Albany, Esperance and West Coast) are presented here.

Significant negative relationships between growth coefficients and biomass in the same year were evident at Albany ($p = 0.02$) and the West Coast ($p = 0.057$) but not at Esperance (Figure

14). A highly significant negative relationship was also found for the growth coefficients of West Coast pilchards and lagged biomass size ($p = 0.003$) but no significant trends were found for Albany and Esperance (Figure 15). Negative relationships were found between the BW coefficient and contemporaneous biomass size for only Albany ($p = 0.014$) (Figure 16) and for both Albany ($p = 0.054$) and the West Coast ($p = 0.032$) for lagged biomass (Figure 17).

The variability in pilchard growth rate in WA, as determined using the two proxy coefficients, provide evidence that pilchard growth rate increased when there had been large declines in population size, as had been suggested for clupeoid populations elsewhere (e.g. Iles, 1973; Murphy 1977). Iles (1973) has previously determined that stock size, amongst other factors, influenced the growth of juvenile pilchards off California. Growth rates of 1+ and 2+ pilchards in the Albany and West Coast regions of WA, a much less productive region than California, are likely to have been influenced by density-dependent effects. Even in those cases where a significant relationship was not found between growth rate and stock size in these two regions, the correlation coefficients were also negative. In contrast, the direction of the non-significant relationships for pilchards in the Esperance region were inconsistent (i.e. both positive and negative) for both the OW- and BW-based indicators. Given the significant relationships at Albany and the West Coast suggest a density-dependent effect, the inconsistencies at Esperance may indicate that (1) the natural dynamics of pilchards in the Esperance region are more variable, (2) the sample size available for fitting the regression was too small ($n = 5$), or (3) biases in sampling pilchards and or imprecision in estimating spawning biomass may be greater than at the other locations. Contradictory or inconclusive evidence for density-dependent effects on growth of herring (*Clupea harengus*) in the Baltic Sea (Rönkkönen et al., 2003) indicates that the results for pilchards off southern WA are not atypical. Furthermore, the ability to ascertain the rates of density-dependent responses and then to predict how these might affect population recovery remains problematic (Botsford, 1981; Post et al., 1999).

The analyses employed in this study have been simplistic, but this avenue was explicitly taken in recognition of both the importance of understanding density-dependent effects on a stock's population dynamics and the difficulties of measuring such effects in wild populations (Rose et al., 2001; Sass and Kitchell, 2006; Goodwin et al., 2006). The relationships for pilchards in WA will be further developed as the time-series of data is extended. In particular, the potential for density dependence of pilchard growth rates at Esperance and Bremer Bay will be further investigated following the sampling and processing of commercial catches in subsequent fishing seasons. Variability in growth rates is likely to be simultaneously influenced by a number of factors, of which intra-specific competition for food is just one (e.g. Rose et al., 2001). More complex multivariate models that consider both biological (e.g. relative recruitment levels) and physical (e.g. temperature) factors will therefore also be investigated.

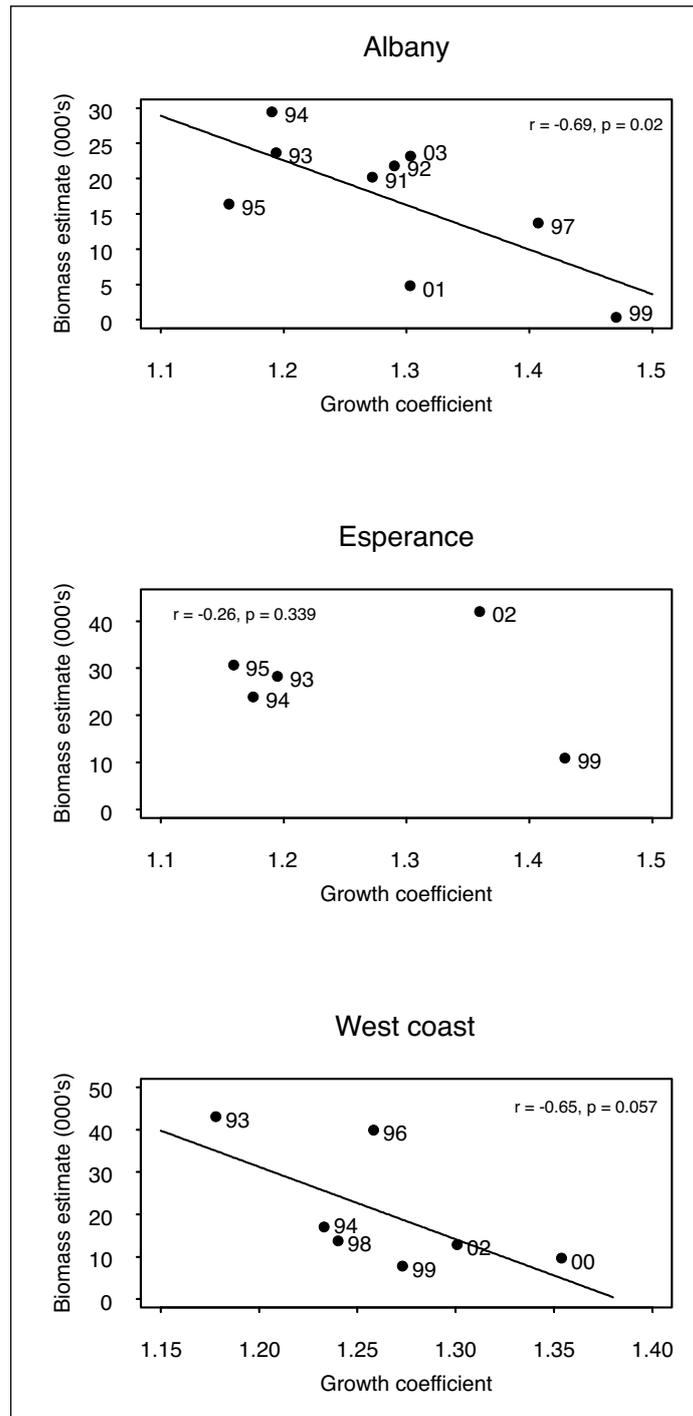


Figure 14. Relationship between mean annual growth coefficients (see text) and spawning biomass in the current year (i.e. no lag) for *Sardinops sagax* at three locations Western Australia. The numbers refer to the years that growth coefficients were estimated.

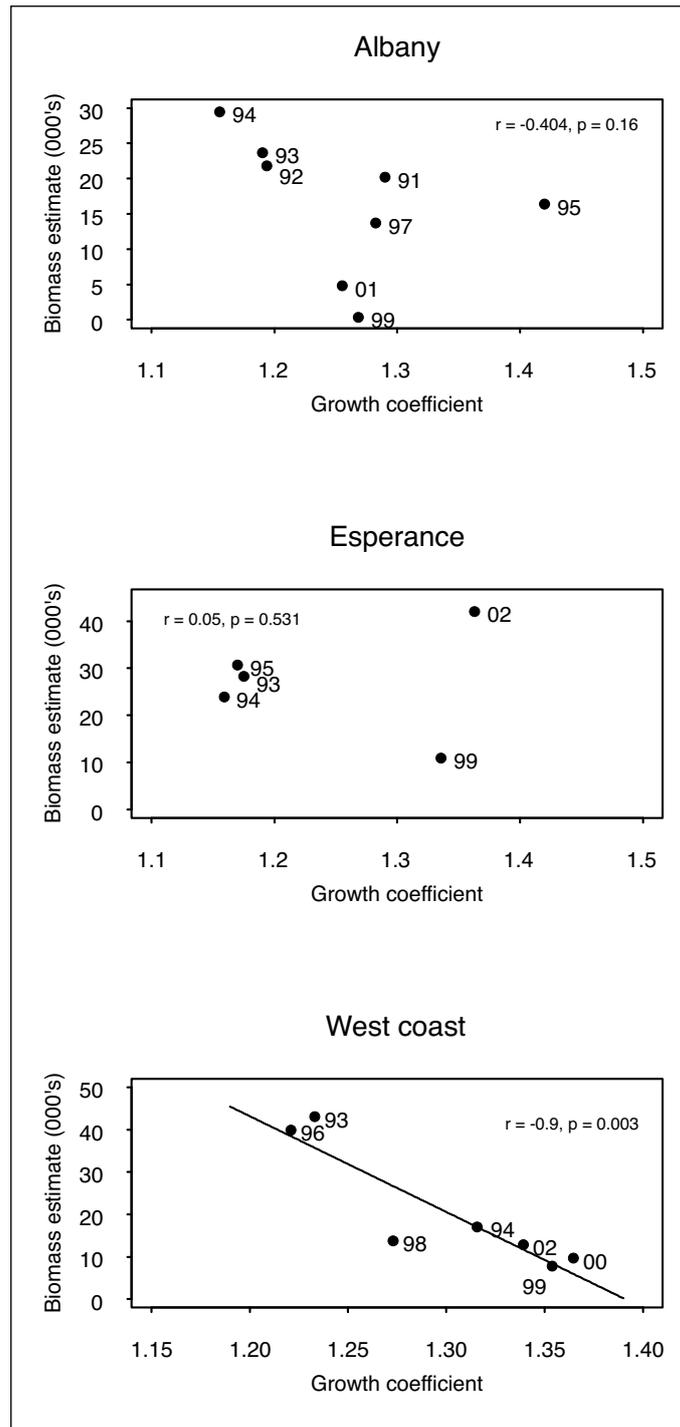


Figure 15. Relationship between mean annual growth coefficients (see text) and spawning biomass in the previous year year (i.e. 1 year lag) for *Sardinops sagax* at three locations Western Australia. The numbers refer to the years that growth coefficients were estimated.

