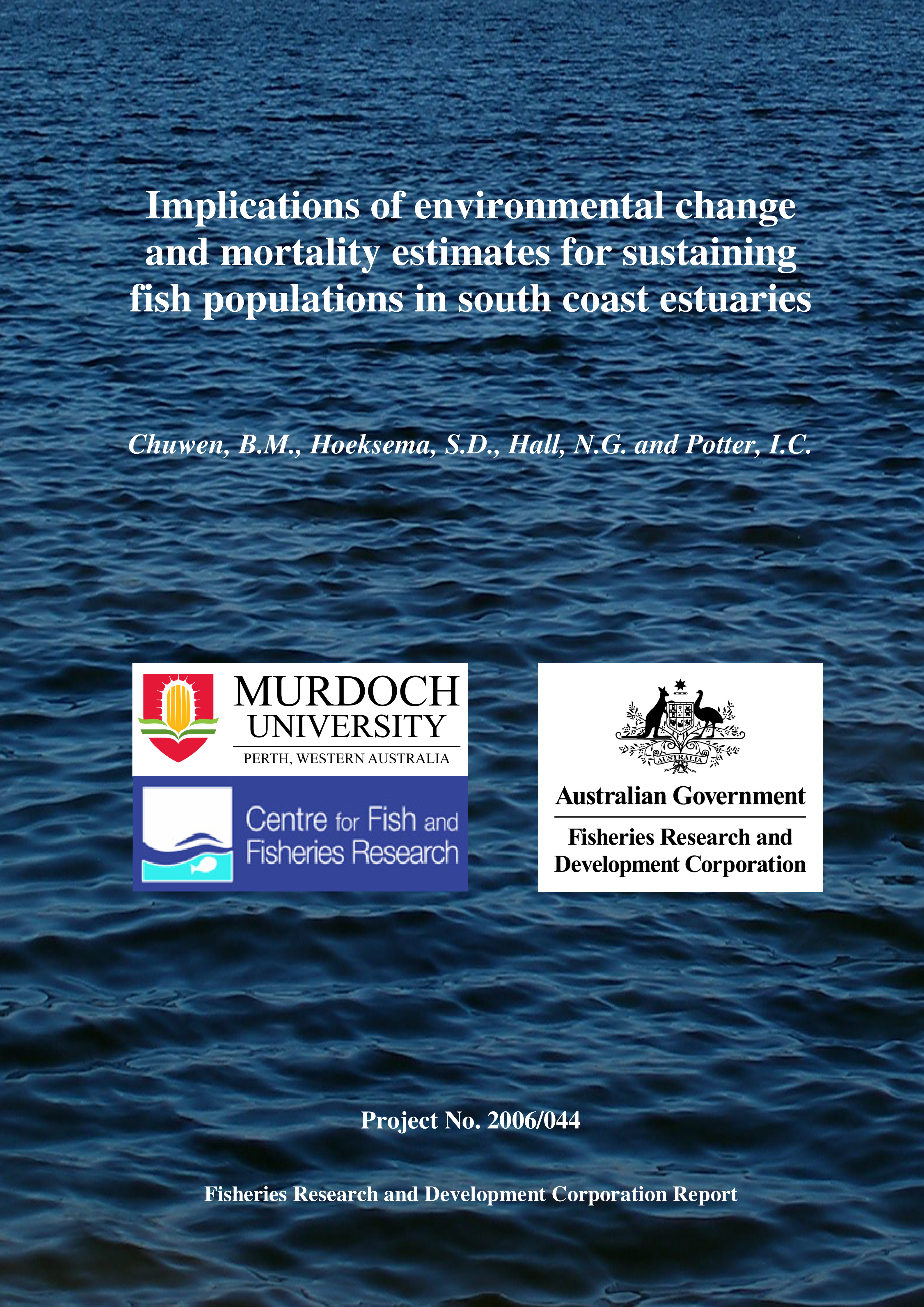

Implications of environmental change and mortality estimates for sustaining fish populations in south coast estuaries

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Centre for Fish and Fisheries Research

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2006/044 Implications of environmental change and mortality estimates for sustaining fish populations in south coast estuaries

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OBJECTIVES

1. Elucidate how environmental change influences the recruitment success of *Acanthopagrus butcheri* and *Cnidoglanis macrocephalus* in south coast estuaries.
2. Determine the current age and size compositions, growth and total mortality of *A. butcheri* and *C. macrocephalus* in four estuaries.
3. Compare the above biological characteristics for *A. butcheri* and *C. macrocephalus* with those derived for these two species in two estuaries 10 and 20 years ago.
4. Determine the implications of the results of the study for management and provide that advice to managers in the Department of Fisheries and the Department of Water.

NON TECHNICAL SUMMARY

OUTCOMES ACHIEVED TO DATE

Managers, scientists and fishers now have an understanding of the implications of the age and size compositions, growth and total mortality of Black Bream and Estuary Cobbler and the current status of the stocks of those species in south coast estuaries. In particular, our fisheries-independent data has shown that the abundance of Estuary Cobbler in Wilson Inlet, which contributes by far the most of any estuary, to the commercial catches of this species, has declined markedly over the last 20 years. The outcomes of this project will assist in the development of plans aimed at sustaining the commercial and/or recreational fisheries for Black Bream and Estuary Cobbler and maintaining the environments of estuaries on the south coast of Western Australia. Such management plans can now be based on sound fisheries-independent data on the biology and contemporary status of those two species and knowledge of their relationships with the environment. Furthermore, managers can now be confident that the closure of certain areas within estuaries is an effective tool to protect the stocks of Estuary Cobbler in those estuaries. In addition, the implications of hypersalinity for the stocks of Black Bream within estuaries are now well understood by fishery managers and local communities.

A wide understanding by fishers and members of local communities of the significance and benefits of the study has been created through their strong engagement with the research team during the course of the study. In addition, through their involvement in the study, two honours and a PhD student have been trained in contemporary techniques in fisheries science and population and community ecology.

The environmental characteristics and the biology of the Black Bream and Estuary Cobbler have been studied in eight estuaries of varying type. The eight estuaries were the permanently-open Oyster Harbour, the seasonally-open Broke, Irwin and Wilson inlets and the normally-closed Wellstead Estuary and Hamersley, Culham and Stokes inlets. In the narrow riverine regions of permanently and seasonally-open estuaries, where Black Bream mainly live, the salinity was greater in the bottom waters than in the surface waters, whereas the reverse was true for oxygen concentrations. In contrast, such stratification was not found in the basins. The extent of seasonal fluctuations in environmental conditions differed markedly among the three seasonally-open estuaries and even between years in one of those systems. Those differences reflected variations in the relationship between the volume of river discharge, which is determined by a combination of the amount of local rainfall, catchment size and extent of clearing of native vegetation, and the amount of intrusion by marine waters, which is largely controlled by the size and duration of the opening of the estuary mouth. The average seasonal salinities in the basins of three of the normally-closed estuaries increased over three years of very low rainfall to nearly twice that of sea water in the deepest estuary (Stokes Inlet) to about four times that of sea water in Hamersley Inlet and to more than eight times that of sea water in the shallowest estuary (Culham Inlet).

On the basis of the number of annually-formed growth zones in their otoliths (ear bones), the maximum ages of Black Bream were found to range downwards from 13 to 15 years in Wilson Inlet, Wellstead Estuary, Culham Inlet and Stokes Inlet, to 9 years in Oyster Harbour and to only 5 years in Irwin Inlet and Hamersley Inlet. The growth of Black Bream varied markedly among estuaries, presumably reflecting differences in the density of Black Bream, quality/quantity of food and/or salinity regime. The width of the annual growth zones in otoliths varied among years, particularly in Stokes Inlet, which is considered to reflect differences in the annual growth of fish. Growth was greatest in years of relatively high rainfall and thus when salinities would have been at the lower end of the range in the years for which annual growth zones were measured. Those environmental conditions are thus regarded as optimal for the growth of Black Bream.

The trends exhibited throughout the year by gonad development demonstrated that Black Bream spawns mainly in spring in estuaries on the south coast of Western Australia. The “average” lengths of Black Bream at maturity in those four estuaries, from which substantial numbers of animals were obtained, were similar, with those for females ranging only from 146-161 mm. While no fish matured at the end of their first year of life in those estuaries, the majority of fish did mature at the end of their second year of life (73-100%). The vast majority or all fish were mature by 200 mm, which is well below the minimum legal length (MLL) of 250 mm for the retention of this species in Western Australia, thus providing the potential for all fish that survive to reach the MLL to reproduce before being legally retained.

Recruitment of Black Bream varied markedly among years and estuaries. Recruitment in the seasonally-open Wilson Inlet, in which the salinity never exceeded that of natural sea water, was greatest in years of relatively low rainfall and thus presumably also of relatively elevated salinities and reduced stratification and associated deoxygenation of the bottom water layer in the rivers. Although massive mortalities of *A. butcheri* in two of the normally-closed estuaries prevented comparisons across this estuary type, strong recruitment in these estuaries is apparently related to years of relatively high rainfall and thus probably to salinities less than that of full strength sea water. There was evidence that total mortality of Black Bream was slightly higher in estuaries with the greatest fishing pressure. The results emphasise that Black Bream is highly plastic in its biological characteristics, particularly in terms of growth and reproduction, and this helps account for the great success of this species in a broad range of estuarine environments.

Estuary Cobbler was most abundant in Irwin and Wilson inlets followed by Oyster Harbour and Wellstead Estuary, while only two individuals were caught in Stokes Inlet and

none was taken in the Broke, Hamersley or Culham inlets. Gill net catch rates in estuaries in which this species was relatively abundant were greatest in summer and least in winter. Growth of Estuary Cobbler was greatest in Irwin Inlet and least in Oyster Harbour, apparently reflecting differences in environmental characteristics, such as the amount of seagrass and its species composition and density. The “average” length of female Estuary Cobbler at first maturity was greatest in Irwin Inlet and least in Wilson Inlet, reflecting differences in growth. The estimate of fishing mortality, which took into account differences in annual recruitment strengths, was far higher in Wilson Inlet (1.47 year^{-1}) than in Irwin Inlet (0.77 year^{-1}).

In Wilson Inlet, the catch rates and lengths of Estuary Cobbler were greater in the area closed to commercial fishing at the seaward end of the estuary than in the area open to fishing, strongly indicating that the closed area acts as a useful refugium for this species. The catch rates of Estuary Cobbler in Wilson Inlet were far lower during the present study (2005-08) than they were during a comparable study in 1987-89. Furthermore, the age structure of the gill net catches has shifted from predominantly 2+ and 3+ year old individuals in 1987-89 to largely 2+ individuals in 2005-08, with very few fish older than 3+ years of age being recorded during that most recent period. In addition, the length and age at maturity both decreased between the two periods. Estimates of total mortality for Estuary Cobbler in Wilson Inlet increased from 0.92 year^{-1} in 1987-89 to 1.82 year^{-1} in 2005-08. A combination of reduced catch rates, changes in age structure, reduced length and age at maturity and increased fishing mortality of the commercially important Estuary Cobbler in Wilson Inlet between 1987-89 and 2005-08 are consistent with the view that this species has been heavily exploited in that system.

KEYWORDS: environment; fishing pressure; catch rates; growth; reproduction; recruitment; fishing mortality; total mortality; *Acanthopagrus butcheri*; *Cnidoglanis macrocephalus*; south coast Western Australian estuaries.

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1. General introduction

1.1. Background

Estuaries along the microtidal and extensive south-western coast of Australia frequently contain substantial commercial and/or recreational fisheries (Lenanton and Potter, 1987). These systems typically comprise a short and narrow entrance channel, which opens into a large basin area that is fed by one or more main tributaries, the lower reaches of which are saline and constitute the upper estuary (Lenanton and Hodgkin, 1985; Potter *et al.*, 1990; Hodgkin and Hesp, 1998; Potter *et al.*, 2010). The larger estuaries on the lower west coast remain permanently open to the ocean, while most of those on the south coast are either seasonally or normally closed through the formation of a sand bar at their mouths during periods of low riverine discharge (Hodgkin and Hesp, 1998; Chuwen *et al.*, 2009a).

Substantial numbers of certain marine fish species, and particularly of their juveniles, enter and occupy for variable periods the permanently and seasonally-open estuaries of south-western Australia (Potter and Hyndes, 1999), as is typically the case with comparable microtidal estuaries in southern Africa (Whitfield, 1999) and macrotidal, permanently-open estuaries in temperate regions of the northern hemisphere (Haedrich, 1983; Elliott and Dewailly, 1995). The estuaries of south-western Australia also contain, however, several fish species that are able to complete the whole of their life cycles within these systems (Potter and Hyndes, 1999). Although some of these species are found only in estuaries, *e.g. Acanthopagrus butcheri*, others are also represented by separate marine populations, *e.g. Cnidoglanis macrocephalus* (Ayvazian *et al.*, 1994; Potter and Hyndes, 1999). While several of these species are small, others attain a substantial size and, because of their excellent eating and/or angling qualities, are targeted by commercial and/or recreational fishers (Lenanton and Potter, 1987).

The clearing of land for agriculture in the catchments of estuaries in the eastern region of the south coast of Western Australia, *e.g.* Wellstead Estuary, Culham Inlet and Stokes Inlet, allied with a decline in winter rainfall, have led to these estuaries becoming markedly hypersaline, eutrophic and sometimes anoxic (Young and Potter 2002; Hoeksema *et al.*, 2006a). The environment of several south coast estuaries has also been modified, to a certain degree, by the artificial breaching of the sand bars at their mouths, which results in exchange between the waters of the estuary and the ocean (*e.g.* Brearley, 2005). The marked environmental changes that have occurred in many south coast estuaries during recent years, combined with an increase in the total amount of fishing, led commercial and recreational fishers and environmental managers to express concern regarding the status of the fisheries in these systems.

This project was developed in consultation with Mr Chris Gunby (Department of Environment), Dr Rod Lenanton (Department of Fisheries, WA), Mr Frank Prokop (RecFishWest) and Mr Richard Stevens (Chair, Black Bream Working Group; Western Australian Fishing Industry Council) in response to increasing concerns about the sustainability of the South Coast Estuarine Fishery and the recreational fisheries that operate in south coast estuaries. Other groups which have supported this proposal include the South Coast Licensed Fisherman's Association, Wilson Inlet Management Advisory Group, Denmark Shire Council and Wilson Inlet Sustainable Environment Group.

1.2. Need

The highly deleterious impact of environmental changes on commercially and recreationally important fish species is becoming of greatly increasing concern to stakeholders and fishery managers. The managers of fisheries on the south coast of Western Australia need to know 1) the extent to which environmental changes can affect the recruitment success of key targeted species and 2) the current estimated levels of total mortality of those species and how they might have changed in the last 10-20 years. Such information is essential for facilitating the development by resource managers of appropriate strategies for managing the fisheries in those estuaries, and is particularly important because those estuaries have undergone radical environmental change and the amount of fishing has increased and fisheries regulations have been modified. Knowledge of the impacts of marked environmental change are necessary to develop strategies that can accommodate the changes that must inevitably be occurring in carrying capacity, recruitment, growth and mortality of fish species in these estuaries.

1.3. Objectives

1. Elucidate how environmental change influences the recruitment success of *Acanthopagrus butcheri* and *Cnidoglanis macrocephalus* in south coast estuaries.
2. Determine the current age and size compositions, growth and total mortality of *A. butcheri* and *C. macrocephalus* in four estuaries.
3. Compare the above biological characteristics for *A. butcheri* and *C. macrocephalus* with those derived for these two species in two estuaries 10 and 20 years ago.
4. Determine the implications of the results of the study for management and provide that advice both in writing and verbally to managers in the Department of Fisheries and the Department of Water.