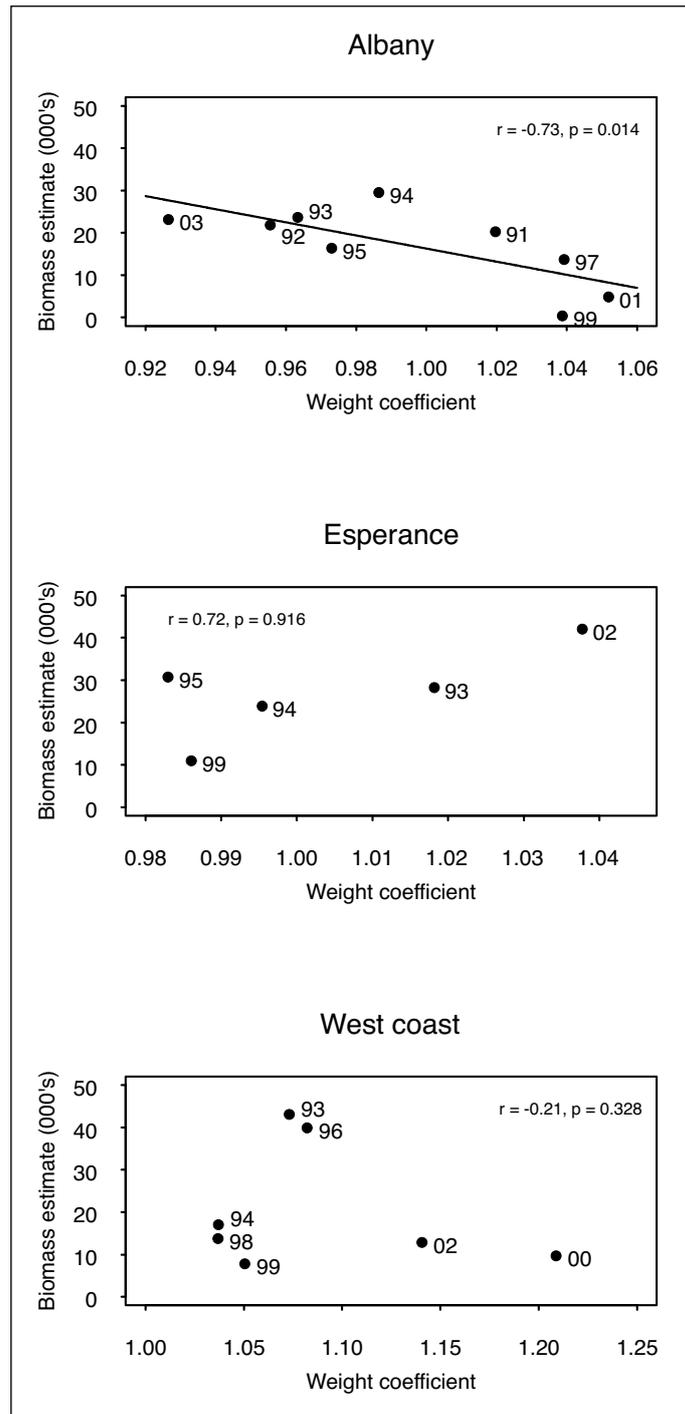


Figure 16. Relationship between mean annual body weight coefficients (see text) and spawning biomass in the current year (i.e. no lag) for *Sardinops sagax* at three locations Western Australia. The numbers refer to the years that growth coefficients were estimated.

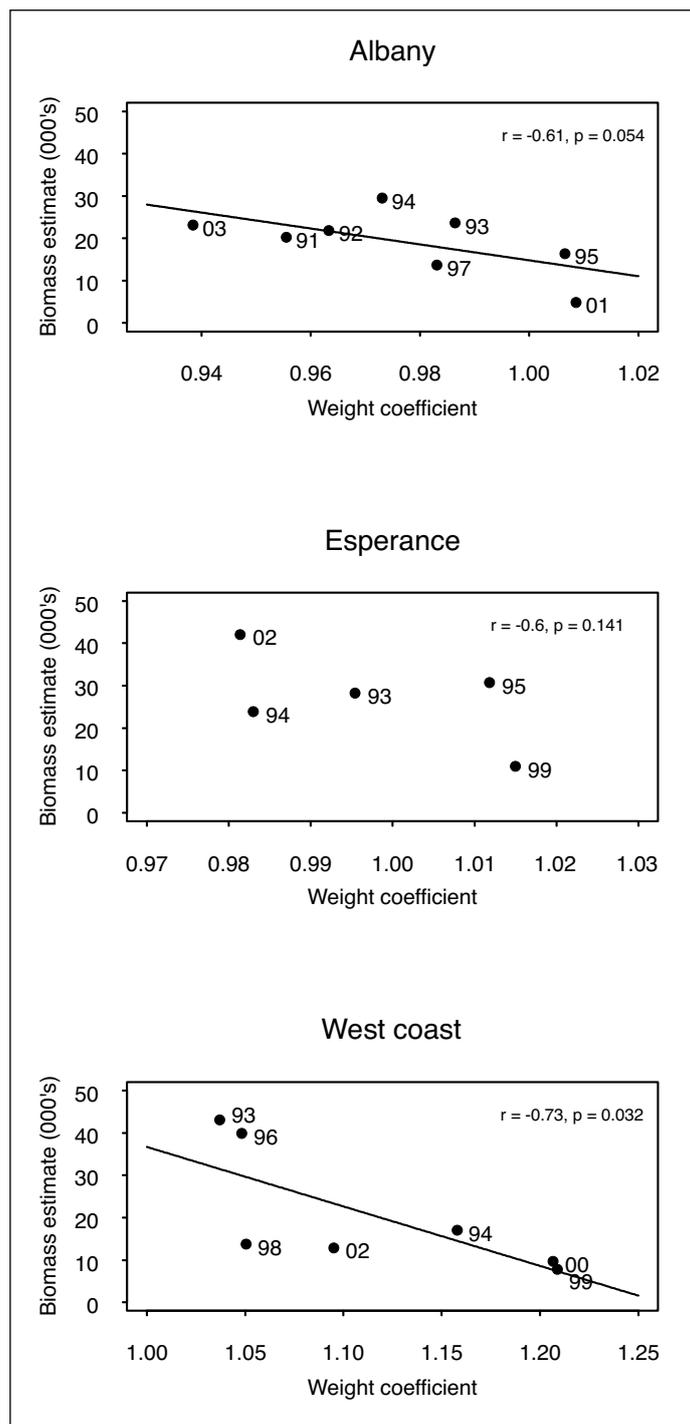


Figure 17. Relationship between mean annual body weight coefficients (see text) and spawning biomass in the previous year year (i.e. 1 year lag) for *Sardinops sagax* at three locations Western Australia. The numbers refer to the years that growth coefficients were estimated.

4.0 Benefits and adoption

The completion of this six-year project coincides with the need to revise the management system for pilchards in WA and to align it with current fisheries management priorities of the WA Government. To put the results of this project in the context of this need for a rationalization of management, a brief history of the provision of stock assessment advice for the purse seine fisheries off southern WA is provided here.

The formal scientific advice for the south coast *Sardinops* fishery was initially based on an age-structured spatial model for the Albany region (Fletcher 1992). The introduction of the DEPM for pilchards on the south coast (Fletcher et al. 1996a, b) represented a significant milestone for management of the purse seine fisheries at Bremer Bay and Esperance because these regions did not have age-structured models at that time. However, at Albany the age-structure model continued to be used alongside the DEPM estimates (which did not occur every year) in the provision of scientific advice. At the same time, i.e. in the early 1990s, fishing industry members were being formally included in management deliberations, via the Purse Seine Ministerial Advisory Committee (PSMAC), and were therefore also becoming considerably more knowledgeable of the management process, including how best to promote their economic concerns to the responsible government portfolio.

The increased focus on economic concerns during the decision-making process was bolstered by the promotion of uncertainty in both the formal stock assessments as well as in the understanding of stock structure and the spatial and behavioural dynamics of the southern WA pilchard population. Nonetheless, Fletcher's (1992) age-structured spatial model did provide a graphical depiction of biomass forecasts, which was particularly useful for highlighting the effects of good and bad recruitment. The output from this quantitative model also had the advantage of being a tangible piece of evidence to present before decision makers, and which could be used to partially alleviate the lack of complete understanding in other areas of the biology and ecology of pilchards. However, industry confidence in using such models has wavered; at one time the Fletcher's Albany model was dismissed from the management process, while later Hall's (2000) age structured model for each south coast region was the basis of significant decisions that impacted on each of the social, economic and sustainability facets of WA's purse seine fisheries.

Over the six-year time-frame for this project members of the WA purse seine industry have closely followed the recovery of the WA pilchard stocks. The regular scrutiny of the DEPM results and model outputs during this period resulted in an increased understanding by industry of the complexities and caveats that must be considered when using stock assessment advice to make management decisions. In accordance with this increased understanding, a key benefit achieved during this project has been the realignment of the provision of scientific advice so that it does not focus only on providing a point estimate on which to base quota levels for any one region. Rather, the scientific advice has been integrated with pragmatic management advice by proactively considering long-term industry goals.

The cessation of catches at the time of the 1998/99 virus event has now been followed by a gradual recovery of pilchard catches to the point where parts of the industry are again viable (Figure 18). While the catches have not approached those from the early to mid 1990s, current catch levels do reflect a combination of stock size, availability of market-sized pilchards, availability of consistent markets and changes in fleet structure. It is also important to note that

pilchard catches in all regions except Albany continue to be well below the TACs but that the available scientific evidence presented in this report indicate that these are for reasons other than the size of the spawning biomass in each region.