2. General materials and methods

2.1. Physical characteristics of the estuaries

The area of the basins of the eight estuaries range from 2.3-2.5 km^2 in Hamersley Inlet and Wellstead Estuary to between 10 and 16 km^2 in the Irwin, Culham and Stokes inlets and Oyster Harbour to 48 km^2 in both the Broke and Wilson inlets (Table 2.1). The catchment areas and the percentages of those catchments that have been cleared for particularly agriculture also vary substantially and, together with rainfall differences, account for the marked extent to which the discharge from the tributaries of the eight estuaries differ (Table 2.1).

Table 2.1. Physical characteristics of the Broke, Irwin and Wilson inlets, Oyster Harbour, Wellstead Estuary and Hamersley, Culham and Stokes inlets and their catchments.

Estuary	Classification	Basin surface area (km^2)	Catchment area (km^2)	Catchment percentage cleared (%)	Mean annual rainfall (mm)	Annual rainfall range (mm)	Mean annual discharge (m^3 x 10^6)
Broke Inlet	Seasonally open	48.0 [a]	700 [b]	9 [b]	1035 [d]	842-1357 [d]	172.0 [b]
Irwin Inlet	Seasonally open	10.0 [a]	2300 [b]	39 [b]	623 [d]	467-863 [d]	163.0 [b]
Wilson Inlet	Seasonally open	48.0 [a]	3400 [b]	60 [b]	765 [d]	577-975 [d]	161.0 [b]
Oyster Harbour	Permanently open	15.6 [a]	3000 [b]	72 [b]	691 [d]	496-895 [d]	108.0 [b]
Wellstead Estuary	Normally closed	2.5 [a]	720 [b]	75 [b]	494 [d]	239-741 [d]	14.0 [b]
Hamersley Inlet	Normally closed	2.3 [a]	1270 [c]	37 [c]	451 [d]	214-647 [d]	1.2 [b]
Culham Inlet	Normally closed	11.3 [a]	3780 [c]	50 [c]	388 [d]	219-551 [d]	3.4 [b]
Stokes Inlet	Normally closed	14.0 [a]	5300 [c]	60 [c]	520 [d]	296-769 [d]	11.9 [b]

Sources: [a] Brearley, 2005; [b] Pen, 1999; [c] Western Australian Department of Water, unpublished data; [d] Commonwealth Bureau of Meteorology, unpublished data.

While the catchment of Broke Inlet is relatively small, the mean annual discharge from its three tributaries is substantial, *i.e.* 172 x 10^6 m^3, due to this catchment being located in the highest rainfall region of south-western Australia (Brearley, 2005). As this catchment has not been cleared of native vegetation for agriculture to the same extent as those of the other south coast estuaries, the amount of nutrient runoff and thus primary productivity in its basin are also less (Hodgkin and Lenanton, 1981; Brearley, 2005). The mouth of Broke Inlet is artificially breached in winter or early spring when, as a result of the highly seasonal rainfall, the water level in the estuary rises to approximately 1.8 m above the Australian Height Datum (AHD).

Although the catchments of Irwin and Wilson inlets are located in a far lower rainfall region than that of the likewise seasonally-open Broke Inlet, the mean annual discharges into the former two estuaries approach that of the latter estuary (Table 2.1). This is due to the former two catchments being far larger and having been more extensively cleared. The basins of Irwin Inlet and of the much larger Wilson Inlet each receive a considerable nutrient input (*e.g.* Radke *et al.*, 2004), which accounts for their containing extensive areas of the seagrass *Ruppia megacarpa* (*e.g.* Carruthers *et al.*, 1999) and substantial growths of epiphytic algae and phytoplankton (Hodgkin and Lenanton, 1981; Dudley *et al.*, 2001; Twomey and Thompson, 2001). Although the sand bars at the mouths of the Irwin and Wilson inlets are breached artificially in the winter or early spring of most years, these breachings occur at a lower water level than in Broke Inlet, *i.e.* *ca* 1 vs 1.8 m above AHD.

The catchment of the permanently-open Oyster Harbour is large and extensively cleared for agriculture, *i.e.* over 70% (Table 2.1). While a large amount of nutrients thus enter the basin of this estuary via its two large tributaries, much of the annual discharge of *ca* 108 x 10^6 m^3 occurs immediately after periods of particularly heavy rainfall and leads to substantial amounts of nutrients being flushed out to sea (Brearley, 2005). The seagrass community in the basin consists of marine species, such as those belonging to *Posidonia* (Carruthers *et al.*, 2007).

The Wellstead Estuary and Hamersley, Culham and Stokes inlets, which are all normally closed to the ocean, are located in a low rainfall region of the south coast (Table 2.1). While precipitation typically occurs in this region predominantly during winter and early spring, extreme cyclonic summer storms do occur in some years and contribute substantially to overall annual rainfall and thus also to discharge in those years. However, as such storms can be highly localised and the extent of clearing among catchments varies, the annual discharges from the rivers into the basins of the Wellstead Estuary and Hamersley, Culham and Stokes inlets differ, and markedly so in some years. Substantial amounts of nutrients from agricultural land within the catchments enter the rivers and subsequently the basins of these estuaries (*e.g.* Radke *et al*., 2004). Culham Inlet is unique in that the lower end of its basin contains a small (ca 0.025 km^2) but relatively deep, *i.e. ca* 3 m, artificial lagoonal-like body of water which is essentially isolated from the main body of the basin by a roadway across its seaward end (Hodgkin, 1997).

2.2. Sampling regime

A number of sites in nearshore, shallow and offshore, deeper waters of the basins of the seasonally-open Broke, Irwin and Wilson inlets, the permanently-open Oyster Harbour and the normally-closed Wellstead Estuary (Fig. 2.1) were sampled by seine and gill net, respectively, in the last month of each season between summer 2005/06 and spring 2007. Additional samples of *Acanthopagrus butcheri*, which were collected from the normally-closed Hamersley, Culham and Stokes inlets (Fig. 2.1) in the middle of each season between summer 2001/02 and spring 2004, have also been included in this analysis (see Hoeksema *et al.*, 2006b). Gill netting was also used in the same seasons to sample sites in offshore, deeper waters of the riverine, saline reaches of the above eight estuaries. Seine netting was also employed to sample sites in nearshore, shallow waters of the main tributary of Oyster Harbour, Wellstead Estuary and Culham and Hamersley inlets (Fig. 2.1), the four tributaries that contained nearshore, shallow habitats that could be effectively sampled with a seine net. In the case of Culham Inlet, sampling with rod and line was also conducted in an upstream riverine pool during the same seasons in which the estuary was sampled. Additional gill and seine net samples were collected in winter and spring 2005 from the same sites in the Broke, Irwin and Wilson inlets, and in spring 2005 from the Wellstead Estuary.

The seine net that was used to sample nearshore, shallow waters was 21.5 m long and consisted of two 10 m long wings (6 m of 9 mm mesh and 4 m of 3 mm mesh) and a 1.5 m bunt of 3 mm mesh. The net, which was laid parallel to the shore and then hauled onto the beach, fished to a depth of 1.5 m and swept an area of *ca* 116 m^2. Fishes in the offshore, deeper waters were sampled in the Broke, Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary using sunken composite multifilament gill nets, comprising 20 m panels, each with a height of 2 m, but containing a different stretched mesh size, *i.e.* either 35, 51, 63,

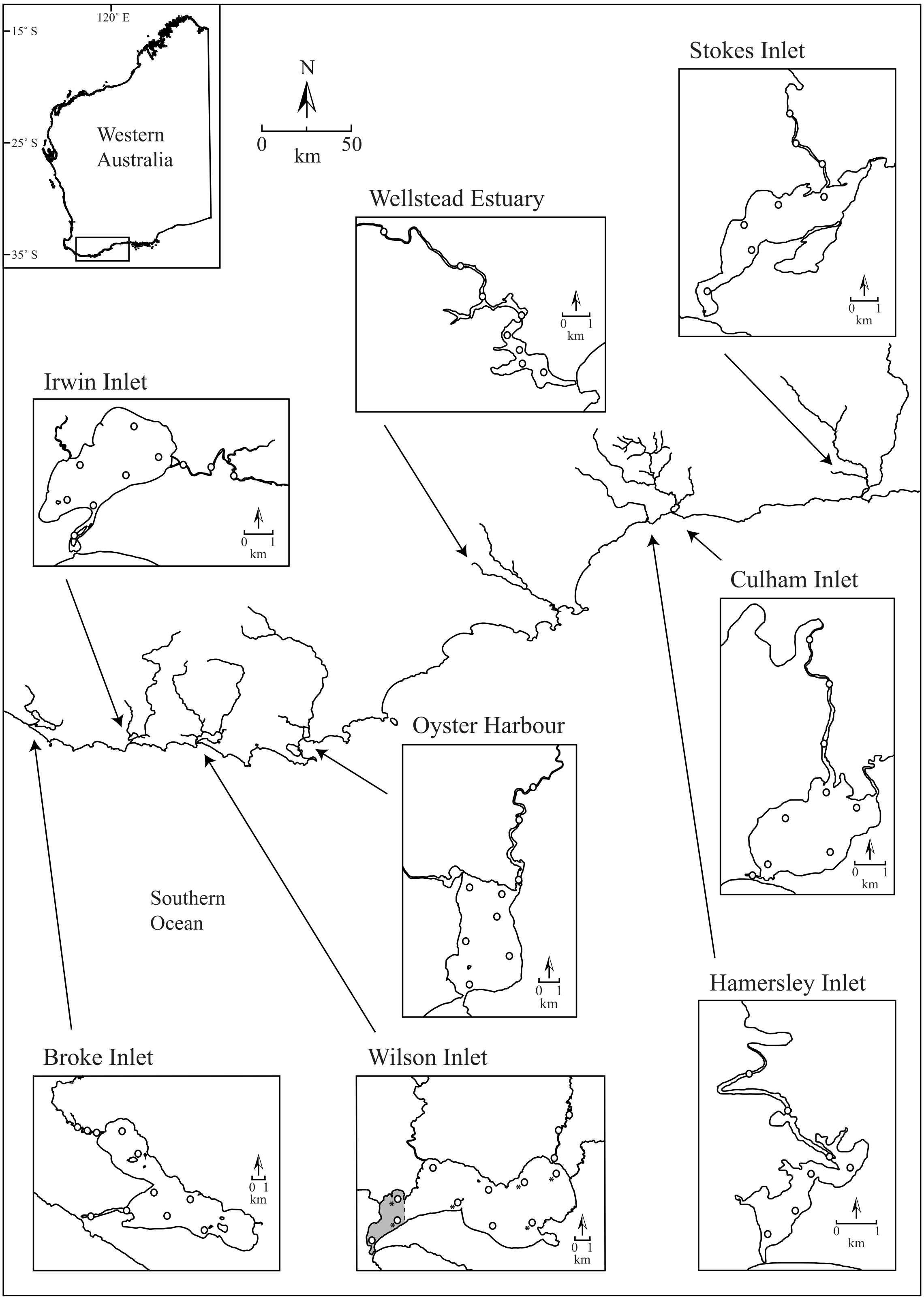

Fig. 2.1. Map showing the location of the estuaries in which *Acanthopagrus butcheri* and *Cnidoglanis macrocephalus* were targeted on the south coast of Western Australia and the location of the sites sampled in each estuary in 2005-08 (○). * denotes sites that were sampled in Wilson Inlet in both 1987-88 and 2005-08. The shaded area in Wilson Inlet denotes the area closed to commercial fishing.

76, 89, 102 or 115 mm and the same nets, excluding the 115 mm mesh panel, were used in the Hamersley, Culham and Stokes inlets. Gill nets were set in the Broke, Irwin and Wilson inlets and Oyster Harbour at dusk and retrieved *ca* 12 h later around dawn, but were set for only 1 h from dusk in the Wellstead Estuary and Hamersley, Culham and Stokes inlets as preliminary sampling demonstrated that gill nets set overnight in these estuaries would, at times, yield unacceptably high catches of fish, and in particular, of the iconic Black Bream. Fishes were euthanased in an ice slurry immediately after capture.

2.3. Growth, length and age compositions

Each fish was measured to the nearest 1 mm (total length, TL) and weighed to the nearest 0.1 g. The two sagittal otoliths of each individual *Acanthopagrus butcheri* and the two lapillus otoliths of each *Cnidoglanis macrocephalus* were removed, cleaned, dried and stored. The translucent zones in whole otoliths were recorded for *C. macrocephalus*, whereas the opaque zones were recorded in the whole otoliths of small (< 100 mm TL) *A. butcheri*. Opaque zones were likewise counted in sectioned otoliths of larger *A. butcheri*. Note that previous studies have demonstrated that it was appropriate to use whole otoliths for such counts in the case of small *A. butcheri* (Sarre and Potter, 2000). In contrast, previous studies have shown that it was necessary to section the otoliths of larger *A. butcheri* to reveal clearly all of their opaque zones (Sarre and Potter, 2000). When whole otoliths were used for ageing, one otolith from each individual was placed in a black dish, covered with methyl salicylate, examined under reflected light using a dissecting microscope and the number of its opaque or translucent zones recorded. For sectioning, otoliths were mounted in clear epoxy resin and cut into *ca* 300-500 μm sections through their primordia and at right angles to the longest axis of the otolith using a low-speed diamond saw (Buehler). The resultant sections were cleaned, mounted on glass slides and viewed under reflected light against a black background using a Leica MZ7.5 dissecting microscope.

Previous work has validated that a single opaque zone is laid down annually in the otoliths of *A. butcheri* by Sarre and Potter (2000) and *C. macrocephalus* by Laurenson *et al.*

(1994). Therefore, the number of opaque zones in either the whole or sectioned otoliths of *A. butcheri* and the translucent zones of those of *C. macrocephalus* (all subsequently termed annual zones) can be used for ageing these two species.

The age of each individual fish was estimated using 1) the date of capture, 2) the number of annual zones in the otolith examined, 3) the date when, on average, newly-formed annuli typically become delineated from the otolith periphery, and 4) a hatching date ("birth date"), representing the mid-point of the spawning period. The number of annual zones for fish caught within two months prior to the date assigned for delineation of the annual zone were adjusted downwards by 1 year if their otoliths possessed a narrow translucent edge (in the case of *A. butcheri* for which the annual zone is opaque) or if they possessed a narrow opaque zone (in the case of *C. macrocephalus*), whereas those caught after that date were adjusted upwards a year if their otoliths possessed an opaque margin, in the case of the former species, or a translucent margin, in the case of *C. macrocephalus.*

The age, *t*, of each individual was calculated as:

$$t = \left\{ \frac{z + \left[\dfrac{m_c + (m_d - m_b) + 1}{12}\right] + \dfrac{d_c}{365.25}}{z + \left[\dfrac{m_c + (m_d - m_b) - 11}{12}\right] + \dfrac{d_c}{365.25}} \middle| \begin{array}{l} if\ m_c < m_d \\ if\ m_c \geq m_d \end{array} \right\}$$

, where z is the number of annual zones, m_c is the month of capture, d_c is the day of the month of capture, m_d is the month of delineation and m_b is the birth month (where the date assigned to the time of delineation and birth date corresponded to the beginning of that month).

The von Bertalanffy growth parameters and their 95% confidence intervals (CI) were estimated from the lengths at age of fish at the date of capture and employing the nonlinear regression procedure in SPSS^{TM}. The von Bertalanffy growth equation for describing the growth of each sex is $L_t = L_\infty(1 - e^{(-k(t-t_0))})$, where L_t is the length (mm) at age t (years), L_∞ is the asymptotic length (mm) predicted by the equation, k is the growth coefficient (year^{-1}) and t_0 is the hypothetical age (years) at which fish have zero length.