West Coast Annual Catch

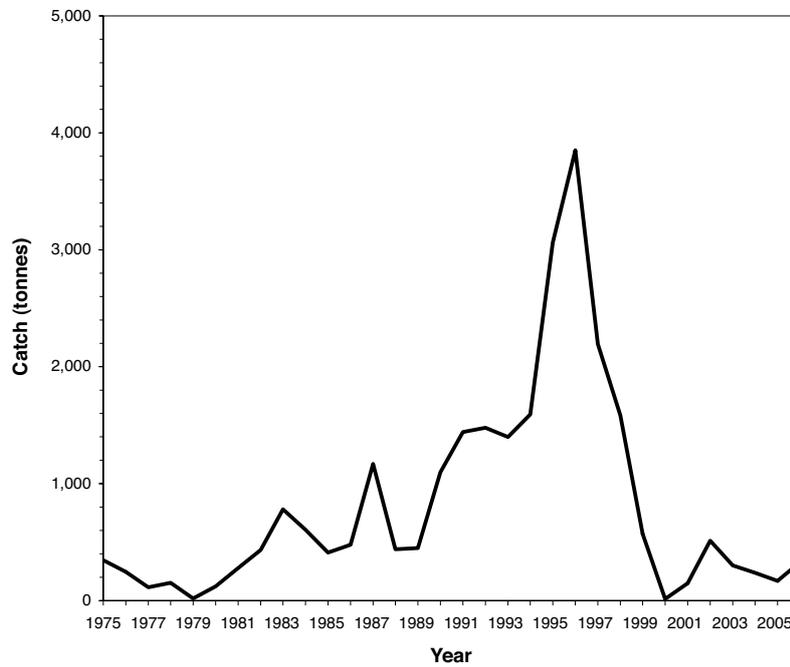


Figure 18a. Pilchard catch (in tonnes) for the West Coast management region of south-western Australia 1975–2006.

South Coast Annual Catch

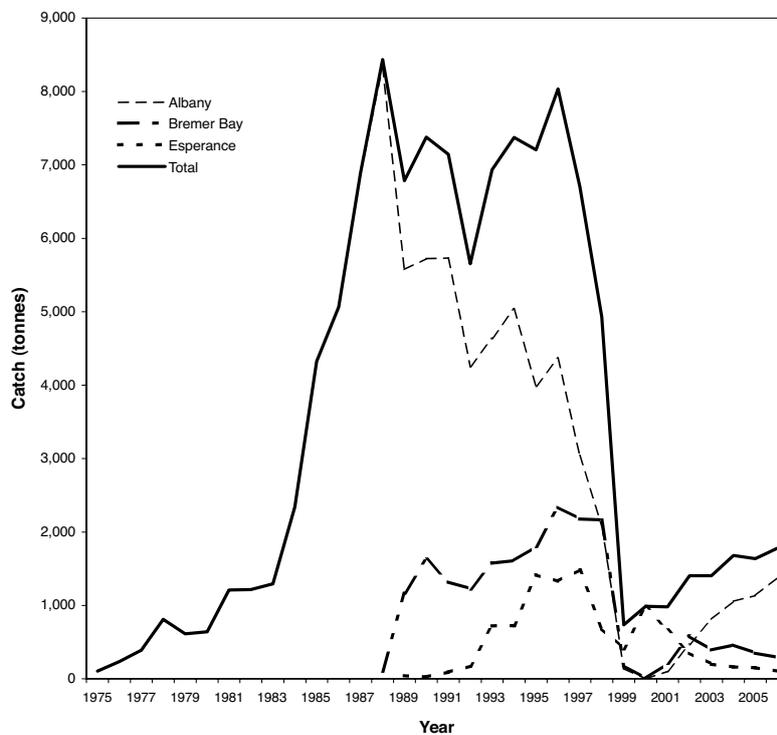


Figure 18b. Pilchard catch (in tonnes) for the South Coast management regions of south-western Australia 1975–2006.

5.0 Further Development

The historical data set of DEPM-based estimates of spawning biomass and the further development of an age-structured model for southern WA pilchards that integrates the DEPM estimates have provided a time-series of biomass trends that can now be considered as indicative of the long-term average size of the pilchard biomass in each purse seine management region. Consequently, further research directed towards estimating pilchard biomass in WA is not warranted because this study indicates that annual point estimates of biomass are not sufficiently precise to be used for setting quota levels. An alternative management system that utilizes more cost-effective criteria such as recruitment and catch levels, and which aims for long term stability in quota levels, has previously been proposed. The results of this project indicate that a review of the purse seine management system in WA should now be pursued to further investigate the implementation of alternative management criteria.

Predictability of inter-annual variability of recruitment strength, spatio-temporal variability in growth rates and stock-recruitment relationships for pilchards in WA remains poor. Currently available data and that which can be obtained from the fishery (i.e. cost effectively) will be further analysed to fill some of the gaps in our knowledge. In particular, recent increases in oceanographic and remote sensing studies in waters off southern WA (e.g. by CSIRO, Curtin University of Technology, University of WA) and the impending larger volume of analysed biogeochemical data (e.g. Hanson et al., 2005) will enhance the ability to consider environmental factors in the assessment of variability in pilchard recruitment and growth rates. Further understanding of the implications of environmental affects on pilchard stocks will in turn improve the scientific advice provided to managers.

6.0 Planned outcomes

Despite the work undertaken to refine the model for southern WA pilchards during this project, the model outputs are still too uncertain to allow management to confidently set anything but low exploitation rates. Given the imprecision of the DEPM, the uncertainty around the point estimates obtained from the integrated model is acknowledged by managers as well as industry members. This has been the basis for responsible decisions regarding the recovery of industry businesses. In addition to the recognition that point estimates of spawning biomass can be imprecise, management of pilchards in WA has used the knowledge that the magnitude and timing of natural variability in their recruitment may operate on decadal time-scales (e.g. Chavez et al., 2003), which necessitates a risk-averse approach to both the setting of exploitation rates and making investment decisions.

The integrated model for southern WA pilchards has been through considerable development and refinement, and has been able to indicate that the spawning biomass has recovered since the 1998/99 mass mortality. However, the main strength of the model in the context of management has been the unambiguous depiction of strong recovery of the stock rather than providing a series of precise point estimates of stock size. The much simpler age structured model developed by Murray and Gaughan (2003) to assess the potential for the pilchard stock to recover after the 1998/99 mass mortality predicted that stocks would recover relatively rapidly. This prediction has now been shown to be correct. The key feature of this model by Murray and Gaughan, and the earlier model by Fletcher (1992) on which the Murray and Gaughan model was based, is that forward projections of biomass trends were primarily reliant, unsurprisingly, on levels of recruitment and fishing mortality. Thus, trends in the growth of the stock based on inputs provided by recruitment and losses due to fishing, under the assumptions of constant growth rates and natural mortality etc, provided depictions of stock recovery that, in retrospect, have been similar to those provided by more the complicated integrated model (i.e. including DEPM surveys).

Because interannual variability in the magnitude of recruitment is a key driver of biomass change in the integrated models, it remains appropriate to use age composition data as an indicator of biomass condition in each region. The value of the DEPM surveys of pilchards in WA throughout the 1990s and the integrated model since 1999 is that there is now a clear indication of the likely long-term size of the pilchard spawning biomass for each management zone, as was previously suggested by Gaughan et al. (2004). This paper summarized the pilchard DEPM surveys undertaken up to 1999, and used these results to propose the likely long-term average spawning biomass in each region. These were then used as a basis for suggesting a small-pelagic fish carrying capacity for each management zone, which industry members should consider as the maximum level against which quota levels could be set.

7.0 Conclusion

Objective 1. Document changes in the biomass of the four exploited adult assemblages of *S. sagax* in WA during the six year period following the 1998/99 mass mortality event.

This objective has been achieved through the application of the DEPM to pilchard stocks off southern WA. The DEPM-based estimates of spawning biomass were sufficient to indicate increases in stock size since 1999 to levels similar to those in the early 1990s. Once the first stages of strong recovery were recorded subsequent to 1999, this information (in combination with the outputs of the integrated model, see below) allowed a resumption of commercial fishing. Importantly, WA's Purse Seine MAC chose to implement a staged increase in total allowable catches (TAC) for the southern management regions so as to ensure a balance between the recovery of the stocks and recovery of their fishery.

Objective 2. Integrate results of biomass surveys into the age-structured model for each assemblage of pilchards. Further develop the age-structured simulation models for *S. sagax* and investigate ongoing behaviour of these under conditions of different levels of information.

This objective was achieved through ongoing development and testing of the integrated model over the period of the project. However, the inability of the model to account for complex spatial dynamics of the pilchards stock necessitates that the model's outputs continue to be cautiously interpreted. The retrospective examination of a much simpler age-structured model by Murray and Gaughan (2003) indicates that knowledge of recruitment variability is the key aspect to tracking changes in spawning biomass of pilchards in WA. Catch sampling should therefore continue for the southern WA purse seine fisheries so that recruitment variability can be monitored.

An integrated model for the West Coast purse seine fishery has not been developed. In retrospect, such a model would have suffered from even greater uncertainty than has been the found for the south coast fisheries because of the much fewer catches, and hence samples, available from the West Coast over recent years. Nonetheless, the lessons from the south coast apply to the west coast. Catch sampling should therefore also continue on the West Coast so that recruitment variability can be monitored.

Objective 3. Determine if there is a measurable change in biological parameters of pilchards important to ongoing development of the simulation models, in particular growth rate, in response to the decrease in stock size.

The difficulties in obtaining large numbers of age estimates using the annuli method, and the inconsistency in annual age distributions of pilchards at each of the regions examined in this study, negated our ability to obtain comparable estimates of the von Bertalanffy growth parameter, k . The lack of estimation of k across years has precluded the inclusion of inter-annual variability in growth to the simulation model. However, the proxies for estimating growth rate (i.e. the OW-based and BW-based growth coefficients) not only provided more comparability over time (~1989 – 2005) but the parameters involved (OW, BW and FL) could be measured objectively and consistently. Thus, rather than estimating growth rate at a given age, this study focussed on assessing relative indicators of growth rate for juvenile fish without concern for their precise ages. Time series for both proxies showed considerable interannual variability; comparisons against stock size found that density dependence may well be an important factor affecting growth rate of pilchards in WA.

8.0 References

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9.0 Appendices

Appendix 1 Intellectual Property

The results of this research are in the public domain, there are no intellectual property implications.

Appendix 2 Staff

Staff employed on the project were:

M. Craine, D. Gaughan, T. Leary, A. Nardi, P. Lewis, P. Stephenson

Appendix 3 The fishery model

An age and sex structured model of the pilchard fishery was developed and fitted to monthly commercial catches, fishery independent spawning biomass estimates, and age composition data.

Monthly commercial catch data were taken from statutory monthly catch and effort returns from 1989 to 2005. Estimates of spawning biomass were obtained from the daily egg production method (DEPM) (Fletcher et al. 1996a) during research surveys conducted between 1989 and 2005. The coefficients of variation varied greatly between areas and years (from 0.18 to 0.7) and when they could not be calculated they were taken as 0.5.

Fletcher and Blight (1996) determined that some pilchards reached an age of nine years, thus the maximum age, A , was taken as 108 months. Age composition data was collected from the catches of commercial vessels from 1989 to 2005. Fish of age 2 to 8 years were used in the analysis with any occasional 1-year-old fish being included in the age 2 year class.

The instantaneous rate of natural mortality, M^γ , was assumed to be constant for pilchard of all ages but could vary between regions. Using a linear relationship between total mortality and fishing effort, Fletcher (1995) used estimates of total mortality from catch curves to calculate an estimate of $M^\gamma = 0.43\text{year}^{-1}$. This value was used in all regions, and as an alternative, M^γ was estimated for each area, Albany, Bremmer Bay, and Esperance.

The eggs and larvae from Albany, Bremer Bay, and Esperance are hypothesized to contribute to a subsequent “pool” of recruits. These early life stages provide a link between the three exploited portions of the stock. Thus, the fisheries at Albany, Bremer Bay and Esperance are considered to exploit a single breeding stock of pilchard. Edmonds and Fletcher (1997) suggested that, following recruitment to the exploited portion of the stock at each fishing location, the fish remain at that location and essentially do not mix across management zones.

Although pilchards in WA can spawn throughout much of the year, spawning generally peaks in autumn and winter (Gaughan et al., 2002) so for simplicity it has been assumed that all age-0 pilchards are recruited at the beginning of July, and ages (measured in months) were calculated using this assumption. The time step, t ($0 \leq t \leq T$), used in the model was monthly, measured relative to the beginning of July 1991 (where $t=0$). The maximum time step was 180 months, corresponds to June 30, 2005. The calendar year was denoted by y .

The assemblage of fish within each area was assumed to be at an unfished equilibrium at July 1991. The initial recruitment (millions of pilchards), $N_{a,0}^{s,\gamma}$, of age a ($a = 0$ to $a = 180$ months) and sex s ($s = f$ and $s = m$) and area γ at the beginning of time-step $t=0$ (July 1989) was calculated as

$$N_{a,0}^{s,\gamma} = \begin{cases} 0 & \text{if } \text{mod}(a,12) \neq 0 \\ \rho_s \theta_{init}^\gamma & \text{if } a = 0 \\ N_{a-12,0}^{s,\gamma} \exp(-M^\gamma) & \text{if } \text{mod}(a,12) = 0 \text{ and } 0 < a < 180 \\ N_{a-12,0}^{s,\gamma} \exp(-M^\gamma) / [1 - \exp(-M^\gamma)] & \text{if } a = 180 \end{cases} \quad (1)$$

where ρ_s is the proportion of each sex with $p_f = 0.59$ and $p_m = 1 - p_f$, θ_{init}^γ is the initial recruitment estimated in the model, and $\text{mod}(a,12) = a - 12\text{int}(a/12)$.

In subsequent years the recruitment, θ_y over all areas was calculated using a Beverton and Holt stock recruitment relationship with steepness set at $h = 0.75$. The numbers of recruits were then apportioned to regions in the same proportion as the initial recruitment, that is

$$\theta_y^\gamma = \frac{\theta_{init}^\gamma}{\sum_\gamma \theta_{init}^\gamma} \theta^\gamma$$

The annual recruitments are variable. Accordingly, an annual recruitment deviation factor was estimated in each fishing region.

While age samples were available to 2005, estimates of the annual recruitment for the more recent years would be relatively imprecise, as the lower vulnerability of the younger fish combined with the few observations of samples containing these year-classes to reduce the information available. Thus to account for the recruitment variability, the recruitments were multiplied by a log-normally distributed recruitment deviation factor. The annual recruitment in each area θ_y^r , is given by

$$\theta_y^r = \begin{cases} \theta_y^r \exp(\eta_y^r) & \text{if } y \leq 2002 \\ \theta_y^r \exp(\varphi_y^r) & y > 2002 \end{cases} \quad (2)$$

where η_y^r is estimated in the model and $\varphi_y^r \sim N(0, \sigma^{\gamma^2})$ are log-normally distributed random variates with standard deviation equal to that of η_y^r .

The proportion of females and males in the recruiting year-class was assumed to be $p_f = 0.58$ and $p_m = 1 - p_f$, respectively (Fletcher et al., 1996b).

The number of pilchard (millions), $N_{a,t}^{s,r}$, of age a and sex s in region r at the beginning of time-step t was calculated for July, 1989, when $t = 0$, as:

$$N_{a,0}^{s,r} = \begin{cases} 0 & \text{if } (a \bmod 12) \neq 0 \\ P_s \theta_{limit}^r & \text{if } a = 0 \\ N_{a-12,0}^{s,r} \exp(-M) & \text{if } (a \bmod 12) = 0 \text{ and } 0 < a < 180 \\ N_{a-12,0}^{s,r} \exp(-M) / [1 - \exp(-M)] & \text{if } a = 180 \end{cases} \quad (3)$$

The numbers of pilchards in successive timesteps were determined by removing the numbers due to fishing and natural mortality and introducing new recruits according to

$$N_{a,t}^{s,r} = \begin{cases} \mathcal{G}_t^r & \text{if } a = 0 \text{ and } t = 1 \\ N_{a-1,t-1}^{s,r} (1 - H_{t-1}^r V_{a-1}^r) \exp\left(-\frac{1}{12} M^\gamma\right) & \text{if } a > 0 \text{ and } a \neq 180 \\ \left[N_{a-1,t-1}^{s,r} (1 - H_{t-1}^r V_{a-1}^r) + N_{a+11,t-1}^{s,r} (1 - H_{t-1}^r V_{a+11}^r) \right] \exp\left(-\frac{1}{12} M^\gamma\right) & \text{if } a = 180 \\ N_{a-1,t-1}^{s,r} (1 - H_{t-1}^r V_{a-1}^r) (1 - D_{1995} V_{a-1}^r) \exp\left(-\frac{1}{12} M^\gamma\right) & \text{if } t = 70 \\ N_{a-1,t-1}^{s,r} (1 - H_{t-1}^r V_{a-1}^r) (1 - D_{1999}) \exp\left(-\frac{1}{12} M^\gamma\right) & \text{if } t = 115 \end{cases} \quad (4)$$

where H_t^r is the harvest rate (proportion of the vulnerable pilchards caught) for time-step t in region γ , V_a^r is the proportion of pilchards of age a in region r that are vulnerable, and D_{1995} and D_{1999} are the estimated proportions of the affected age-classes that died as a result of disease in 1995 and 1998–1999, respectively.

The mass mortality of adult pilchard in April–May of 1995 was initially estimated as resulting in a diminution of the stock by 10–15% at each location. However, a subsequent review of information on the impact of this disease from all locations throughout Australia has led to the conclusion that the mortality was possibly twice that reported by Fletcher et al. (1997). For the model, it was assumed that there was an instantaneous mortality of 25% for vulnerable pilchards ($D_{1995} = 0.25$), where vulnerability at age determined the proportion of each year-class that was affected.

The second outbreak of disease, which was first observed in South Australia, caused mass mortality of pilchards between November 1998 and March 1999. The disease front moved slowly westward, with pilchards in each region experiencing ~70% mortality. (Gaughan et al., 2000). Accordingly the model assumed a 70% loss ($D_{1999} = 0.7$) in February 1999.

Although pilchards were assumed to be recruited into each assemblage at age 0, they do not become immediately vulnerable to exploitation (i.e. recruit to the exploited portion of the stock). Fletcher (1995) suggested that recruitment to the exploited portion of the stock at each fishing location commences when pilchards reach two years of age, with full recruitment occurring by age four. Preliminary investigation using the age-structured model suggested that the age at which pilchard became vulnerable might differ between regions, with some pilchard not becoming recruited into the fishery until age five or older. The pattern of vulnerability at age suggested by these initial explorations appeared to follow a logistic relationship.

Accordingly, vulnerability at age, V_a^r , was represented by a logistic equation where the parameters determining the mean age (years), $a_{0.5}^r$, at which the pilchard became vulnerable and the age (years), $a_{0.95}^r$, at which 95% of the fish became vulnerable within the region were estimated within the model. Vulnerability at age a (months) was assumed constant throughout the period from July to June, while fish remained within the same age-class (years). Hence,

$$V_a^r = \frac{1}{1 + \exp\left[-\ln(19)\left(\frac{\text{int}(a/12) - a_{0.5}^r}{a_{0.95}^r - a_{0.5}^r}\right)\right]}, \quad (5)$$

where $\text{int}(x)$ is a function that returns the largest integer less than or equal to the real number, x .