The von Bertalanffy growth equations for the females and males of each species in each estuary for which there were sufficient sample sizes, were compared using a likelihood-ratio test (Cerrato, 1990). The hypothesis that the lengths at age of each sex in an estuary could be described by a single growth curve was rejected at the $\alpha = 0.05$ level of significance if the test statistic, calculated as twice the difference between the log-likelihoods obtained by fitting a common growth curve for both sexes and by fitting a separate growth curves for each sex, exceeded $\chi_2^{\alpha}(q)$, where q is the difference between the numbers of parameters in the two approaches. The log-likelihood, λ, for each curve, ignoring constants, was calculated as $\lambda = -\frac{n}{2}\log_e \frac{ss}{n}$, where n refers to sample size and ss refers to the sum of squared residuals between the observed and expected lengths at age. Likelihood-ratio tests were also used to compare the growth curves for each sex in the populations of each pair of estuaries.

Likelihood-ratio tests were used to compare the growth of each sex of *A. butcheri* during the current study (2005-2007) with that of the corresponding sex of this sparid in the same estuary in 1995-96. This comparison used the raw data collected by Sarre and Potter (2000) for this species in 1995-96. Such tests were also used to compare the growth of both sexes of *C. macrocephalus* in Wilson Inlet during the current study with that determined for this species in that estuary in 1987-1989. This latter comparison used the raw data collected by Laurenson *et al.* (1994). As the likelihood-ratio test identifies very small differences in growth as being statistically significant when applied to data with large sample sizes, emphasis was placed on the level of difference in growth that was likely to be of biological significance. Such differences between growth curves were considered to exist when any of the predicted lengths at integer ages ≥ 1 year differed by > 5% in the range of the dominant ages within the data.

2.4. Reproductive variables

The gonads of each individual *Acanthopagrus butcheri* and *Cnidoglanis macrocephalus* were identified macroscopically as either ovaries or testes, or as indeterminate in the case of juvenile fish with very small gonads that could not be sexed, and those that could be sexed were removed and weighed to the nearest 0.01 g. On the basis of their macroscopic characteristics and following the scheme of Laevastu (1965), the gonads of each fish were allocated to one of the following eight stages of gonadal development.
I = immature, II = resting adult, III = developing, IV = maturing, V = prespawning,
VI = spawning, VII = spent, and VIII = recovering spent (see Table 2.2). Fish that possessed gonads at stages III-VIII during the spawning period were classified as mature.

Table 2.2. Macroscopic appearances of stages in the development of *Acanthopagrus butcheri* gonads and corresponding histological characteristics of ovaries. Stages largely follow the scheme of Laevastu (1965) and histological terminology is modified from Wallace and Selman (1981).

Stage	Macroscopic appearance	Histological characteristics of ovaries
I Virgin	Gonads very small. Ovary transparent, no oocytes visible. Testes grey, strand-like.	Oocytes highly organised along ovarian lamellae, chromatin nucleolar and early perinucleolar oocytes (small previtellogenic oocytes) present.
II Immature	Gonads larger than stage I. Ovaries translucent, testes white and ribbon-like.	Similar to stage I with late perinucleolar oocytes present. These oocytes are found in all subsequent stages.
III Developing	Ovaries becoming opaque, orange to red in colour with whitish oocytes visible. Testes white. Occupy up to 1/3 of the ventral cavity.	Cortical alveolar oocytes abundant.
IV Maturing	Ovaries orange with blood capillaries and yolk granule oocytes visible. No milt appears when pressure is applied to the trunk of males. Occupy 1/3 to 1/2 of the ventral cavity.	Cortical alveolar and yolk granule oocytes abundant.
V Prespawning	Gonads 1/3 to filling ventral cavity. Milt appears with firm pressure from testes. Ovaries large with numerous capillaries, no hydrated oocytes visible.	Many yolk granule oocytes tightly grouped. Few cortical alveolar oocytes present.
VI Spawning	Hydrated oocytes visible through ovary wall. Milt appears with slight pressure. 1/3 to filling the ventral cavity.	Hydrated oocytes, migrating nuclei oocytes and/or postovulatory follicles present.
VII Spent	Gonads smaller than stage V or VI. Ovaries and testes flaccid though not fully empty.	Large amounts of connective tissue. Remnant yolk granule and cortical alveoli oocytes generally undergoing atresia.
VIII Recovering spent	Testes and ovaries red to brown, flaccid with scarring visible at ends of ovaries. No oocytes are visible in ovaries.	Oocytes unorganised, with extensive connective tissue and large “empty” areas.

In each month, the gonads of up to 20 females of each species, covering a wide range of lengths and the full suite of gonadal stages observed in that month, were retained and prepared for histological examination. For this purpose, a portion of the mid-region of one ovarian lobe was placed in Bouin's fixative for *ca* 48 h and dehydrated in a series of increasing concentrations of ethanol. The ovarian portions were then embedded in paraffin wax, cut transversely into 6 µm sections and stained with Mallory's trichrome. The stages in oocyte development in each section were then determined by examination using a Leica MZ7.5 dissecting microscope to validate that the stages in gonadal development assigned to each ovary on the basis of its macroscopic appearance were appropriate (see Fig. 2.2, Table 2.2).

Logistic regression analysis was used to fit curves to the probabilities that female and male fish at each length during the spawning period would possess gonads at one of stages III-VIII. The logistic equation describing the probability of an individual possessing mature gonads, P, at length, L, was $P = 1/[1+\exp(-\log_e(19)(L-L_{50})(L_{50}-L_{95}))]$, where L_{50} and L_{95} are the total lengths at which 50 and 95% of the individuals, respectively, would be expected to possess gonads at stages III-VIII. On the basis of its length, the likelihood of the jth fish possessing or not possessing gonads at stages III-VIII was calculated as P_j or $1 - P_j$, respectively. Setting $X_j = 0$ if the jth fish did not possess gonads at such a stage and $X_j = 1$ if it did possess such gonads, the overall log-likelihood, λ was calculated as: $\sum_j \{X_j \log_e P_j + (1-X_j)\log_e[(1-P)_j]\}$. The logistic equation was fitted by maximizing this log-likelihood, using SOLVER in Microsoft Excel$^{\text{TM}}$. The data were randomly resampled and analysed to create 100 sets of bootstrap estimates of the parameters of the logistic equation and of the probabilities of females and males being mature for each of a range of specified lengths. The 95% confidence intervals of the probability of maturity at each specified length were taken as the 2.5 and 97.5 percentiles of the corresponding predicted values resulting from this resampling analysis. The medians of the bootstrap estimates were used as the point estimates of each parameter and of the probability of maturity at each specified length. The logistic regressions describing the length-at-maturity data for each water body were compared using the likelihood-ratio test outlined previously. The results were then considered in the

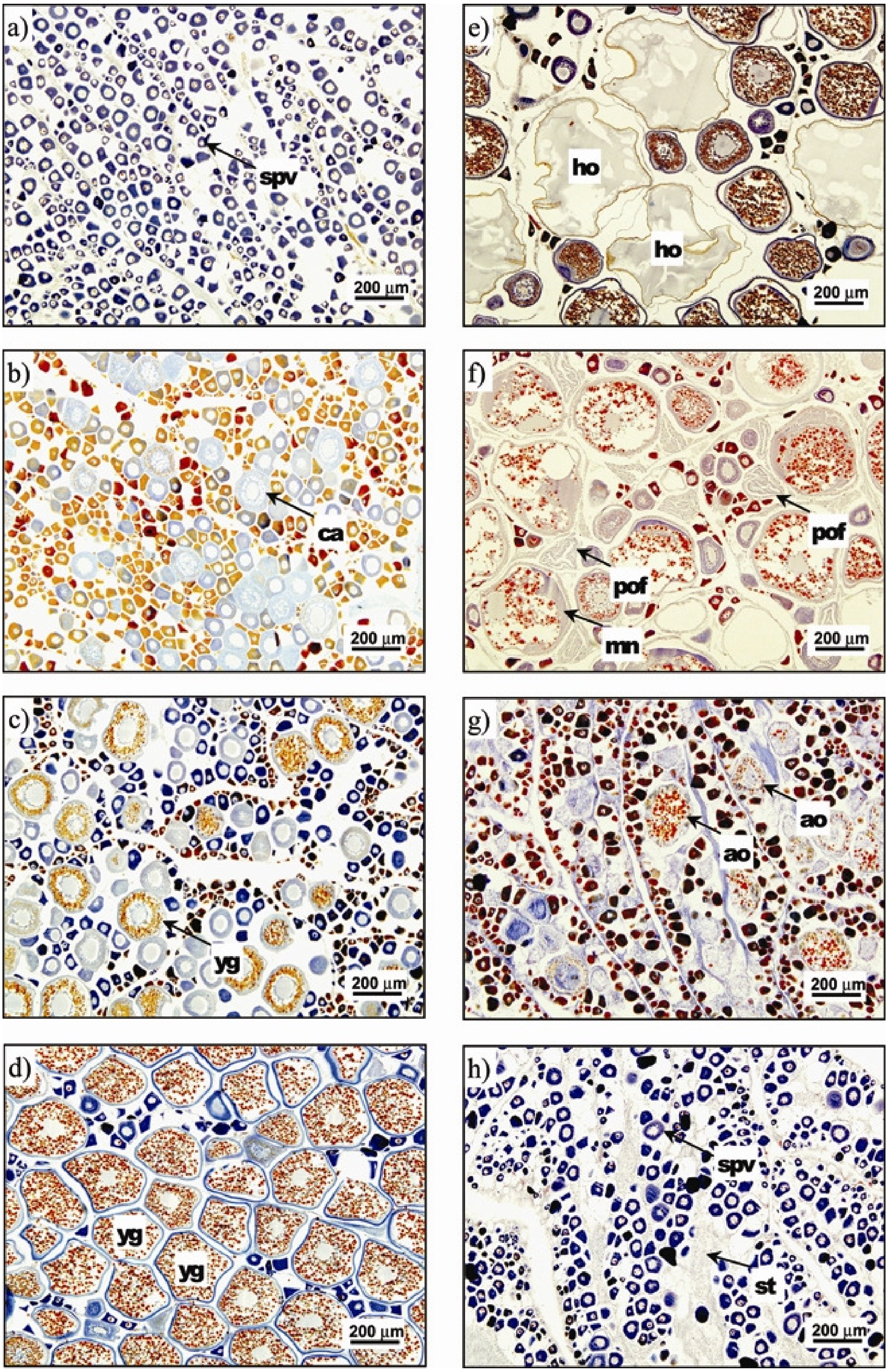

Fig. 2.2. Histological sections of stage (a) II, (b) III, (c) IV, (d) V, (e, f) VI, (g) VII and (h) VIII ovaries of *Acanthopagrus butcheri* possessing small previtellogenic oocytes (spv), cortical alveolar oocytes (ca), yolk-granule oocytes (yg), hydrated oocytes (ho), migrating nuclei oocytes (mn), post-ovulatory follicles (pof), atretic oocytes (ao) and connective tissue (ct).

context of the growth of the juveniles of the two sexes of *A. butcheri* within the different estuaries.

The gonadosomatic index (GSI) was calculated separately for each female and male fish using the equation GSI = *W1/W2* × 100, where *W1* = wet weight of the gonad and *W2* = total wet weight of the fish. The index was calculated using data for fish ≥ the estimated L_{50} at first maturity for females and males.

The timing, duration and peak time of spawning of each species in each estuary were estimated from the trends exhibited by the mean monthly gonadosomatic indices and the monthly prevalences of the various gonad stages.

2.5. Gill net selectivity

Gill net mesh selectivities for *Acanthopagrus butcheri* and *Cnidoglanis macrocephalus*, which were required to eliminate possible selection bias in research samples of age composition, were calculated for the composite gill net using the method described by Kirkwood and Walker (1986). In this method, the functional form of the relationship between selectivity and fish length was assumed to be that of a gamma distribution. Thus, the selectivity for mesh *i* and length class *j* may be calculated as $S_{ij} = \left(\frac{L_j}{\alpha_i \beta_i}\right)^{\alpha_i} \exp\left[\alpha_i - \frac{L_j}{\beta_i}\right]$, where L_j is the midpoint of length class *j* ($j = 1, \cdots, J$), and α_i and β_i are parameters of the gamma distribution that are determined from the mesh size m_i of mesh *i* ($i = 1, \cdots, I$). Kirkwood and Walker (1986) express the parameters of the selectivity curves in terms of two alternative parameters, θ_1 and θ_2, however, where $\alpha_i \beta_i = \theta_1 m_i$ and $\beta_i = -0.5\left(\theta_1 m_i - \left(\theta_1^2 m_i^2 + 4\theta_2\right)^{0.5}\right)$. Thus, given the values of the parameters θ_1 and θ_2, and the mesh size of the gill net m_i, the values of α_i and β_i may be calculated.

The model was fitted to length-composition data for *A. butcheri* and *C. macrocephalus* collected by fishing with composite gill nets using the method described by Kirkwood and Walker (1986). For this calculation, estimates of the relative proportion of fish in the population from length class *j* are calculated as $\mu_j = {\sum_{i=1}^{I} n_{ij}} \Big/ {\sum_{i=1}^{I} S_{ij}}$, where n_{ij} are the number of fish of length class *j* in the sample from mesh *i*. The log-likelihood is calculated as

$LL = \sum_{i=1}^{I} \sum_{j=1}^{J} \left[n_{ij} \log_e \left(\mu_j S_{ij} \right) - \mu_j S_{ij} \right]$. Estimates of the parameters θ_1 and θ_2 were obtained by maximising the value of the log-likelihood, using the Solver procedure within an Excel$^{\text{TM}}$ implementation of the model.

Values of mean selectivity at age, S_α, for the composite seven-panel research gill net were estimated as $S_a = \sum_{i=1}^{I} S_{ia} \Big/ \sum_{a=0}^{A} \sum_{i=1}^{I} S_{ia}$, where S_{ia} is the selectivity of fish with length L_a, the length at the midpoint of the ages within age class *a* ($a = 0, \cdots, A$).

2.6. Recruitment variability and mortality estimates

Estimates of the fishing mortality (*F*) of fully-vulnerable *Acanthopagrus butcheri* and *Cnidoglanis macrocephalus* were determined by fitting a model to the age composition data for the combined sexes of each species in random samples collected from gill net catches in consecutive years. Data from Wilson Inlet, Oyster Harbour and Wellstead Estuary in 2006-07 and Stokes Inlet in 2002-2003 were used in the case of *A. butcheri* and from Irwin Inlet in 2006-2008 and at the same sites in Wilson Inlet in 2006-2008 and 1988-1989 in the case of *C. macrocephalus*. The model assumed a constant known level of natural mortality (*M*), *i.e.* the value of the estimate derived from the maximum age recorded for this species by employing Hoenig's (1983) equation for fish.

It was assumed that the fish with lengths greater than or equal to the MLL, *i.e.* 430 mm for *C. macrocephalus* and 250 mm for *A. butcheri*, are exposed fully to a constant level of fishing mortality, while those with lengths less than the MLL experience no fishing mortality, *i.e.* are affected only by natural mortality. Thus, the age composition of the population within each estuary will reflect this pattern of fishing mortality. However, the age compositions of the populations were assumed to be influenced not only by natural and fishing mortality, but also by interannual variability in recruitment strength. Thus, the number of fish of age *a* in year *t* may be assumed to be represented by $N_{a,t} = \overline{R} \exp\left[R_{t-a} - \frac{\sigma_R^2}{2} \right] \exp\left[- \sum_{j=a_{1_c}}^{a-1} Z_j \right]$, where $\overline{R}$ is the mean level of annual recruitment, R_{t-a} is the deviation in the natural logarithm of annual recruitment for year class $t-a$, σ_R^2 is the variance of the values of R_{t-a}, a_1 is the first age present in the age-composition data,

$Z_a = M + F_a$ is the total mortality experienced by fish of age *a*, and the fishing mortality of fish of this age is zero if the length of fish at the midpoint of the ages in the age class is less than the MLL, otherwise $F_a = F$, the fishing mortality of fully-vulnerable fish. The above equation may be rewritten as $N_{a,y,t} = R_y \exp[-Ma]\exp\left[-F\sum_{j=0}^{a-1} S_j^*\right]$, where S_a^* is the selectivity of the fishing gear used by commercial fishers and it is assumed that $S_a^* = 0$ if the length of fish at the midpoint of the ages in the age class is less than the MLL, otherwise $S_a^* = 1$. Denoting the expected count of fish of age a caught from year class *y* in year *t* by $\hat{c}_{a,y,t}$, we may express this as $\hat{c}_{a,y,t} = I_t S_a N_{a,y,t}$,where I_t is a factor that represents a combination of the catchability of the fully-vulnerable fish and the fishing effort used to collect the sample in year *t*, i.e. the sampling intensity in that year, and S_a is the selectivity of the research fishing gear that was used when collecting the age composition data. Thus, $\log_e \hat{c}_{a,y,t} = \log_e I_t + \log_e S_a + \log_e R_y - Ma - F\sum_{j=0}^{a-1} S_j^*$. Setting $K_a = \sum_{j=0}^{a-1} S_j^*$, we may express this model as $\log_e \hat{c}_{a,y,t} = \log_e I_t + \log_e S_a + \log_e R_y - Ma - FK_a$.