When pilchards became scarce after the mortality events, the fishery moved to catching lower valued smaller fish. Consequently the vulnerability was altered so that $a_{0.5}^r$ and $a_{0.95}^r$ were reduced in proportion to the ratio of the biomass of 4+ to 2+ compared to the initial ratio of 4+ to 2+ fish in 1989.

Vulnerable biomass (t) was calculated at each time-step from the estimated numbers (millions) of pilchard of each age and sex at the beginning of the time-step, using estimates of the vulnerability at age and the calculated body weight (kg) at age:

$$\hat{B}_t^{s,r} = 1000 \sum_{a=0}^A N_{a,t}^{s,r} V_a^r W_a^s \quad (6)$$

$$\hat{B}_t^r = \sum_{s=m,f} \hat{B}_t^{s,r} \quad (7)$$

where $\hat{B}_t^{s,r}$ is the estimated vulnerable biomass (tonnes) at the beginning of time-step t for pilchard of sex s in region r , \hat{B}_t^r is the estimated total vulnerable biomass (tonnes) in region r at the start of time-step t , and W_a^s is the body weight (kg) of a fish of age a and sex s .

Growth in fork length (mm) has been represented for pilchards of each sex using the von Bertalanffy growth curve:

$$L_a^s = L_\infty^s \left\{ 1 - \exp\left[-K^s \left(\frac{a}{12} - t_0^s\right)\right] \right\} \quad (8)$$

where L_a^s is the fork length (mm) of a fish of age a months ($0 \leq a \leq A$) and sex s ($s = m$ for males, $s = f$ for females), and L_∞^s , K^s , and t_0^s are parameters of the growth curve. For males, the estimated parameters were $L_\infty^m = 164$ mm, $K^m = 0.728$ year⁻¹ and $t_0^m = 1$ year, while for females the values were $L_\infty^f = 172$ mm, $K^f = 0.6629$ year⁻¹ and $t_0^f = 1$ year (Fletcher, 1995).

The relationship between the body weight (kg) and fork length (mm) had been determined (Gaughan, unpublished data) as:

$$W_a^s = 0.000000010834(L_a^s)^{2.9782}. \quad (9)$$

Since the model was driven by catch, harvest rates applied at each step within the model were calculated from the observed catch (t), C_t^r , and the current estimate of vulnerable biomass:

$$H_t^r = \begin{cases} \frac{C_t^r}{\hat{B}_t^r} & \text{if } C_t^r < 0.9\hat{B}_t^r \\ 0.9 & \text{otherwise} \end{cases} \quad (10)$$

where H_t^r is the harvest rate (proportion of the vulnerable biomass at the start of the month

that is caught) for month t in region r .

To ensure that the model parameters produced estimates of vulnerable biomass sufficiently large to allow the removal of the observed catch, a maximum harvest rate of 90% was imposed; a penalty function, P_1^r , was subtracted from the log-likelihood if the catch exceeded 90% of the vulnerable biomass in the region.

$$P_1^r = 100 \sum_{\substack{t=0 \\ C_t^r > 0.9B_t^r}}^T (C_t^r - 0.9\hat{B}_t^r)^2 \quad (11)$$

The factor, 100, was arbitrarily chosen, but appeared sufficient to ensure that harvest rates in excess of 90% did not occur.

Observations of spawning biomass, B_t^r , and their estimated coefficients of variation, c_t^r , determined from the daily egg production surveys, were input to the model. Where no coefficient of variation was available, it was assumed that $c_t^r = 0.5$. Errors were assumed to be normally distributed, and an estimate of the log-likelihood associated with these biomass observations (ignoring constant terms) was obtained using

$$\lambda_1^r = - \frac{\sum_{\substack{t \\ B_t^r \text{ exists}}} (B_t^r - \hat{B}_t^r)^2}{2(c_t^r B_t^r)^2} \quad (12)$$

Estimates of the proportion, $\hat{p}_{j,t}^r$, of vulnerable fish within each age-class j , combining sexes, were calculated for the 2+ to 8+ age classes after recruitment in July using the equation

$$\hat{p}_{j,t}^r = \frac{\sum_{s=m,f} N_{a,t}^{s,r} V_a^r}{\sum_{\substack{a=0 \\ j \geq 2}}^A \sum_{s=m,f} N_{a,t}^{s,r} V_a^r}, \quad (13)$$

where $j = \text{int}(a/12)$.

It was assumed that the observed age samples represented samples from a multinomial distribution. The log-likelihood associated with these samples was therefore calculated (ignoring constant terms) using

$$\lambda_2^r = \sum_{\substack{t \\ \text{month}=7 \\ \text{Sample exists}}} \left\{ n_{\text{Total},t}^r \sum_{j=2}^8 \{ p_{j,t}^r \log(\hat{p}_{j,t}^r) \} \right\}, \quad (14)$$

where $n_{\text{Total},t}^r$ was the total number of pilchard from the 2+ to 8+ age classes that were sampled for age determination during the 12 month period from July to June. The observed proportion,

$p_{j,t}^r$, in the age class, j , was derived from estimates of the number of pilchard in each age-class within the catch (from the 2+ to 8+ age classes).

A log-likelihood, λ_4^r , was also calculated (ignoring constant terms) for the random elements of the recruitment deviations:

$$\lambda_3^r = -\frac{\sum (\eta_y^r)^2}{2\sigma_\eta^2}, \quad (15)$$

where the standard deviation of recruitment deviations, σ_η , was assumed to be 0.4. The selection of this value was arbitrary, and the sensitivity of the model to this choice remains to be tested.

The log-likelihoods for the observed biomass estimates from research surveys, age composition data, and recruitment deviations were combined, together with the penalty functions to form the overall log-likelihood:

$$\lambda^r = \lambda_1^r + \lambda_2^r + \lambda_3^r - P_1^r. \quad (16)$$

These objective functions were maximised simultaneously for each region to obtain maximum likelihood estimates of the parameters for that region. The penalty function had little effect (i.e. were negligible) when the parameter estimates approached the maximum likelihood estimates.

Parameters were estimated for each of the regions (Albany, Bremer Bay, and Esperance) including the initial recruitment, estimates of natural mortality (when fitted) the two parameters of the selectivity curve, and annual recruitment deviations. The model was fitted to the data simultaneously for each fishing region using the software package, AD Model Builder (Fournier, 1994) to determine parameter estimates and their standard deviations and the variance-covariance matrix. A Bayesian procedure was used to determine estimates of uncertainty of the mature biomass using Markov Chain Monte Carlo procedure in AD Model Builder.

Appendix 4 **Developing a statistically robust growth coefficient**

4.1 **Developing a statistically robust growth coefficient across all ages/sizes**

This Appendix describes the statistical techniques used to

(1) develop a growth rate coefficient for pilchards that used all sizes of pilchards, and (2) compare annual growth rates in each of four management zones.

Several relationships will be assessed to determine whether or not there had been a change in growth rates of pilchards associated with changing stock size. Each of the relationships was developed using indirect techniques so that we could take advantage of the 14-year data set of biological measurements of length, weight, and otolith weight.

Model description

For each year ($j = 1, \dots, T$), fit a nonlinear power curve to the otolith weight (y) v. total length (x) data, viz.

$$y_{ij} = a_j x_{ij}^{b_j} + \varepsilon_{ij}.$$

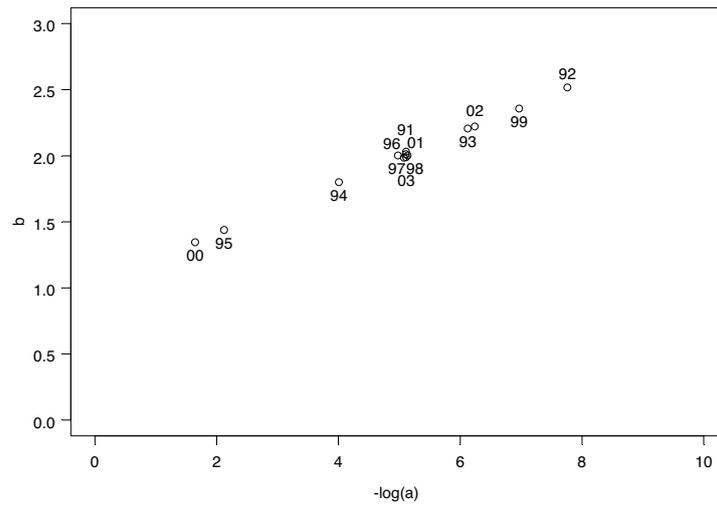
The a_j 's and b_j 's are parameters to be estimated and $\varepsilon_j \sim N(0, \sigma_j^2)$. We can make inferences on differences in growth rates among years, but it is difficult to assert that pilchards grew faster during one year than some other year, since a and b are highly correlated ($r > 0.9$) for each j . Moreover, the a_j v. b_j relationship closely follows a locus described by $a_j k^{b_j} = p$, where k and p are fixed constants. That is, there is an approximate linear relationship between $\log a_j$ and b_j for each j when the a_j 's and b_j 's are respectively highly correlated (see Figures below and Appendix 4.2 below for the derivation). We use this property to our advantage to estimate a single-parameter growth coefficient for 1+ - 2+ year pilchards in each year.

Put $a_j k^{b_j} = p$ for all $j = 1, \dots, T$. Then the model becomes

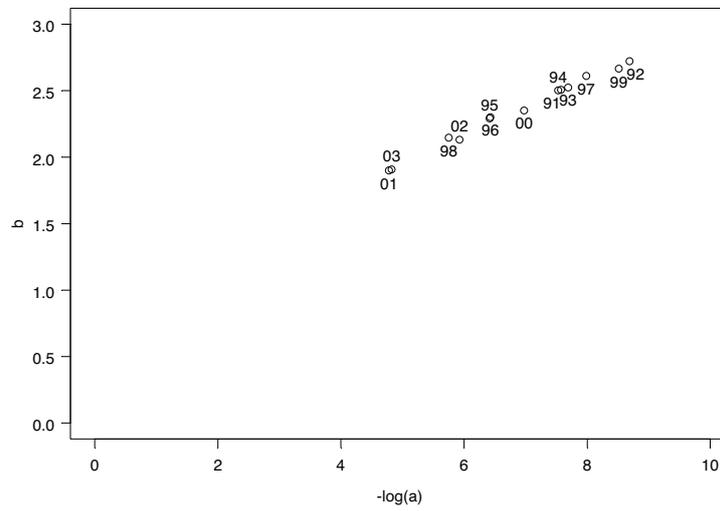
$$y_{ij} = a_j x_{ij}^{(\log p - \log a_j) / \log k} + \varepsilon_{ij}.$$

a_j, k, p are parameters to be estimated using non-linear regression and the errors ε_{ij} are normally distributed with zero mean and constant variance across *all* years of data.

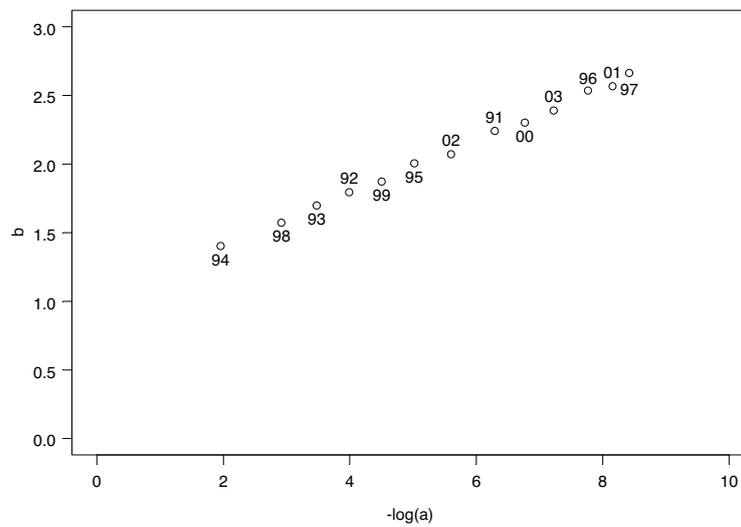
Albany



Esperance



West coast



Albany

There are generally significant growth rate differences between sex over all years (1989–2003) in Albany. The nonlinear regression model is therefore

$$y_{ij} = e^{\alpha_j + \beta} x_{ij}^{(\phi - \alpha_j - \beta)/\kappa} + \eta_{ij}.$$

Bremer Bay

There is no significant sex growth rate difference for any years 1989–2003. The nonlinear regression model used is thus

$$y_{ij} = e^{\alpha_j} x_{ij}^{(\phi - \alpha_j)/\kappa} + \eta_{ij}.$$

Esperance

Using the separate year models, there are significant growth rate differences by sex during the years coinciding with herpesvirus outbreaks (1995–96, 1998–99), however, generally no significant sex differences in growth rate exist for the remaining years. A proportional growth rate variable is included in the improved dummy variable nonlinear regression model, viz.

$$y_{ij} = e^{\alpha_j + \beta h_1} x_{ij}^{(\phi - \alpha_j - \beta h_1)/\kappa} + \eta_{ij}.$$

Here, $h_1 = \begin{cases} 1 & \text{for } j = 1996 - 97, 2000 - 03, \\ 0 & \text{otherwise,} \end{cases}$

β is a sex difference parameter to be estimated and $\eta_{ij} \sim N(0, \sigma^2)$. We fit a nonlinear regression model to the given years (1991–2003) of otolith weight and total length data and use maximum likelihood methods to estimate the parameters.

West Coast

There are significant sex growth rate differences for years 1989–94 and 1997–99, but not for 1995–96 and 2001–03. The model used is

$$y_{ij} = e^{\alpha_j + \beta h_2} x_{ij}^{(\phi - \alpha_j - \beta h_2)/\kappa} + \eta_{ij},$$

where

$h_2 = \begin{cases} 1 & \text{for } j = 1989 - 94, 1997 - 99, \\ 0 & \text{otherwise.} \end{cases}$

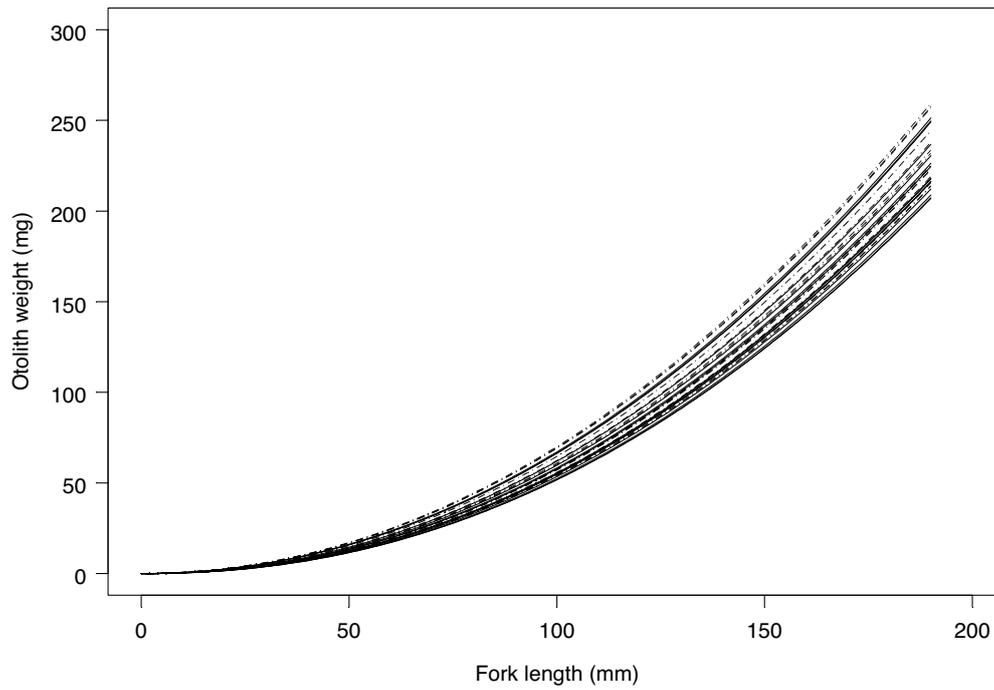
Results

Although ϕ and κ are highly correlated ($r \geq 0.998$) for all regions, the important result of the model is that the α_j 's and β are generally not highly correlated with ϕ or κ , and are only highly correlated where the growth rates are similar. This gives us confidence with the interpretation of the growth rates and sex differences of pilchards for each year. Growth rate differences over year groups of data, where they were significant, were 2–10 times larger in magnitude than sex differences.

Albany

Parameter	Coefficient (s.e.)
$\hat{\alpha}_{1989}$	-5.805 (0.074)
$\hat{\alpha}_{1990}$	-5.838 (0.077)
$\hat{\alpha}_{1991}$	-5.841 (0.077)
$\hat{\alpha}_{1992}$	-5.910 (0.085)
$\hat{\alpha}_{1993}$	-5.694 (0.072)
$\hat{\alpha}_{1994}$	-5.494 (0.100)
$\hat{\alpha}_{1995}$	-5.302 (0.141)
$\hat{\alpha}_{1996}$	-5.293 (0.146)
$\hat{\alpha}_{1997}$	-5.268 (0.153)
$\hat{\alpha}_{1998}$	-5.595 (0.085)
$\hat{\alpha}_{1999}$	-6.004 (0.107)
$\hat{\alpha}_{2000}$	-5.666 (0.103)
$\hat{\alpha}_{2001}$	-5.970 (0.096)
$\hat{\alpha}_{2002}$	-5.813 (0.074)
$\hat{\alpha}_{2003}$	-5.999 (0.101)
$\hat{\kappa}$	7.127 (0.802)
$\hat{\phi}$	9.394 (1.703)
$\hat{\beta}$	0.119 (0.033)

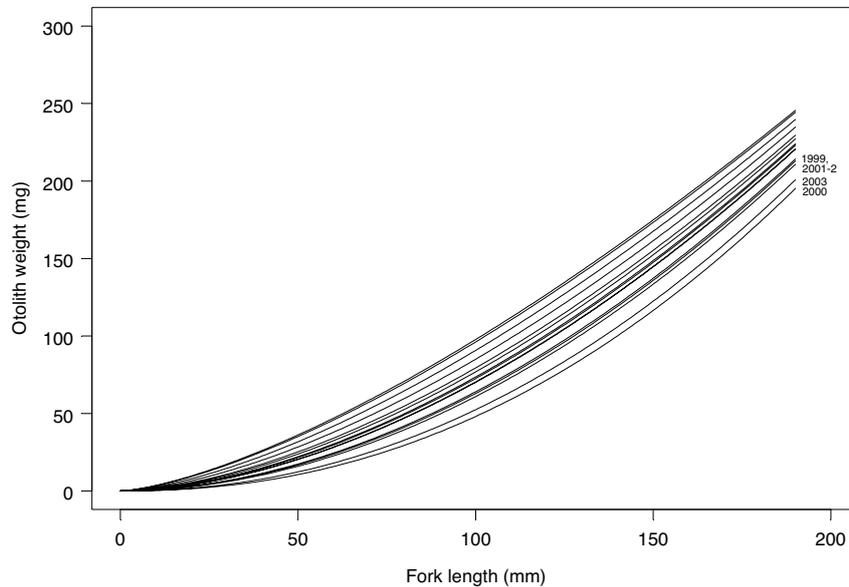
For Albany, a similar model analysis to that of Esperance shows that the otolith weight relative to fork length increases (decreases) as α increases (decreases) for all fish less than $e^{\hat{\kappa}} \approx 1245$ mm. From this result and the table, we see that growth rates for Albany were significantly ($p < 0.01$) slower during 1995–97 than for all other years. All other years of growth rates were approximately equal. Females grew significantly ($p < 0.01$) faster than males over all years.