If we assume that *M*, S_a and S_a^* are known, then the observed counts may be considered to be variates from a Poisson model, which, in R (R Development Core Team, 2008), would be expressed as $\text{Count} \sim \text{Year} + \text{YearClass} + \text{Ksuba} + \text{offset}\left(\log_e\left[S_a \exp(-Ma)\right]\right)$, where Year and YearClass are categorical variables and the coefficients relating to Year reflect the relative sampling intensity, those relating to YearClass represent the relative levels of recruitment, and that relating to Ksuba (*i.e.* K_a) is the fishing mortality. There are thus number of years + number of year classes = 2 * number of years + number of age classes – 1 parameters that must be estimated from number of years * number of age classes observations. Since the sampling intensity employed to collect the age composition data for both species in the different years was constant, thus $I_t = 1$, Year may be removed, leaving the Poisson regression model $\text{Count} \sim \text{YearClass} + \text{Ksuba} + \text{offset}\left(\log_e\left[S_a^* \exp(-Ma)\right]\right)$. This was fitted to the age composition data to obtain estimates (and standard errors) of both the relative year class strengths and the fishing mortality for *A. butcheri* and *C. macrocephalus*. Note that each estimate of year class strength represents the abundance of that year class relative to the abundance of a reference year class from the age composition data for the period, thus estimates derived from age composition data for different estuaries and/or periods are not directly comparable.

3. Environmental characteristics

3.1. Introduction

Estuaries are among the most productive of all ecosystems (Schelske and Odum, 1961) and thus provide an abundant food source for the many commercially and recreationally important marine fish species that typically use these systems as nursery areas (West *et al*., 2003; Platell *et al*., 2006; Elliott *et al*., 2007). They are therefore crucial for ensuring the sustainability of a number of marine fisheries in both the northern and southern hemispheres (Pollard, 1981; Kennish, 1990; Whitfield, 1998; Sardiña and Cazorla, 2005). Furthermore, some estuaries contain substantial fisheries that are based on certain marine species and on species that complete their life cycle within estuaries (Lenanton and Potter, 1987; Costa *et al*., 2002; Jaureguizar *et al*., 2003; Jung and Houde, 2003). The productivity of estuaries is also exploited by a number of bird species to replenish their energy reserves as they migrate to and from their breeding grounds (Elliott and McLusky, 2002). In view of their great importance to a range of biota and to fisheries, it is of world-wide concern that estuaries are now considered the most degraded of all temperate marine ecosystems (Jackson *et al*., 2001).

The estuaries of southern Australia and southern Africa, which are located in microtidal regions, are of relatively recent origin, having commenced their formation during the Holocene marine transgression only about 7000 years ago (Hodgkin and Hesp, 1998; Cooper, 2001; Roy *et al*., 2001). They typically consist of a narrow and short entrance channel, a wide central basin area and the saline lower reaches of their tributaries (Day, 1981; Hodgkin and Lenanton, 1981). Although the larger estuaries on the lower west coast of Australia typically remain permanently open, only a few of those on the south coast are continuously connected with the ocean. The vast majority of the other estuaries on the latter coast are separated from the sea for varying periods through the formation of sand bars across their mouths (Hodgkin and Lenanton, 1981; Potter and Hyndes, 1999). The sand bars of some of these estuaries are naturally or artificially breached in the winter or spring of each year as the volume of water in the estuary increases as a result of the heavy rainfall that occurs in the region during those seasons (*e.g.* Ranasinghe and Pattiaratchi, 1999). However, in contrast to these seasonally-open systems, the volume of water in the estuaries in the drier regions of the south coast does not frequently increase to the levels required to breach the sand bar at their mouths, and consequently such estuaries remain normally closed to the ocean (Hodgkin and Lenanton, 1981). Cooper (2001) considered that, as some examples of this type of system in southern Africa become hypersaline and even dry out for periods and thus become inhospitable for biota, they should not be considered to be estuaries. Such systems are however traditionally regarded as estuaries by biologists (*e.g.* Day, 1981; Whitfield, 1999; Potter *et al.*, 2010).

As elsewhere in the world, the estuaries of south-western Australia are becoming increasingly influenced by the effects of anthropogenic activities. For example, the clearing of deep-rooted native vegetation from the catchments of some estuaries has led to the water table rising and thus to salt being brought to the soil surface, from where it is washed into the rivers and eventually the estuaries where it increases their salt load (Pen, 1999). During dry periods, high evaporation rates in those estuaries that are closed to the ocean can lead to salinity becoming so elevated that it leads to massive mortalities among certain fish species (Young and Potter, 2002; Hoeksema *et al.*, 2006a). Furthermore, large inputs of nutrients from agricultural land have led to some estuaries, particularly the Peel-Harvey Estuary on the lower west coast of Australia, becoming highly eutrophic (McComb and Lukatelich, 1995) and to Wilson Inlet on the south coast becoming mesotrophic (Lukatelich *et al.*, 1987). These problems arise as south-western Australia is located in a microtidal region and thus tidal water movement in its estuaries is limited and therefore restricted in its ability to flush excess salt and nutrients out to sea.

The increasing pressure to which the estuaries of south-western Australia are being exposed through anthropogenic activities, allied with the likely effects of climate change, mean that managers require a good understanding of the physico-chemical characteristics of these estuaries so that they can develop plans aimed at preventing their further degradation and maintaining their biodiversity. This is particularly the case with estuaries on the south coast of this region, for which there are limited published data and where the full range of estuary types from permanently open to normally closed are represented. Physico-chemical data are also required by biologists to enable them to elucidate the ways in which factors such as the duration of the opening of the estuary mouth and the level of salinity influence the abundance of the key species and thus also the faunal compositions in these systems.

Salinity, water temperature and dissolved oxygen concentration at the surface and bottom of the water column at sites located throughout one permanently-open estuary, three seasonally-open estuaries and four normally-closed estuaries on the south coast of Western Australia were recorded seasonally for at least two years. The data were used to compare the physico-chemical characteristics of the two main and morphologically distinct regions of these estuaries, *i.e.* the basins and saline lower reaches of their rivers. They were also used to examine how the above three environmental variables in particularly seasonally-open estuaries varied in response to the markedly seasonal (winter/early spring) pattern of rainfall and thus also of discharge and the extent to which those of normally-closed estuaries changed during protracted dry periods. Particular emphasis was also placed on exploring the degree to which the extent of connectivity of an estuary with the ocean influences the environmental characteristics of both individual estuaries and estuary types.

3.2. Materials and methods

Environmental data were recorded at sites in the basins and rivers of the permanently-open Oyster Harbour, the seasonally-open Broke, Irwin and Wilson inlets and the normally-closed Wellstead Estuary and Hamersley, Culham and Stokes inlets, which are located along *ca* 500 km of the south coast of Western Australia (Fig. 2.1). We refer largely to Brearley (2005) for data on the physico-chemical characteristics of south coast estuaries over 20 years ago. Much of these snapshot data were derived from reports produced by E.P. Hodgkin and R. Clark for the Western Australian Environmental Protection Authority.

During the present study, records were kept of whether the mouth of each of the eight estuaries was open to the ocean and, if so, the duration of that opening. Salinity, water temperature and dissolved oxygen concentration were measured at the surface and bottom of the water column at each site in each season in both the basin and river of each of the eight estuaries (Fig. 2.1) using a Yellow Springs International Model 85 Oxygen, Conductivity, Salinity and Temperature Meter. When salinity at a site exceeded the maximum that could be recorded by this meter, *i.e.* 80, a sample was taken and diluted by a known volume of distilled water and the resultant salinity used to calculate that of the water at that site. Five to 10 sites were sampled in the basin of each estuary, the actual number reflecting the size of the basin, and a further 3 sites were located in each of the rivers (see Fig. 2.1). The mean ± 1SE for each environmental variable at the surface and bottom of the water column in the basin and river of each estuary in each season was calculated, using the values recorded at each site in each region of each estuary. Each site was located 50 to 200 m from the shore in the basins and *ca* 5 m from the shore in the rivers and in depths that typically ranged from 1 to 3 m, but occasionally reached 6 m.

Measurements in the Broke, Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary were recorded in the last month of each of the eight seasons between summer 2005/06 and spring 2007 and also in the first three estuaries in winter and spring 2005 and in Wellstead Estuary during spring 2005. Environmental variables in the Hamersley, Culham and Stokes inlets were measured in the middle of each season between summer 2001/02 and spring 2004. As the amount of rainfall and thus discharge was very low during this period, some of the selected sites in the basin of Culham Inlet became dry. Furthermore, the water level in Hamersley Inlet declined to the point where a boat could no longer be used to enter its river, which was not otherwise able to be readily accessed as it runs through an area of very dense vegetation and is bounded by steep cliffs.

3.3. Results

3.3.1. Environmental characteristics of Broke, Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary

Mean salinities at the surface and bottom of the water column in offshore, deeper waters of the basins in each of the Broke, Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary in any given season were almost invariably the same, with conspicuous haloclines developing only in Irwin Inlet in winter (Fig. 3.1). Salinities in nearshore waters of the basin of an estuary in any season did not differ significantly from those at the surface of the water column in nearby offshore waters of that estuary ($p > 0.05$).

The extent of seasonal variation in the salinities of the basins of the five estuaries differed markedly (Fig. 3.1). Thus, while salinities underwent marked seasonal changes in

Irwin Inlet in both years and in Broke Inlet in the second year, such changes were far less pronounced in Wilson Inlet, Oyster Harbour and Wellstead Estuary. Furthermore, salinities in

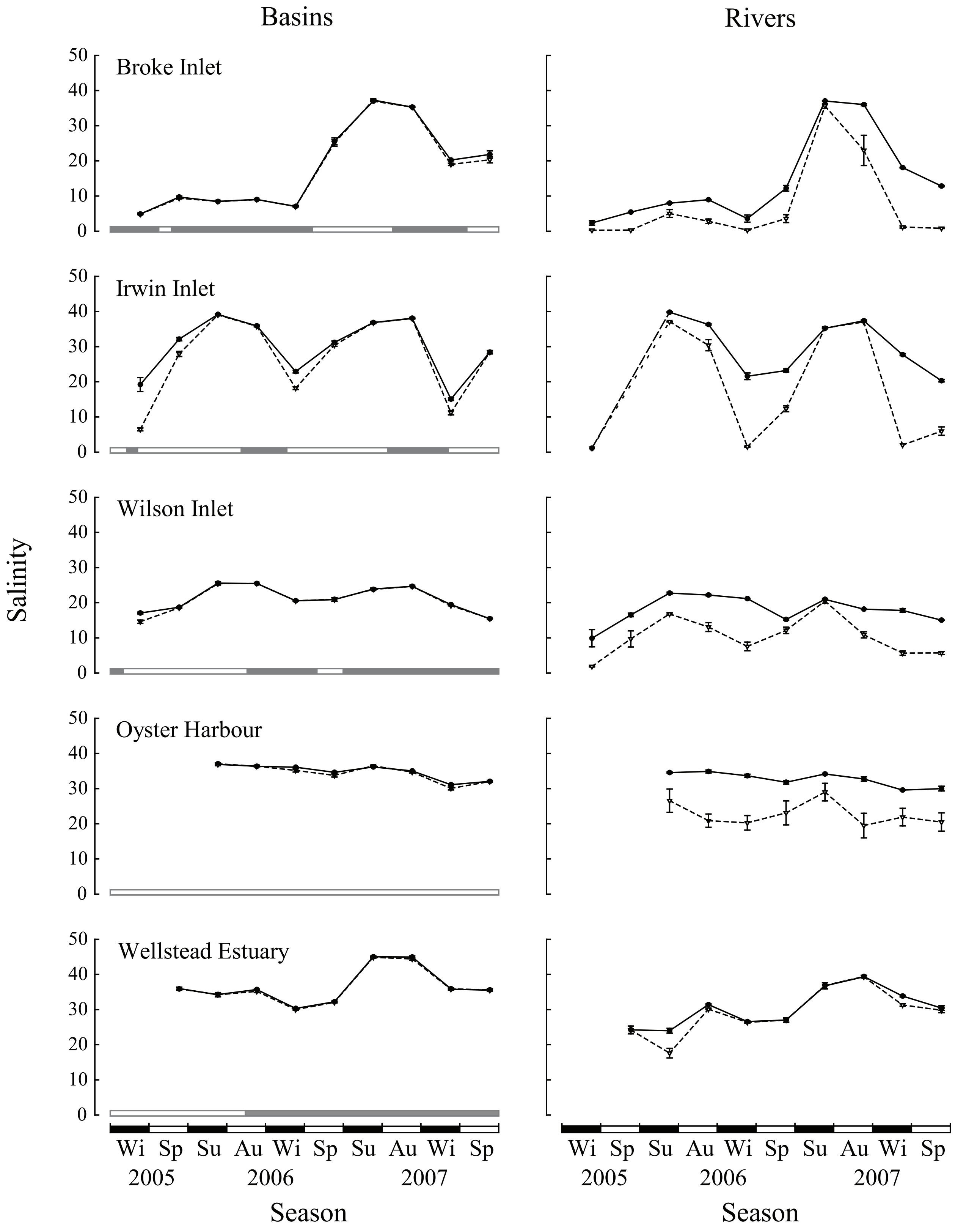

Fig. 3.1. Mean seasonal salinities ± 1SE at the surface (dashed line) and bottom (solid line) of the water column in the basins and rivers of the Broke, Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary. Open and closed grey bars on the x-axis of the basin plots for each estuary denote when the estuary mouth was open and closed, respectively.

Oyster Harbour remained at or close to that of full-strength sea water throughout the sampling period, while Wellstead Estuary became hypersaline in two seasons and salinities always remained substantially below that of full-strength sea water in Wilson Inlet (Fig. 3.1).

The mean seasonal salinities at the bottom of the water column in the main tributary of each estuary followed similar trends to those described above for their basins (Fig. 3.1). Although mean salinities in any season were often less at the bottom of the water column in the river than in the basin of an estuary, this difference was only pronounced in Wellstead Estuary. In contrast to the situation in the basins, marked haloclines were present in most seasons in the main tributaries of the Broke, Irwin and Wilson inlets and Oyster Harbour, with the maximum differences between surface and bottom of the water column being 16.9, 25.7, 13.6 and 14.0, respectively, all of which were recorded during winter (Fig. 3.1).

Mean seasonal water temperatures followed similar seasonal trends in the basins and rivers of each of the five estuaries, reaching their maxima of 23-25°C in summer and declining to their minima of 11-13°C in autumn or winter (Fig. 3.2). Although thermoclines formed in the rivers of the Broke, Irwin and Wilson inlets and Oyster Harbour in some seasons, the differences between temperatures at the surface and bottom of the water column exceeded 2.5°C only in Wilson Inlet in autumn 2007, when the difference was 5.1°C (Fig. 3.2).

Mean seasonal dissolved oxygen concentrations in the basins and rivers underwent consistent seasonal changes, with maxima occurring during winter and minima during summer (Fig. 3.3). They were almost invariably lower at the bottom than top of the water column and generally markedly so in the rivers. The maximum difference between oxygen concentrations at the surface and bottom of the water column, which was recorded in the river of Irwin Inlet during winter 2007, was 5.3 mg l^{-1} (Fig. 3.3).

3.3.2. Environmental characteristics of Hamersley, Culham and Stokes inlets

Mean seasonal salinities in the basins of Stokes, Hamersley and Culham inlets in summer 2001/02 were 31, 35 and 53, respectively (Fig. 3.4). While mean seasonal salinities in each estuary subsequently increased in an essentially progressive manner, the extent of those changes varied markedly among the three estuaries. Thus, over a three year period, mean

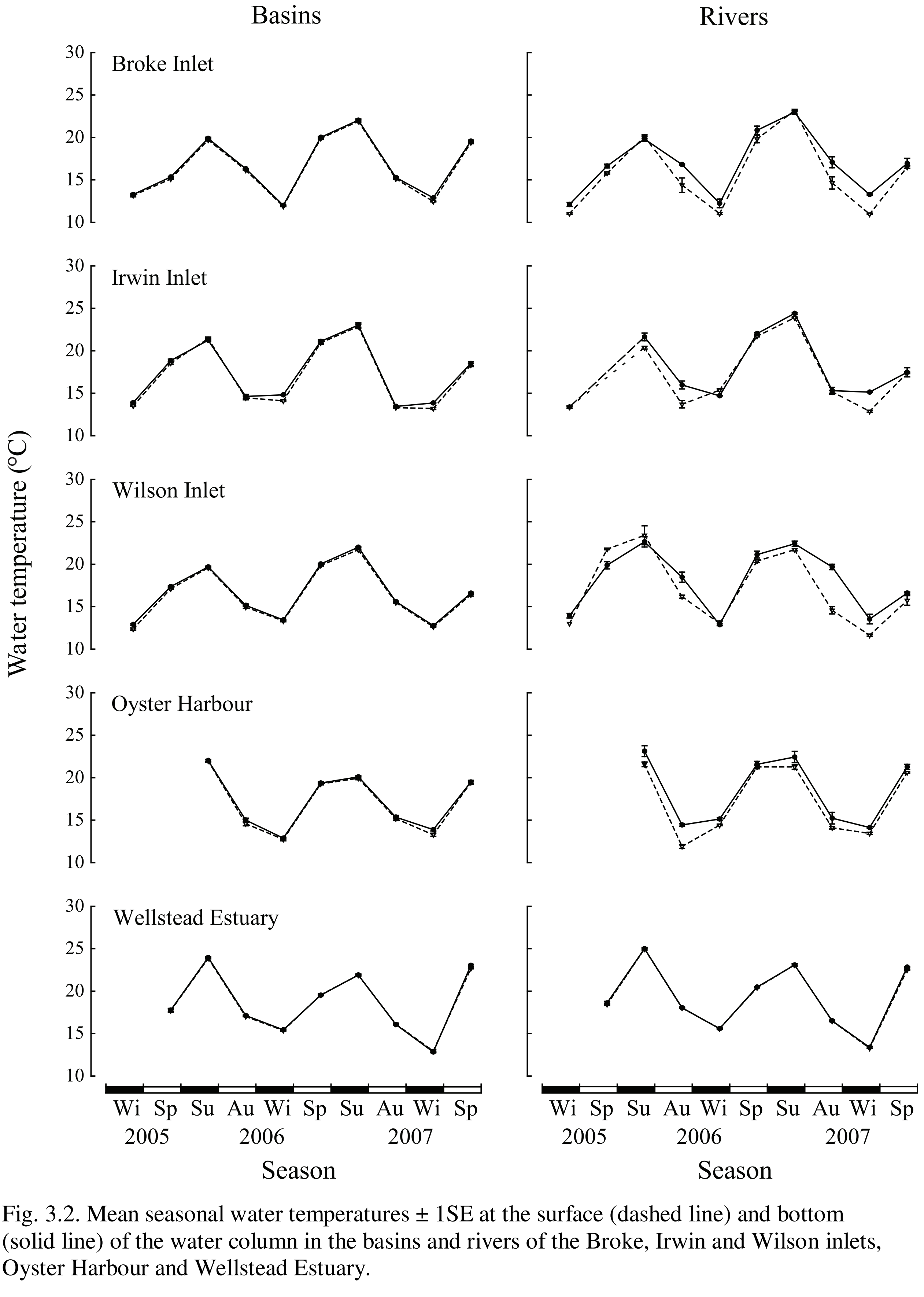

Fig. 3.2. Mean seasonal water temperatures ± 1SE at the surface (dashed line) and bottom (solid line) of the water column in the basins and rivers of the Broke, Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary.

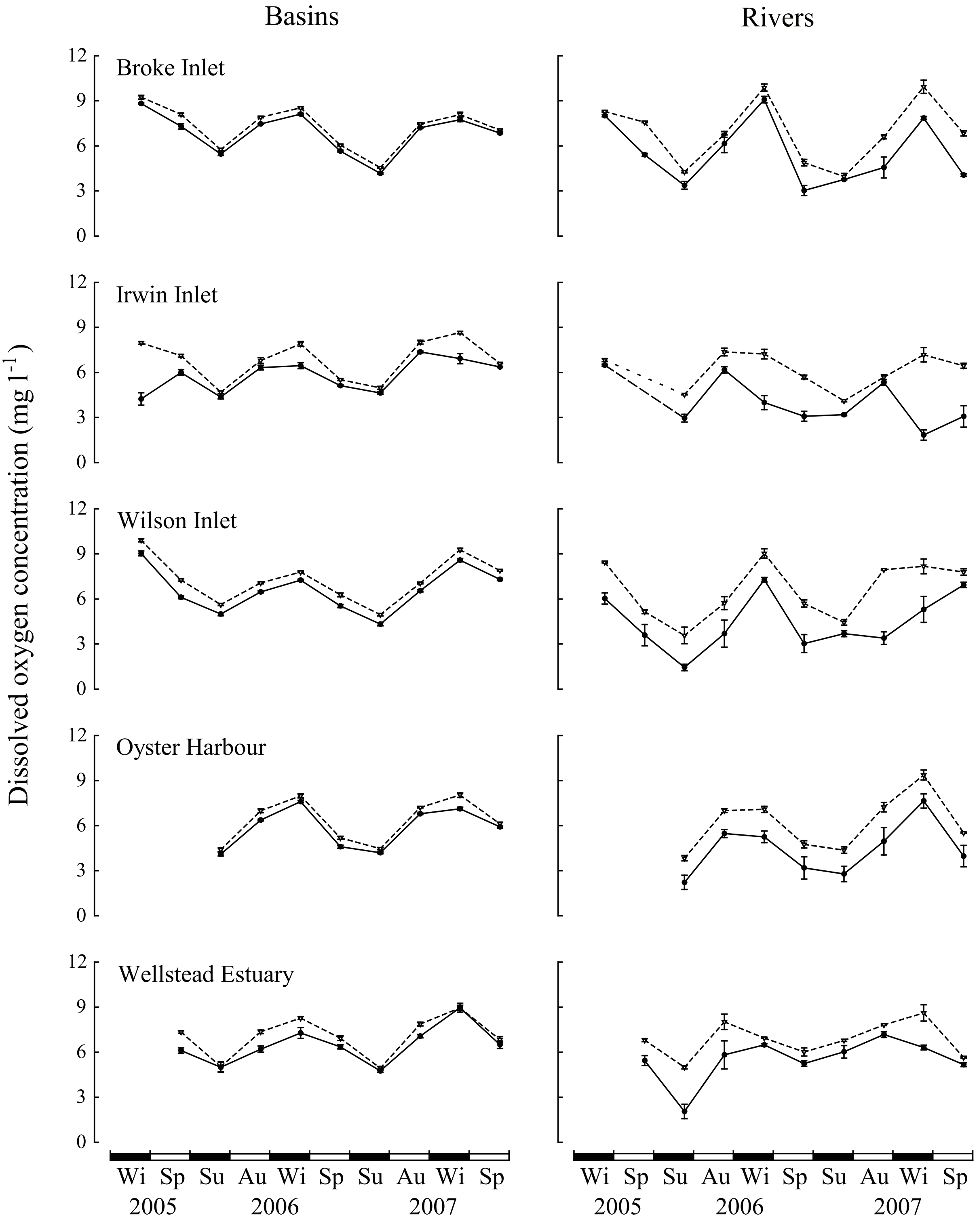

Fig. 3.3. Mean seasonal dissolved oxygen concentrations ± 1SE at the surface (dashed line) and bottom (solid line) of the water column in the basins and rivers of the Broke, Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary.

salinities rose to as high as 145 in Hamersley Inlet and to an even far greater value of 296 in Culham Inlet, but to only 64 in Stokes Inlet (Fig. 3.4). However, salinities did decline in the basins of Hamersley Inlet and more particularly Culham Inlet for a period after summer 2002/03, following the only substantial precipitation that occurred in the catchments of these two estuaries during the whole of the three year study period. In contrast to the situation in the main basin area of Culham Inlet, mean seasonal salinities in the lagoonal-like area at the entrance of Culham Inlet, which receives marine water from seepage and waves that wash over the sand bar, never exceeded 51 (Fig. 3.4).

While mean salinities at the bottom of the water column in the rivers of the Culham and Stokes inlets in each season were always slightly lower than those in their basins, they likewise increased with time (Fig. 3.4). Mean salinities at the surface of the water column in these two estuaries were always the same or similar to those at the bottom of the water column except in autumn 2003 in Culham Inlet and winter 2003 in Stokes Inlet, when they were considerably lower (Fig. 3.4) due to an increase in discharge in those seasons. Mean seasonal salinities in the upstream pools of Culham Inlet, which were formed in the deeper areas of the river as water levels became reduced, ranged from 12.0 to 28.6 (Fig. 3.4).

Mean water temperatures in the basins and rivers of the Hamersley, Culham and Stokes inlets (data not shown) followed the same trends as those already described for the Broke, Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary (see Fig. 3.2).

Mean seasonal dissolved oxygen concentrations in the basins of Hamersley, Culham and Stokes inlets rose sharply in 2002 from 5-6 mg l^{-1} in summer to 10-11 mg l^{-1} in winter and then declined markedly in spring and then summer (Fig. 3.4). After rising to higher levels in autumn and winter 2003, mean seasonal dissolved oxygen concentrations declined in the basins of Hamersley and Culham inlets over the ensuing seasons. Dissolved oxygen concentrations in the basin of Stokes Inlet underwent a less marked overall decline and exhibited seasonal cyclicity, with maxima in winter and minima in summer. Marked oxyclines were formed in the rivers of the Stokes and Culham inlets in certain seasons (Fig. 3.4).

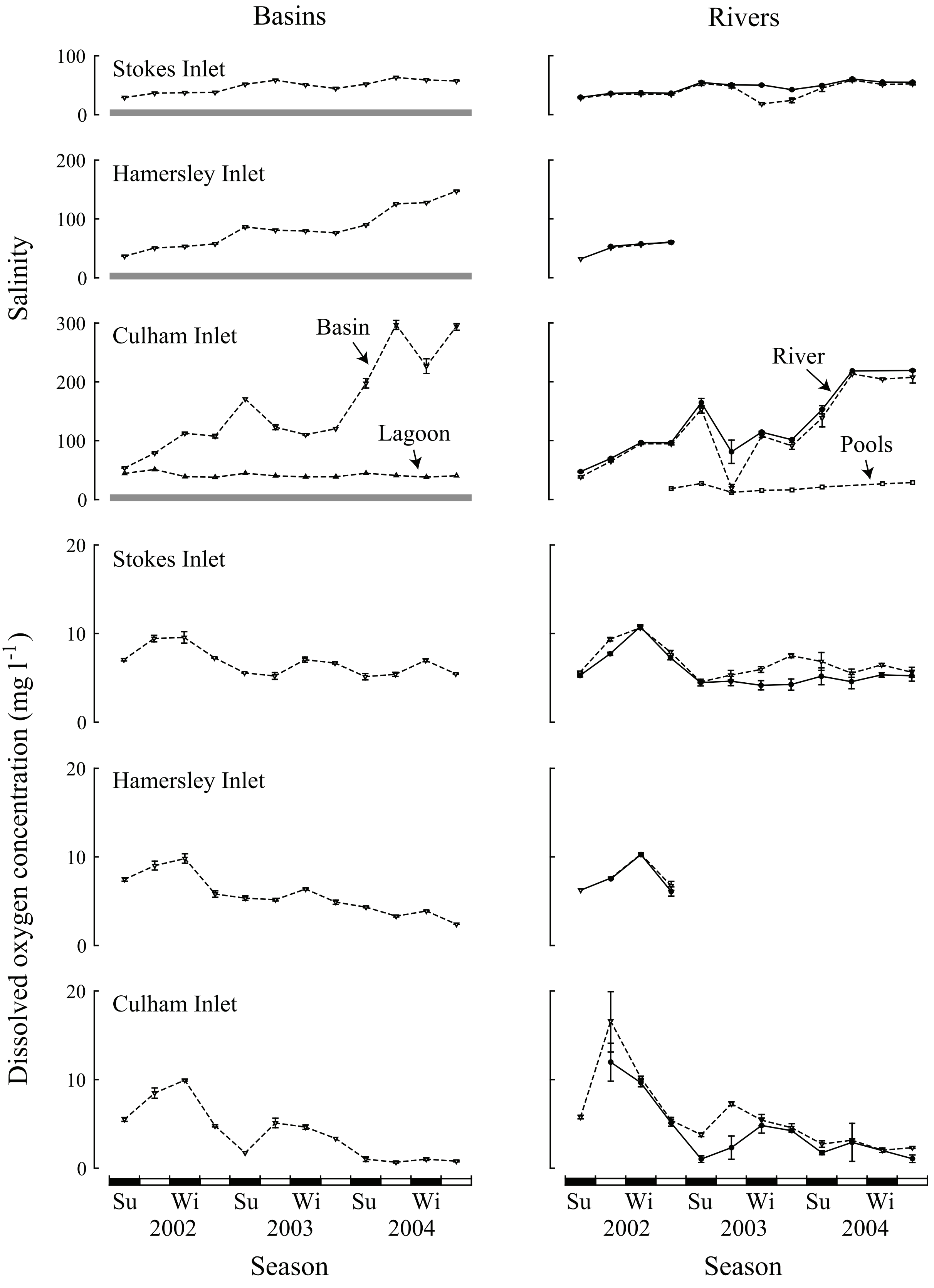

Fig. 3.4. Mean seasonal salinities and dissolved oxygen concentrations ± 1SE at the surface of the water column in the basins of the Stokes, Hamersley and Culham inlets (dashed line). N.B. Surface and bottom values were essentially the same in the basins. As mean salinities and dissolved oxygen concentrations at the surface and bottom of the water column differed in some seasons in the rivers, the trends exhibited by those variables at the bottom of the water column (solid line) are also shown. Closed grey bars on the x-axis of the basin plots denote that the mouth was closed.

3.4. Discussion

3.4.1. Permanently-open, seasonally-open and normally-closed estuaries

The data collected on the environmental characteristics of eight estuaries on the south coast of Western Australia during the present study, allied with historical records kept by governmental agencies and residents as to whether the mouths of these systems were open or closed, confirm that it is appropriate to classify each of those estuaries as either permanently open, seasonally open or normally closed to the ocean. Thus, the fact that the mouth of Oyster Harbour, which is located close to the city of Albany and thereby regularly visited by the staff of local authorities and by residents, has never been observed to be closed, clearly validates the classification of this estuary as permanently open. Consequently, Oyster Harbour is one of only four of the 29 estuaries on the south coast of Western Australia between 115.1° and 121.8° E that fall into this category, the others being the Blackwood River Estuary, Nornalup-Walpole Estuary and Waychinicup Estuary (Brearley, 2005).