Bremer Bay

Parameter	Coefficient (s.e.)
$\hat{\alpha}_{1989}$	-3.244 (0.119)
$\hat{\alpha}_{1990}$	-3.692 (0.121)
$\hat{\alpha}_{1991}$	-3.771 (0.115)
$\hat{\alpha}_{1992}$	-3.954 (0.117)
$\hat{\alpha}_{1993}$	-3.971 (0.099)
$\hat{\alpha}_{1994}$	-3.436 (0.121)
$\hat{\alpha}_{1995}$	-2.843 (0.130)
$\hat{\alpha}_{1996}$	-2.464 (0.147)
$\hat{\alpha}_{1997}$	-2.027 (0.177)
$\hat{\alpha}_{1998}$	-2.122 (0.177)
$\hat{\alpha}_{1999}$	-4.608 (0.136)
$\hat{\alpha}_{2000}$	-6.183 (0.180)
$\hat{\alpha}_{2001}$	-4.503 (0.096)
$\hat{\alpha}_{2002}$	-4.795 (0.109)
$\hat{\alpha}_{2003}$	-5.673 (0.144)
$\hat{\kappa}$	5.554 (0.037)
$\hat{\phi}$	5.947 (0.069)

For Bremer Bay, the otolith weight for a given fork length increases (decreases) as α increases (decreases) for all fish with fork length less than $e^{\kappa} = 258$ mm. This range includes all fish in the sample (1989–2003). From the table, we see that the fish (independently of sex) were growing significantly ($p < 0.01$) faster from 1999 onwards (following the major herpesvirus outbreak) compared with pre-1999 growth rates.



Esperance

Parameter	Coefficient (s.e.)
$\hat{\alpha}_{1991}$	-7.565 (0.147)
$\hat{\alpha}_{1992}$	-7.414 (0.132)
$\hat{\alpha}_{1993}$	-7.362 (0.118)
$\hat{\alpha}_{1994}$	-7.619 (0.149)
$\hat{\alpha}_{1995}$	-7.788 (0.168)
$\hat{\alpha}_{1996}$	-8.035 (0.210)
$\hat{\alpha}_{1997}$	-8.098 (0.212)
$\hat{\alpha}_{1998}$	-7.446 (0.124)
$\hat{\alpha}_{1999}$	-6.624 (0.090)
$\hat{\alpha}_{2000}$	-6.464 (0.100)
$\hat{\alpha}_{2001}$	-6.085 (0.142)
$\hat{\alpha}_{2002}$	-6.097 (0.145)
$\hat{\alpha}_{2003}$	-6.094 (0.150)
$\hat{\phi}$	3.034 (0.235)
$\hat{\kappa}$	4.228 (0.101)
$\hat{\beta}$	-0.171 (0.035)

The first derivative w.r.t. α of the function

$$f(\alpha; x, \hat{\phi}, \hat{k}) = \alpha x^{(\hat{\phi} - \log \alpha) / \hat{k}}$$

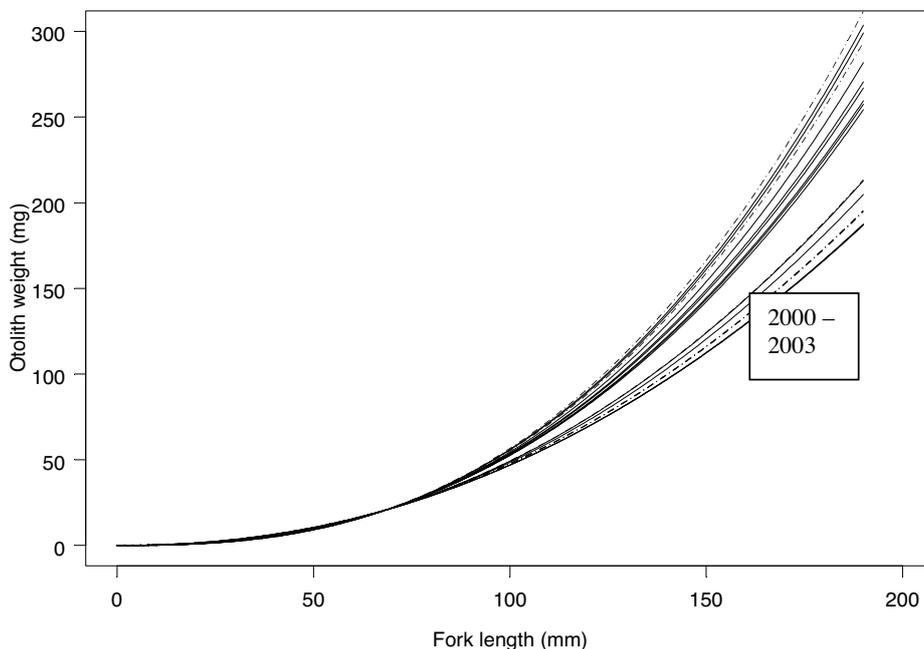
is defined by

$$f'(\alpha; x, \hat{\phi}, \hat{k}) = x^{(\hat{\phi} - \log \alpha) / \hat{k}} \left(1 - \frac{\log x}{\hat{k}} \right).$$

Thus, f is decreasing for $x > e^{\hat{k}} \approx 69$. That is, an increase (decrease) in α will lead to a decrease (increase) in otolith weight relative to fork length provided the fork length is greater than 69 mm. We therefore restrict the following result for Esperance to those fish which are longer than 69 mm. This eliminates only 3 out of 12012 fish between 1991 and 2003.

From the table and the above analysis, we see that growth rates for Esperance were broadly split into four significantly distinct year groups. Growth rates from slowest to fastest were (1) 1996–97, (2) 1991–95 and 1998, (3) 1999–2000 and (4) 2001–03. The means for each group differ pairwise by at least three sample standard deviations. Females grew significantly ($p < 0.01$) faster than males during 1995–96 and 2000–2003.

The following figure illustrates the significant difference in fitted fork length against otolith weight between the 2000–2003 group (longer) and the other three year groups (shorter). Solid lines indicate relationship for females, broken lines for males.

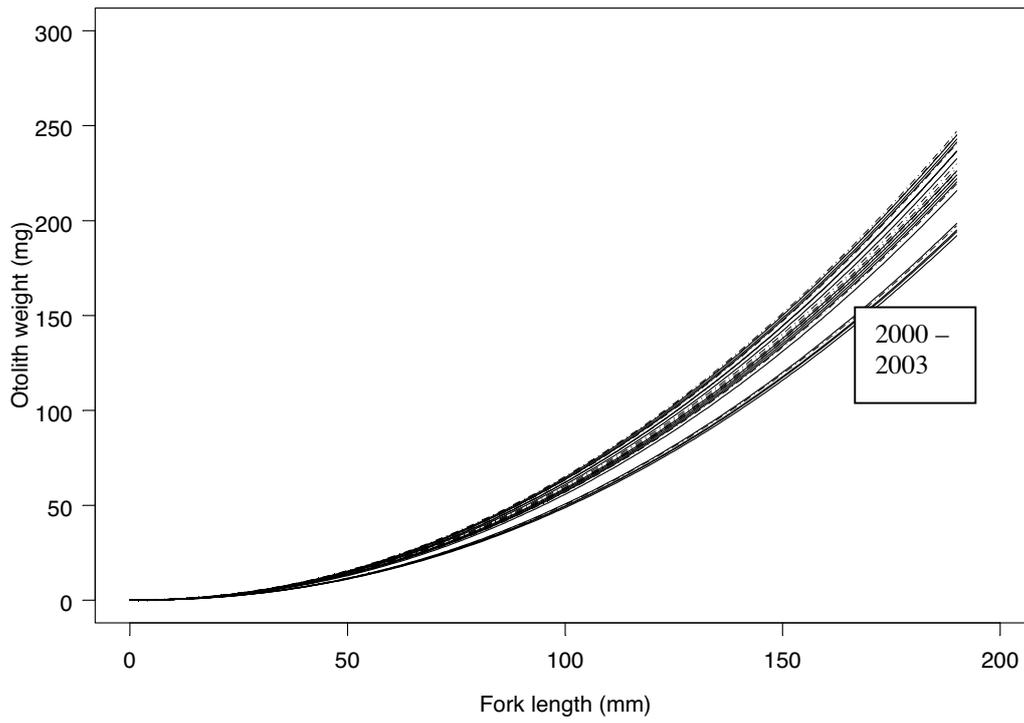


West Coast

Parameter	Coefficient (s.e.)
$\hat{\alpha}_{1989}$	-5.631 (0.108)
$\hat{\alpha}_{1990}$	-5.613 (0.104)
$\hat{\alpha}_{1991}$	-5.570 (0.105)
$\hat{\alpha}_{1992}$	-5.468 (0.120)
$\hat{\alpha}_{1993}$	-5.407 (0.131)
$\hat{\alpha}_{1994}$	-5.422 (0.131)
$\hat{\alpha}_{1995}$	-5.387 (0.136)
$\hat{\alpha}_{1996}$	-5.468 (0.119)
$\hat{\alpha}_{1997}$	-5.507 (0.114)
$\hat{\alpha}_{1998}$	-5.593 (0.104)
$\hat{\alpha}_{1999}$	-5.679 (0.100)
$\hat{\alpha}_{2000}$	-5.922 (0.145)
$\hat{\alpha}_{2001}$	-5.947 (0.147)
$\hat{\alpha}_{2002}$	-5.911 (0.136)
$\hat{\alpha}_{2003}$	-5.870 (0.133)
\hat{k}	9.300 (2.593)
$\hat{\phi}$	13.912 (5.453)
$\hat{\beta}$	0.037 (0.010)

For the west coast the model analysis is valid provided the otolith weight relative to fork length increases (decreases) as α increases (decreases) for all fish less than $e^k \approx 10938$ mm, i.e. all fish. From this result and the table, we see that growth rates for the west coast were significantly ($p < 0.01$) faster from 2000–03 than prior to 2000 when growth rates were approximately equal. Females grew significantly ($p < 0.01$) faster than males during 1989–1994 and 1997–2000.