The sand bars that form across the mouths of the Broke, Irwin and Wilson inlets have almost invariably been artificially breached each year for at least 45 years, when, following high winter rainfall, the water in the basins rises to a level that threatens to inundate the surrounding land, which in the cases of Irwin and Wilson inlets are of agricultural and/or urban value. These artificial breachings bring forward slightly the time in the year when the bar at the mouth of the estuary would otherwise naturally have been breached. In the case of Wilson Inlet, official records show that since 1955, the mouth of this estuary has been open at some time in every year except 2007 and anecdotal observations indicate that such openings occurred in all previous years since the early 1900s.

From the above, it follows that the Broke, Irwin and Wilson inlets are appropriately categorised as seasonally-open estuaries. The decision by managers not to breach the bar of Wilson Inlet in 2007 was due to water levels in the basin being particularly low as a result of limited discharge from the catchment following low rainfall in winter. This type of situation will become more prevalent in the future if the climate becomes drier and/or damming and extraction of water from the rivers of this estuary increase. This will obviously have an impact on the ecology of Wilson Inlet and thus have implications for the management of this system and its resources.

In contrast to the mouths of the permanently-open Oyster Harbour and seasonally-open Broke, Irwin and Wilson inlets, those of the other four estuaries investigated were either closed for the whole of the study period (Hamersley, Culham and Stokes inlets) or for all but the initial part of the study (Wellstead Estuary). Records for earlier years confirm that the mouths of these estuaries often remain closed for years at a time and thereby justify the classification of these systems as normally closed (*e.g.* Young and Potter, 2002; Brearley, 2005).

The four permanently- or seasonally-open estuaries are located in the western part of the south coast, *i.e.* west of 118.5° E, as are the other main estuaries in these two categories (Brearley, 2005). In contrast, the four estuaries that are normally closed, and also others of this category, lie further to the east. It is therefore highly relevant that rainfall and thus discharge into each of the estuaries in the former region greatly exceeds that into those in the latter region. Thus, whereas the volume of water discharged into each of the estuaries in the western region is sufficient to lead to the bars of those estuaries being breached either naturally or artificially each year, the volume of discharge into those in the eastern region is only sufficient for that purpose in years when rainfall is exceptionally high.

Hodgkin and Hesp (1998) considered that, when first flooded by the Holocene marine transgression, all estuaries in south-western Australia were permanently open and tidally influenced for some distance upstream. Subsequently, a combination of sedimentary and hydrodynamic processes and reductions in sea level increasingly led to sand bars being formed at the mouths of these estuaries (Hodgkin and Hesp, 1998). The size of a sand bar and the amount of riverine discharge, allied in some cases with artificial intervention, dictates when and for how long that bar is breached. Hodgkin and Hesp (1998) proposed that there was an “evolutionary” gradient from permanently-open estuaries, in which the sand bar is of insufficient size to form a barrier to the ocean, to those that are barred but open seasonally, to those that are normally closed by a sand bar and eventually to those that are permanently separated from the sea and which should then thus be regarded as saline coastal lakes or lagoons. The above proposed historical sequence of changes is reflected in the trend for the extent of land-locking among estuaries on the south coast of Western Australia to increase in an eastwards direction as overall rainfall, and thus fluvial discharge, declines. In a similar context, Perillo (1995) and Perillo *et al.* (1999) classified estuaries into primary and secondary systems, with the former category representing those that are essentially unmodified from their original form and the latter representing those that have been relatively modified through the interaction of marine processes and riverine discharge.

3.4.2. Comparisons of trends exhibited by environmental characteristics in the different estuaries

Any consideration of the environmental characteristics of estuaries on the microtidal south coast of Western Australia and comparisons between the different estuaries of that coast needs to recognise that, irrespective of estuary type, they almost invariably contain two morphologically very distinct main regions, *i.e.* wide basins and the narrow, lower saline reaches of the tributaries, and that, when open, they possess a short and narrow entrance channel (Hodgkin and Lenanton, 1981). Many of the larger estuaries along the southern coasts of eastern Australia and Africa likewise contain distinct basin and riverine regions (Day, 1981; Roy *et al.*, 2001; Potter *et al.*, 1990). The structure of the vast majority of these estuaries contrasts markedly with the wedge shape typically exhibited by estuaries in macrotidal temperate regions of the northern hemisphere (Day, 1981).

While salinity in the river of all estuaries underwent seasonal changes similar to that of their respective basins, marked haloclines formed throughout much of the year in the rivers but not the basins of the permanently-open Oyster Harbour and seasonally-open Broke, Irwin and Wilson inlets. The lack of haloclines in the basins is largely attributable to this region being wide, shallow and exposed and thereby subjected to wind-driven mixing. The difference in salinity with depth in the rivers was accompanied by oxyclines, with dissolved oxygen concentrations being consistently greater near the surface.

Rainfall is substantial and highly seasonal in the western part of the south coast of Western Australia, with more than 70% of the total annual rainfall occurring between early winter and late spring (Commonwealth Bureau of Meteorology, unpublished data). As riverine discharge to the estuaries of this region thus peaks during winter and early spring, salinity will decline during this period and particularly so if the mouth of the estuary is closed and thereby prevents the movement of water out of the system. The volume of water in seasonally-open estuaries eventually increases to the point where the sand bar at the estuary mouth becomes breached either naturally or artificially. This releases estuarine water to the ocean and allows the intrusion of oceanic water, thus leading to an increase in salinity within the estuary. The above sequence of events results in salinity undergoing annual cyclical changes, which are well illustrated by those recorded in the Irwin and Wilson inlets between winter 2005 and spring 2007 (Fig. 3.1). However, the extent of the salinity changes was far less pronounced in Wilson Inlet than Irwin Inlet, with salinities ranging from only 15 to 26 in the former system, compared with 6 to 39 in the latter. It is thus relevant that the basin of Irwin Inlet has a volume which is only *ca* 20% of that of Wilson Inlet and yet receives a similar volume of riverine discharge. Consequently, discharge into Irwin Inlet has a far more marked effect on salinity and leads to a more pronounced and protracted breaching of the sand bar at the estuary mouth, which, in turn, results in the intrusion of large volumes of oceanic water. On the basis of pooled data for sites throughout the basin, the maximum salinities in Wilson Inlet in September 1987 and April 1989 were also appreciably less than full strength sea water (Potter *et al.*, 1993).

Data for Broke Inlet illustrate very well how the salinity in an estuary can vary greatly between years as a result of marked interannual differences in the size and duration of the opening of the estuary mouth. Thus, salinities between winter 2005 and winter 2006, when the opening of the mouth of this estuary was small and of limited duration, did not exceed 10, whereas those in summer 2006/07 and autumn 2007, when the estuary mouth was much larger and remained open for a far longer period, reached that of full strength sea water and underwent a marked cyclical change during the year. In contrast to the situation in the above seasonally-open estuaries, salinities in the basin of Oyster Harbour were always close to that of full-strength sea water due to the entrance channel of this permanently-open estuary being relatively wide and deep and thus allowing a very substantial exchange of water between the ocean and the estuary.

Environmental data for Stokes, Hamersley and Culham inlets demonstrate that salinity regimes can vary markedly among normally-closed estuaries as they do amongst seasonally-open estuaries. Thus, over the same three consecutive years of particularly low rainfall (2002-2004), mean seasonal salinities in the basins rose only to 64 in Stokes Inlet, whereas they reached 145 in Hamersley Inlet and the remarkably high level of 296 in Culham Inlet. These differences largely reflect the fact that the basin of Stokes Inlet is the deepest of the three estuaries (Brearley, 2005), whereas that of Culham Inlet is the shallowest and thus the most susceptible to the effects of evaporation. Although salinities rose to exceptionally high levels in the basin and the river of Culham Inlet, they remained < 51 in the lagoonal-like area at the seaward end of its basin and < 29 in the upstream pools, to which the upper part of the river becomes reduced when water levels become low during extended dry periods. The lagoonal-like area and upstream pools are thus able to constitute refugia for fish species when salinities in the main body of the estuary rise to exceptional levels (Hoeksema *et al.*, 2006a). While salinities in the basin of the normally-closed Wellstead Estuary remained between 30 and 45 in 2005-2007, they exhibited marked fluctuations in 1996-1998 when, for example, they rose sharply from *ca* 53 to 112 during the relatively dry period between July 1996 and March 1997 and then declined to *ca* 14 in September 1997 following substantial riverine discharge and subsequently rose again following a brief opening of the sand bar (Young and Potter, 2002).

Our data demonstrate that the environmental characteristics of individual estuaries on the south coast of Western Australia are influenced by interactions among a number of factors. These factors include the volume of fluvial discharge, which is determined by a combination of the amount of local rainfall, catchment size and the extent to which native vegetation has been cleared, the size and depth of the estuary basin and the amount of marine intrusion, which is largely controlled by the size and duration of the opening of the estuary mouth. Thus, while certain attributes are common to each category of estuary, *i.e.* permanently open, seasonally open or normally closed, each individual estuary has a unique combination of features that dictates its particular environmental characteristics.

4. Black Bream *Acanthopagrus butcheri*

4.1. Introduction

Among the large estuarine-spawning species of south-western Australia, Black Bream *Acanthopagrus butcheri* (Sparidae), which is endemic to southern Australia, is the only species that is not also represented by marine populations (Chaplin *et al.*, 1998; Potter and Hyndes, 1999). This species makes a major contribution to the recreational and commercial fisheries in the estuaries of south-western Australia (Lenanton and Potter, 1987; Loneragan *et al.*, 1987; Smallwood and Sumner, 2007) and Victoria (*e.g.* Kailola *et al.*, 1993; Grixti *et al.*, 2008) and is particularly abundant in the estuarine regions of their tributaries (*e.g.* Hindell *et al.*, 2008; Chuwen *et al.*, 2009b). It can dominate the fish faunas of offshore, deeper waters in the basins of some estuaries that become closed to the sea for protracted periods and which thus limit the opportunities for recruitment of marine species (*e.g.* Young and Potter, 2002). The ability of *A. butcheri* to tolerate a wide range of salinities from close to that of fresh water (Young *et al.*, 1997) to that of nearly twice full-strength sea water (Young and Potter, 2002; Hoeksema *et al.*, 2006a) demonstrates that this sparid must be highly euryhaline and helps account for its great success in a range of different estuary types.

The genetic composition of the different populations of this species in south-western Australia varies, implying that the individuals in any one estuary tend to remain in that estuary (Chaplin *et al.*, 1998). Although this could suggest that genetic differences might account for the marked differences in the growth that are exhibited by *A. butcheri* among estuaries (*e.g.* Hobday and Moran, 1983; Morison *et al.*, 1998; Sarre and Potter, 2000), this possibility was not supported by the results of a culture experiment. In that experiment, brood stocks of *A. butcheri* were collected from two different estuaries in which the growth of this species had been shown to vary markedly (Sarre and Potter, 2000). The groups of juveniles derived from both broodstocks were reared separately, but under the same environmental and dietary conditions (Partridge *et al.*, 2003). The growth of both groups was essentially the same, suggesting that the marked difference in growth among the two wild populations was due to environmental rather than genetic factors. It thus appears relevant that this species is a highly opportunistic feeder and that its dietary composition differs markedly among the various estuaries in which it is found and thus in estuaries at different latitudes and therefore with different temperature regimes (Sarre *et al.*, 2000; Chuwen *et al.*, 2007). In addition, Sarre and Potter (2000) found that the early growth of *A. butcheri* was least in the estuary in which the densities of particularly the juveniles of this sparid were greatest, suggesting that density could influence the growth of the juveniles of this species. If the differences in growth of *A. butcheri* among estuaries reflect differences in environmental conditions, it would follow that the growth of this species in an estuary would vary between years when environmental conditions varied markedly.

Acanthopagrus butcheri spawns between the late winter to early summer period (Haddy and Pankhurst, 1998; Sarre and Potter, 1999) and predominantly in the upper reaches of estuaries (Haddy and Pankhurst, 2000). The length of *A. butcheri* at maturity can vary markedly among estuaries, particularly as a result of variations in the growth of juveniles among different populations (Sarre and Potter, 1999). Thus, while *A. butcheri* generally reaches sexual maturity at the end of its second year of life, the length at maturity is greater in those populations in which juvenile growth is most rapid.

Recruitment of juvenile *A. butcheri* varies markedly among years in a number of estuaries along the south-western coastline of Western Australia (Sarre and Potter, 2000) and Victoria (Hobday and Moran, 1983; Morrison *et al.*, 1998). Hobday and Moran (1983) found that year classes tended to be strongest in years with relatively dry springs and that weak year classes tended to occur during years with high river flows in spring and low water temperatures. On the basis of the results of planktonic sampling at various depths in two estuaries, Nicholson *et al.* (2008) suggested that the mortality of the eggs and early larvae of *A. butcheri* increased, when, due to stratification and reduced river flow, hypoxic conditions prevailed. This proposal is supported by the fact that, under laboratory conditions, the mortality rates of those two life stages increased under hypoxic conditions (Hassell *et al.*, 2008a, b).

Sarre and Potter (2000) found evidence that little or no spawning and/or recruitment success of *A. butcheri* occurred in the normally-closed Wellstead Estuary in those years when discharge was particularly heavy and thus provided the potential for eggs, larvae and small juveniles to be flushed out of the estuary. In contrast, *A. butcheri* exhibited more consistent annual recruitment in estuaries that lie in regions of south-western Australia where total annual rainfall is greater and more seasonally predictable, with all age classes up to 15+ being represented (Sarre and Potter, 2000).

Because of the high value of *A. butcheri* to recreational and commercial fishers, and the very different levels of fishing for this species in the different estuaries, fishing mortality for this sparid is likely to vary considerably among estuaries. Furthermore, as natural mortality is likely to be influenced greatly by the environmental conditions to which individuals of this species in the different estuaries are subjected, total mortality is likely also to vary considerably among estuaries, even in the absence of fishing. The influence of the latter type of factor can be extreme, such as was the case with the massive mortality of *A. butcheri* that resulted from salinities increasing to lethal levels for this species in two estuaries on the south coast of Western Australia (Hoeksema *et al.*, 2006a).

Representative samples of *A. butcheri* were collected from eight estuaries that covered the range of estuary types found on the south coast of Western Australia and in which the salinity regimes and levels of primary productivity, and the fishing pressure on and density of *A. butcheri* differed (Chuwen *et al.*, 2009b). The resultant data have been used to estimate the growth, length and age at maturity, spawning period, annual recruitment and mortality of *A. butcheri* in each of those estuaries. This enabled the following hypotheses to be tested.
1) The growth of *A. butcheri* differs among estuaries in which fish densities and/or environmental characteristics differ, but which are located at approximately the same latitude.
2) The growth of individuals of *A. butcheri* varies among years and particularly in normally-closed estuaries in which environmental characteristics can vary markedly among years.
3) The lengths at first maturity of female and male *A. butcheri* are greatest for those populations in which juvenile growth is most rapid. In addition, the levels of total mortality of *A. butcheri* in the different estuaries were determined and the annual spawning and/or recruitment success of juveniles of *A. butcheri* on the south coast of Western Australia were considered in the context of inter-annual variations in environmental conditions.

4.2. Materials and methods

The materials and methods that are relevant to both species considered in this report are given in the *General materials and methods* section, while those that relate specifically to *Acanthopagrus butcheri* are given below.

4.2.1. Variation in the growth of otoliths

Preliminary plots indicated that the relationships between total fish length and otolith radius for each sex of *A. butcheri* differ among estuaries. Given that previous studies provided strong evidence that the growth of *A. butcheri* is influenced far more by environmental than genetic factors (Partridge *et al.*, 2003), it is possible that the fish length-otolith radius relationship for *A. butcheri* varies according to environmental conditions. As back-calculation of fish length from the radius at each of the growth zones relies on the relationship between these variables being constant (Wilson *et al.*, 2009), it was considered inappropriate to use back-calculated estimates of fish length to explore whether growth varies among years. However, it was appropriate to use inter-annual differences between the increments of the radii of successive growth zones to investigate whether the growth of the otolith differed among years and whether any such differences could be related to inter-annual variations in environmental conditions.

The distances from the primordium to the outer edge of each successive opaque zone and to the outer edge of the otolith, *i.e.* radii, in each of 50 randomly-selected otoliths from female and male *A. butcheri* from each of Wilson Inlet, Oyster Harbour, Wellstead Estuary and Stokes Inlet were measured. Measurements were made to the nearest 0.1 µm along the same axis of each sectioned otolith employing Leica IM 1000 software and images captured with a Leica DC300 camera attached to a Leica MZ75 dissecting microscope and using reflected light.

The data for each sex within each estuary were analysed to determine whether the distance between the outer edges of each pair of successive opaque zones in otoliths (hereafter termed an annual growth increment) was influenced by the year in which the growth occurred, which was considered a fixed factor. For each annual growth increment, the data for those growth years for which there were more than three observations were analysed using a one-way analysis of variance (ANOVA) employing the procedure “lm” in R (R Development Core Team, 2008). Use of the initial radius at the outside edge of the first of the pair of opaque zones associated with the annual growth increment as a covariate was also explored. However, as the results were essentially the same as those obtained in the absence of the covariate, the results from that more complex analysis are not reported.

4.2.2. Variability in annual recruitment of 0+ juveniles

Attempts to relate recruitment strength with particular environmental variables are hindered by the lack of sound historical data on such variables as salinity and dissolved oxygen concentration and of riverine discharge in most of the estuaries. Thus, such attempts have to rely on proxies for these variables, *e.g.* rainfall for riverine discharge, recognising that these proxies have limited reliability. However, the 0+ age class of *A. butcheri* was caught by seine netting in the rivers and basins of some estuaries and thus there are environmental data during the periods of that sampling, which can be compared with the recruitment of 0+ fish. The mean densities of 0+ *A. butcheri* were calculated from the seine net catches obtained in the spring to winter seasons of 2005/06 and 2006/07 from sites in the river and basin of Oyster Harbour and Wellstead Estuary, the estuaries in which comparable nearshore habitats could be sampled in both of those regions. As spawning occurs in spring, those samples represented the 2005 and 2006 year classes, respectively. The same procedure was adopted for the 0+ age class in the Culham and Stokes inlets in 2001/02, 2002/03 and 2003/04, which thus corresponded to the 2001, 2002 and 2003 year classes, respectively.

4.3. Results

4.3.1. Growth and length and age compositions

The length-frequency compositions of *Acanthopagrus butcheri* varied markedly among the different estuaries sampled (Fig. 4.1). Thus, the total lengths ranged from 55 to 309 mm in Hamersley Inlet, from 10-12 to 368-376 mm in Wellstead Estuary, Culham Inlet and Stokes Inlet, from 46 to 371 mm in Irwin Inlet and from 49 to 389 mm in Oyster Harbour. The two individuals caught in Broke Inlet measured 320 and 405 mm TL. Age composition also varied markedly among estuaries, with the age classes present ranging from 0+ to 5+ in

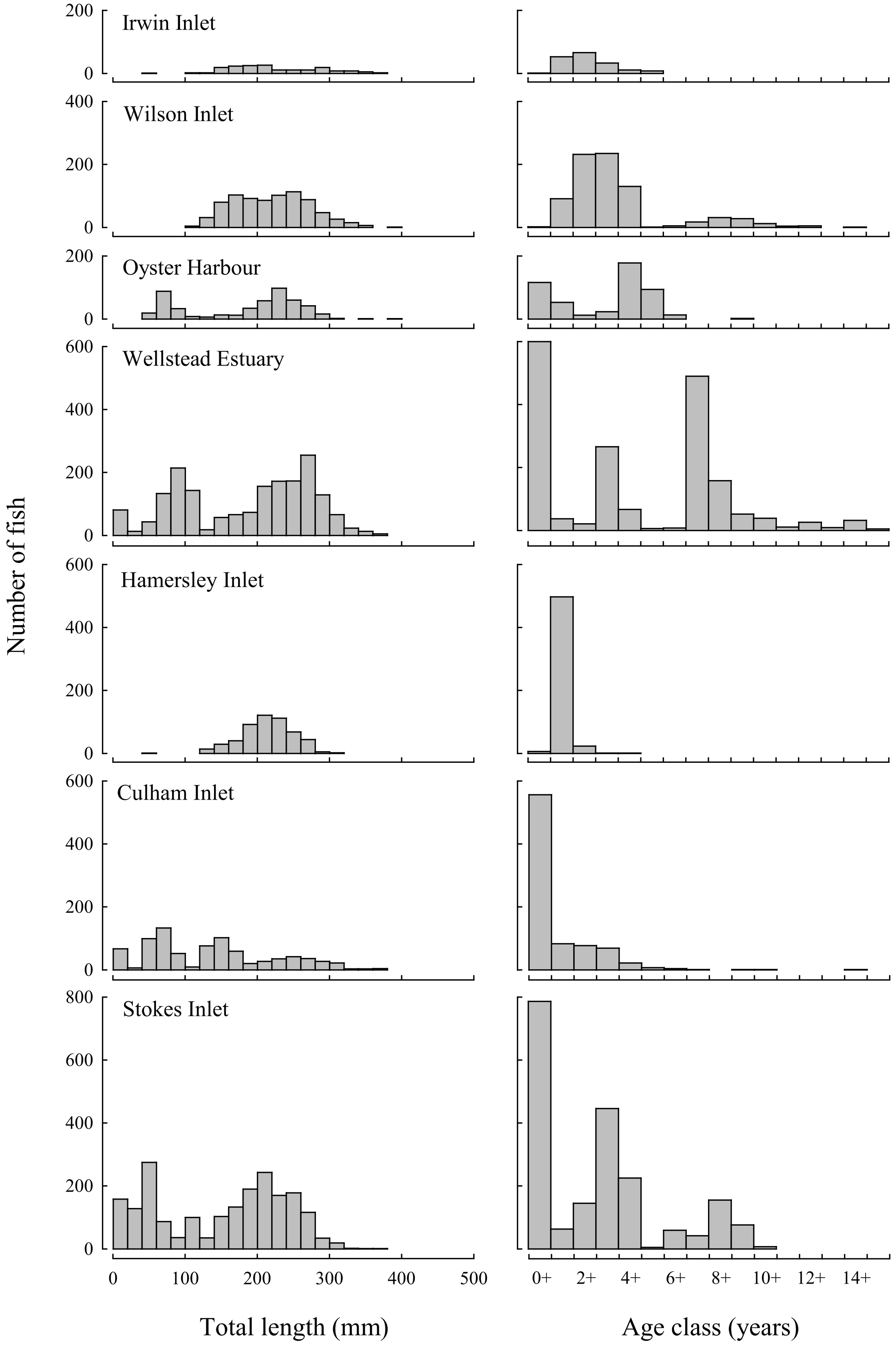

Fig. 4.1. Length and age class frequency distributions of *Acanthopagrus butcheri* caught with seine and gill nets in Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary in 2005/07 and in Hamersley, Culham and Stokes inlets in 2002-2004.

Hamersley and Irwin inlets, 0+ to 13+ in Stokes Inlet and 0+ to 15+ in Wilson Inlet, Wellstead Estuary and Culham Inlet (Fig. 4.1). The two *A. butcheri* caught in Broke Inlet were 3+ and 5+ years of age (data not shown).

von Bertalanffy growth curves provided good fits to the lengths at age for both the females and males of *A. butcheri* from Wilson Inlet, Oyster Harbour, Wellstead Estuary, Culham Inlet and Stokes Inlet (Fig. 4.2, Table 4.1). However, although the growth curves fitted the data well for both females and males in Irwin Inlet (Fig. 4.2), the L_∞s estimated for this species were very high in that estuary (Table 4.1), reflecting, in part, the lack of older individuals of this species. The very restricted age range for fish in Hamersley Inlet meant that reliable estimates for the growth parameters for this species could not be obtained. The likelihood-ratio test demonstrated that, while the growth curves of the females and males in Oyster Harbour, Wellstead Estuary, Culham Inlet and Stokes Inlet differed significantly (all $p < 0.001$), those in the Irwin and Wilson inlets did not (Fig. 4.2). These differences were considered likely to be of biological significance only in the case of Oyster Harbour, where, unlike the situation in the other estuaries, the differences between the estimated lengths for each sex at ages 1, 2, and 5 years each exceeded 5%.

The growth curves for the corresponding sex in Irwin and Wilson inlets, Oyster Harbour, Wellstead Estuary and Stokes Inlet differed significantly among the populations in each pair of estuaries (all $p < 0.001$). These differences were considered biologically significant in each pairwise comparison as the differences between the estimated lengths at ages 1, 2, 5 and 10 years were typically > 5% and frequently exceeded 10% (Fig. 4.3).

The growth curves determined for females and males of *A. butcheri* in Wellstead Estuary during the present study (2005-2007) differed significantly from those derived for the corresponding sexes of this sparid in the same estuary in 1995-96 (both $p < 0.001$) (Fig. 4.4). These differences were considered biologically significant as the differences between the estimated lengths for both females and males at ages 2, 5, 10 and 15 were almost invariably > 10% and exceeded 15% at ages 5, 10 and 15 years in the case of females and at 5 years in the case of males (Fig. 4.4).

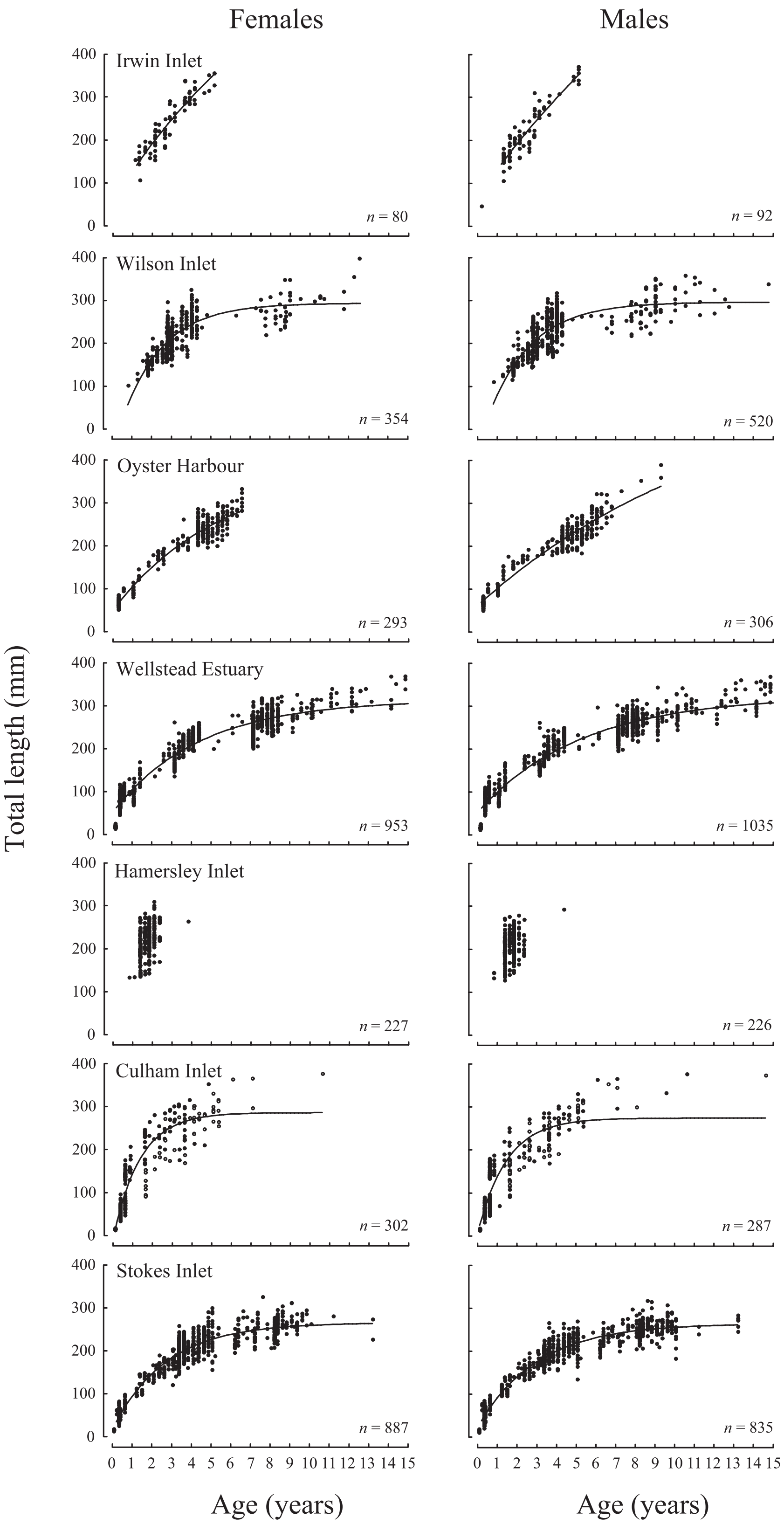

Fig. 4.2. von Bertalanffy growth curves fitted to lengths at age of female and male *Acanthopagrus butcheri* from Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary, Culham and Stokes inlets. The lengths at age of females and males from Hamersley Inlet are also shown. *n* = number of fish.

Table 4.1. von Bertalanffy growth parameters and their 95% confidence intervals (CI), derived from lengths at age of female and male *Acanthopagrus butcheri* caught in Irwin Inlet, Wilson Inlet, Oyster Harbour, Wellstead Estuary, Culham Inlet and Stokes Inlet. L_∞ is the asymptotic length (mm), k is the growth coefficient (y^{-1}), t_0 is the hypothetical age at which fish would have zero length (y) and R^2 is the coefficient of determination. n, L_{max}, W_{max}, and A_{max} represent sample sizes, maximum lengths (mm), maximum weights (g) and maximum ages (years), respectively. N.B. Confidence intervals were not estimated for the females or males in Irwin Inlet due to low sample sizes.

						von Bertalanffy parameters			
		n	L_{max}	W_{max}	A_{max}	L_∞	k	t_0	R^2
Irwin Inlet	Females	80	355	822	5	944.0	0.08	-1.07	0.86
	Lower CI								
	Upper CI								
	Males	93	371	868	5	3508.3	0.02	-1.52	0.84
	Lower CI								
	Upper CI								
Wilson Inlet	Females	354	398	1059	12	294.1	0.48	0.36	0.67
	Lower CI					283.4	0.41	0.11	
	Upper CI					304.7	0.56	0.61	
	Males	520	358	1125	14	296.7	0.46	0.32	0.67
	Lower CI					288.1	0.40	0.10	
	Upper CI					305.3	0.52	0.54	
Oyster Harbour	Females	293	332	671	6	383.0	0.18	-0.77	0.95
	Lower CI					333.7	0.14	-0.93	
	Upper CI					432.3	0.23	-0.60	
	Males	306	389	952	9	608.2	0.08	-1.38	0.93
	Lower CI					408.9	0.04	-1.65	
	Upper CI					807.5	0.11	-1.11	
Wellstead Estuary	Females	953	368	1064	15	313.3	0.23	-0.78	0.93
	Lower CI					305.1	0.21	-0.88	
	Upper CI					321.5	0.24	-0.69	
	Males	937	368	1028	15	323.6	0.19	-0.94	0.92
	Lower CI					314.6	0.17	-1.04	
	Upper CI					332.6	0.20	-0.84	
Culham Inlet	Females	302	376		10	285.9	0.73	0.03	0.88
	Lower CI					272.9	0.62	-0.02	
	Upper CI					298.9	0.84	0.08	
	Males	287	374		14	274.0	0.68	-0.04	0.83
	Lower CI					257.9	0.56	-0.11	
	Upper CI					290.2	0.80	0.04	
Stokes Inlet	Females	887	325	889	13	265.6	0.37	-0.18	0.94
	Lower CI					261.3	0.35	-0.23	
	Upper CI					270.0	0.39	-0.14	
	Males	835	317	640	13	264.4	0.33	-0.28	0.94
	Lower CI					260.5	0.31	-0.33	
	Upper CI					268.3	0.35	-0.23	