Growth rate differences between sexes during the 2000–2003 period (longer) and the 1989–1999 period (shorter) can clearly be seen in the following figure illustrating the west coast otolith weight v. fork length fits.



4.2 Derivation required to ensure the models of the otolith weight – fork length relationships are robust

Our model is

$$OW_{ij} = a_j L_{ij}^{b_j} + \varepsilon_{ij}, \quad \varepsilon_{ij} \sim N(0, \sigma^2) \quad (1)$$

The residual sum of squares over all years $j = 1, \dots, T$ is obtained from the likelihood, viz.

$$S = \sum_{j=1}^T \sum_i (OW_{ij} - a_j L_{ij}^{b_j})^2$$

The minimum for S w.r.t. a_j and b_j are found by solving the equations:

$$\frac{\partial S}{\partial a_j} = -2 \sum_i (OW_{ij} - a_j L_{ij}^{b_j}) L_{ij}^{b_j} = 0$$

$$\frac{\partial S}{\partial b_j} = -2 a_j \sum_i (OW_{ij} - a_j L_{ij}^{b_j}) L_{ij}^{b_j} \log L_{ij} = 0$$

These equations are linearly dependent for each j iff

$$\begin{vmatrix} -\sum_i L_{ij}^{2b_j} & \sum_i L_{ij}^{b_j} \log L_{ij} (OW_{ij} - 2a_j L_{ij}^{b_j}) \\ \sum_i L_{ij}^{b_j} \log L_{ij} (OW_{ij} - 2a_j L_{ij}^{b_j}) & a_j \sum_i L_{ij}^{b_j} (\log L_{ij})^2 (OW_{ij} - 2a_j L_{ij}^{b_j}) \end{vmatrix} \equiv 0 .$$

The determinant gives

$$\begin{aligned} -a_j \sum_i L_{ij}^{2b_j} \sum_i L_{ij}^{b_j} (\log L_{ij})^2 (\varepsilon_{ij} - a_j L_{ij}^{b_j}) &\equiv \left[\sum_i L_{ij}^{b_j} \log L_{ij} (\varepsilon_{ij} - a_j L_{ij}^{b_j}) \right]^2 \\ &= a_j^2 \left(\sum_i L_{ij}^{2b_j} \log L_{ij} \right)^2 \end{aligned} \quad (2)$$

Supposing that, for each j , the following assumptions hold:

- $\sum_i \varepsilon_{ij}^2$ is small, say $< 4\%$ compared with $\sum_i OW_{ij}^2$ for each j , i.e., $E[OW_{ij}] = a_j L_{ij}^{b_j}$,

where a_j and b_j are MLE estimators, is a significantly “good fit”;

- $b_j > 1$ for all j .

- $\frac{1}{a_j}$ is approximately proportional to $\overline{L_{\bullet j}}^{-b_j}$. That is, $-\log a_j$ and b_j are linearly related.

The above assumptions are sufficient to guarantee the above equation approximately holds.

More importantly, if the first two assumptions hold, then the third is a necessary condition for the above equation to approximately hold, since the a_j 's and b_j 's are *unbiased* MLE estimators.

Conjecture 1:

LHS and RHS of (1) are within 0.5% of being equal.

Proof:

Consider the following property.

Property 1:

Assume

$$\sum_i x_i y_i = r_1 \sqrt{\sum_i x_i^2} \sqrt{\sum_i y_i^2} > 0$$

$$\sum_i y_i z_i = r_2 \sqrt{\sum_i y_i^2} \sqrt{\sum_i z_i^2} > 0.$$

Then, by the Cauchy-Schwarz inequality,

$$\sum_i y_i^2 \sum_i x_i z_i \leq \frac{1}{r_1 r_2} \sum_i x_i y_i \sum_i y_i z_i$$

Now consider the following two quantities in (1), viz.

$$\sum_i L_{ij}^{2b_j} \sum_i L_{ij}^{2b_j} (\log L_{ij})^2 \quad \text{and} \quad \left(\sum_i L_{ij}^{2b_j} \log L_{ij} \right)^2.$$

We show that they are almost equal. The Cauchy-Schwarz inequality guarantees that

$$\sum_i L_{ij}^{2b_j} \sum_i L_{ij}^{2b_j} (\log L_{ij})^2 \geq \left(\sum_i L_{ij}^{2b_j} \log L_{ij} \right)^2.$$

Property (1), with $x_i = z_i = L_i^{b_j}$, $y_i = L_i^{b_j} \log L_i$, gives

$$\sum_i L_{ij}^{2b_j} \sum_i L_{ij}^{2b_j} (\log L_{ij})^2 \leq \frac{1}{r^2} \left(\sum_i L_{ij}^{2b_j} \log L_{ij} \right)^2,$$

where $0.9995 < r < 1$. Hence, the two quantities are within 0.1% of one another.

This leaves the third term in (1). Here, we are required to show that

$$\left| \frac{\sum_i L_{ij}^{2b_j} \sum_i L_{ij}^{b_j} (\log L_{ij})^2 \varepsilon_{ij}}{a_j \left(\sum_i L_{ij}^{2b_j} \log L_{ij} \right)^2} \right| < 3.3 \times 10^{-3}.$$

Now, from the second given property, we find:

$$\text{Cor}(L_j^{b_j} \log L_j, L_j^{b_j}) > 0.9995$$

$$\text{Cor}(L_j^{b_j} \log L_j, \log L_j) \approx 0.97$$

$$\text{Cor}(L_j^{2b_j}, \log L_j) \approx 0.9,$$

so

$$\left| \frac{\sum_i L_{ij}^{2b_j} \sum_i L_{ij}^{b_j} (\log L_{ij})^2 \varepsilon_{ij}}{a_j \left(\sum_i L_{ij}^{2b_j} \log L_{ij} \right)^2} \right| < 1.15 \left| \frac{\sum_i L_{ij}^{b_j} (\log L_{ij})^2 \varepsilon_{ij}}{a_j \bar{L}_j^{b_j} \sum_i L_{ij}^{b_j} (\log L_{ij})^2} \right|.$$

Since the a_j 's and b_j 's are MLE's, the following property holds from (1):

$$\overline{OW}_j = a_j \bar{L}_j^{b_j}.$$

Thus, there is a $k > 0$, in fact, $k > 100$ for our data such that

$$a_j \bar{L}_j^{b_j} \geq k \quad \forall j = 1, 2, 3, \dots$$

It suffices to prove that, for all j ,

$$\left| \frac{\sum_i L_{ij}^{b_j} (\log L_{ij})^2 \varepsilon_{ij}}{\sum_i L_{ij}^{b_j} (\log L_{ij})^2} \right| < 2 \times 10^{-1}.$$

Solve the linear regression problem

$$(\log L_{ij})^2 = \alpha_j + \beta_j \log L_{ij} + \eta_{ij}$$

Since

$$\sum_i L_{ij}^{b_j} \log L_{ij} \varepsilon_{ij} = \sum_i L_{ij}^{b_j} \varepsilon_{ij} = 0,$$

and using the Cauchy-Schwarz inequality, it follows that

$$\begin{aligned} \left| \frac{\sum_i L_{ij}^{b_j} (\log L_{ij})^2 \varepsilon_{ij}}{\sum_i L_{ij}^{b_j} (\log L_{ij})^2} \right| &= \left| \frac{\sum_i L_{ij}^{b_j} \eta_{ij} \varepsilon_{ij}}{\sum_i L_{ij}^{b_j} (\log L_{ij})^2} \right| \\ &\leq \frac{\max_i L_{ij}^{b_j} \sqrt{\sum_i \eta_{ij}^2} \sqrt{\sum_i \varepsilon_{ij}^2}}{\sum_i L_{ij}^{b_j} (\log L_{ij})^2} \\ &< 2 \times 10^{-1}. \end{aligned}$$

This proves the result, viz. that the parameters a_j and b_j are pairwise highly correlated for all j .

Conjecture 2:

Given that the a_j 's and b_j 's are pairwise highly correlated for all j (from Conjecture 1), then $-\log a_j$ and b_j are approximately linearly related within a maximum of 14% error tolerance.

Proof:

We prove that $\log \overline{OW}_{.j}$, and thus $\log \left(a_j \overline{L}_{.j}^{b_j} \right)$ is constant up to 12% error tolerance for the given data. We prove that $\overline{L}_{.j}^{b_j}$ is proportional to $\overline{L}_{.j}^{b_j}$ up to 2% error tolerance, and thus $\log a_j + b_j \log \overline{L}_{.j}$ is constant up to 14% error tolerance for the given data.

Proof that $\overline{L}_{.j}^{b_j}$ and $\overline{L}_{.j}^{b_j}$ are approximately proportional:

Fit the following transformed equation by least squares linear regression:

$$OW_{ij}^{1/b_j} = \alpha_j^{1/b_j} L_{ij} + \zeta_{ij}, \tag{3}$$

where $\zeta_j \sim N(0, \sigma^2)$.

For $b_j > 1$, the bias in otolith weight for (1) using (3) is greater than zero and at most the multiplicative bias generated by the logarithmic transformation, viz.

$$\exp \left(\text{Var} \left[\log OW_{ij} - \tilde{\alpha}_j - \tilde{\beta}_j \log L_{ij} \right] / 2 \right), \text{ viz. 2\% from the first property originally given.}$$