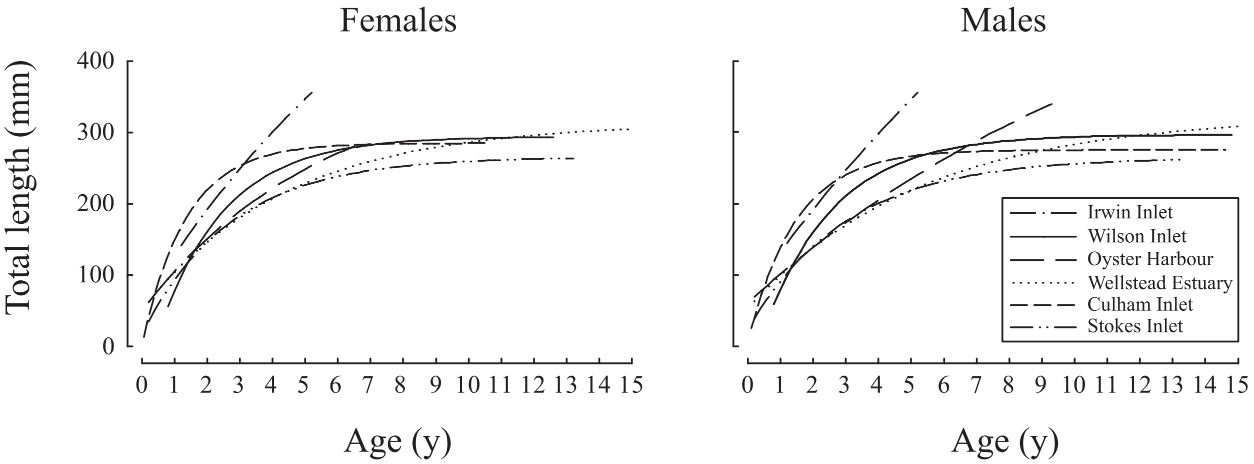

Fig. 4.3. Comparisons between von Bertalanffy growth curves for the females and males of *Acanthopagrus butcheri* in Irwin and Wilson inlets, Oyster Harbour, Wellstead Estuary and Culham and Stokes inlets.

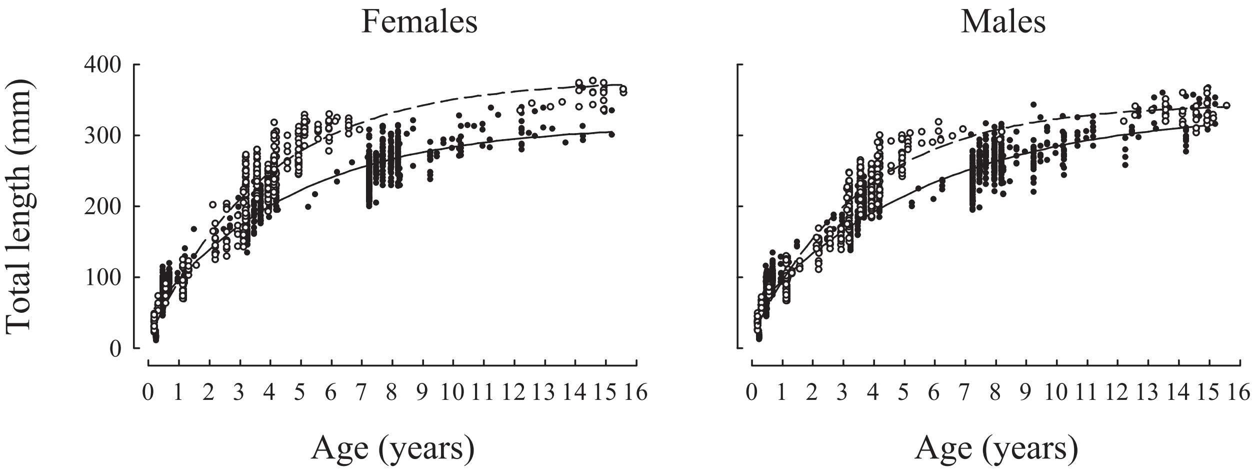

Fig. 4.4. Comparisons between von Bertalanffy growth curves for the females and males of *Acanthopagrus butcheri* in Wellstead Estuary derived from data collected in 1995-96 (open circles and dashed line) and in 2005-07 (solid circles and solid line).

4.3.2. Interannual variations in the growth of otoliths

Analysis of variance (ANOVA) of the otolith increment data for each annual growth increment for each sex within each estuary demonstrated that, in certain estuaries and for certain annual growth increments, the increment radii of the otoliths of *A. butcheri* differed significantly among years (Table 4.2). Thus, for example, the growth of otoliths was particularly strong for growth zones 1, 4 and 6 in Stokes Inlet during the 1998/99 year of growth (Table 4.2). Similarly, there was a consistent and significant decline in the growth of the third annual growth increment in Oyster Harbour in 2003/04 relative to that recorded for 2002/03. While significant differences in inter-annual growth were found for some other year, sex and estuary combinations, the pattern was not consistent among the different annual growth increments, possibly due to inadequate sample sizes given the variability of growth increment widths in the data.

Table 4.2. Overall *F*-statistics and significance levels for one-way ANOVAs of otolith increment widths for female and male *Acanthopagrus butcheri* in Wilson Inlet, Oyster Harbour, Wellstead Estuary and Stokes Inlet for each increment number (Y) and, the mean increment radii, standard error (SE) and significance level for each pairwise comparison with the first year. ns = not significant, * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. *n* = number of samples.

	Increment number	Year	Mean radius (µm)	SE	*n*	*F*-statistic	Significance	Significance (*vs* 1st year)
Wilson Inlet								
Females	1	Y			41	5.47	***	
		1997/98	311.3	21.4	4			
		1998/99	315.8	40.7	4			ns
		2001/02	416.5	11.6	5			**
		2002/03	363.5	14.0	13			*
		2003/04	314.9	8.0	11			ns
		2004/05	379.8	14.6	4			*
Males	1	Y			41	5.08	**	
		1996/97	341.4	15.0	4			
		1997/98	297.2	16.3	4			ns
		2001/02	394.5	19.6	6			ns
		2002/03	390.2	12.9	15			ns
		2003/04	338.1	13.6	12			ns
	4	Y			14	5.75	*	
		1999/00	91.0	7.2	4			
		2000/01	65.6	5.1	4			*
		2004/05	92.5	5.6	6			ns
Oyster Harbour								
Females	3	Y			33	15.43	***	
		2002/03	147.8	6.3	10			
		2003/04	123.5	3.0	23			***
Males	3	Y			33	23.74	**	
		2002/03	142.0	5.0	9			
		2003/04	118.5	4.0	24			**
Wellstead Estuary								
Females	3	Y			43	3.63	*	
		1999/00	84.0	8.7	4			
		2000/01	83.3	3.1	20			ns
		2003/04	92.5	2.5	4			ns
		2004/05	99.6	4.3	15			ns
Males	2	Y			41	6.90	**	
		1999/00	124.6	5.1	22			
		2003/04	159.4	8.4	15			***
		2004/05	134.2	14.7	4			

Table 4.2. Continued.

	Increment number	Year	Mean radius (µm)	SE	*n*	*F* statistic	Significance	Significance (*vs* 1st year)
Stokes Inlet								
Females	1	Y			52	21.56	***	
		1993/94	382.9	9.4	19			
		1995/96	408.4	12.1	6			ns
		1998/99	495.3	14.2	27			***
	3	Y			48	7.36	**	
		1995/96	131.0	3.9	19			
		1997/98	122.2	6.0	6			ns
		2000/01	109.8	3.9	23			***
	4	Y			27	66.37	***	
		1996/97	84.4	2.7	21			
		1998/99	132.0	6.7	6			***
	6	Y			26	6.13	*	
		1998/99	90.2	2.9	20			
		2000/01	74.3	5.9	6			
Males	1	Y			40	6.37	**	
		1992/93	317.7	27.1	4			
		1993/94	375.8	10.7	20			*
		1995/96	388.9	12.2	7			*
		1998/99	432.6	15.3	9			***
	2	Y			42	4.07	*	
		1993/94	218.4	6.7	4			
		1995/96	166.1	7.8	22			**
		1996/97	158.5	6.1	6			**
		1999/00	159.7	8.5	10			**
	3	Y			40	6.93	***	
		1994/95	131.2	8.8	4			
		1995/96	129.2	3.0	20			ns
		1997/98	141.7	13.4	7			ns
		2000/01	100.8	5.7	9			*
	4	Y			33	12.11	***	
		1995/96	85.4	4.2	4			
		1996/97	85.2	4.0	22			ns
		1998/99	125.7	9.0	7			**
	6	Y			33	12.06	***	
		1997/98	63.7	10.9	4			
		1998/99	94.8	3.5	22			**
		2000/01	62.1	5.2	7			ns

4.3.3. Reproductive biology

In the four estuaries for which the sample sizes of *A. butcheri* were substantial, and for which logistic regression analysis could be applied successfully, the minimum length of mature females during the spawning period ranged from 134 mm in Stokes Inlet to 139 mm in Wellstead Estuary, to 148 mm in Wilson Inlet and 163 mm in Oyster Harbour (Fig. 4.5). The corresponding minimum lengths of the males at maturity in each of those estuaries were within 3-8 mm of those of the females (Fig. 4.5). The L_{50}s at maturity in Wilson Inlet, Oyster Harbour and Stokes Inlet ranged only from 159 to 161 mm for females and only from 155 to

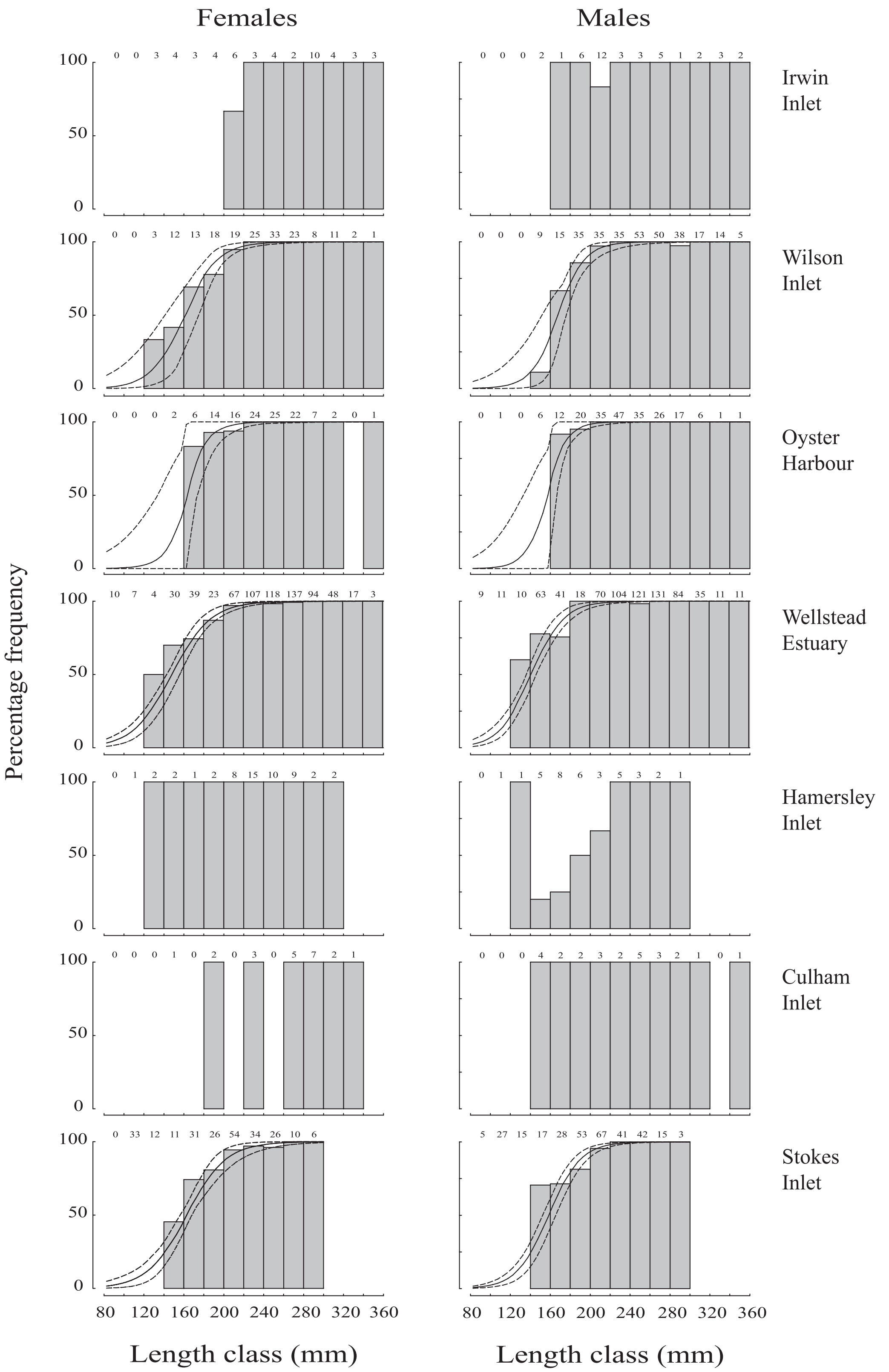

Fig. 4.5. Percentage frequencies of occurrence of mature fish, *i.e.* with ovaries or testes at stages III-VIII, in sequential total length classes of *Acanthopagrus butcheri*. The logistic curves (solid lines) and their 95% confidence intervals (dotted lines) were derived from logistic regression analyses that described the relationship between total length and the probability that an individual fish was mature. Sample sizes in each length class are shown.

164 for males (Table 4.3) and the L_{50}s for neither sex differed significantly between any pair of these estuaries (all $p > 0.05$). Although the L_{50}s for females (146 mm) and males (138 mm) in Wellstead Estuary were less than those for the corresponding sexes in the other three estuaries, those differences were not significant (all $p > 0.05$). More than 95% of females were mature by lengths of 227 mm in Stokes Inlet, 216 mm in Wellstead Estuary, 206 mm in Wilson Inlet and 214 mm in Oyster Harbour. The lengths at which > 95% of males were mature in each of those estuaries were between 14 and 32 mm of those of the females.

No *A. butcheri* became mature at the end of their first year of life. However, 73% of females were mature at the end of their second year of life in Wilson Inlet, as were 96-100% in Oyster Harbour, Wellstead Estuary and Stokes Inlet and essentially all individuals were mature by the end of their third year of life in those estuaries. The same trends were exhibited by males in the above four estuaries.

Mean monthly GSIs of female *A. butcheri* > L_{50} at first maturity in each of the estuaries rose from < 1.0 in autumn to between 2.4 and 3.2 in Wilson Inlet, Oyster Harbour, Wellstead Estuary and 5.5 in Stokes Inlet in winter, and then to peaks > 4.3 in spring in each of the estuaries, before declining markedly to < 1.0 in summer (Fig. 4.6). The mean seasonal GSIs of males followed a similar trend to that for females (Fig. 4.6).

Table 4.3. Lengths at first maturity (L_{50}) and their 95% confidence intervals (CI) for female and male *Acanthopagrus butcheri* in Wilson Inlet, Oyster Harbour, Wellstead Estuary and Stokes Inlet.

		L_{50} (mm)	Lower 95% CI	Upper 95% CI
Wilson Inlet	Females	159.0	138.6	172.4
	Males	164.0	149.2	173.4
Oyster Harbour	Females	161.3	131.4	171.2
	Males	155.1	130.3	163.8
Wellstead Estuary	Females	145.6	140.4	153.6
	Males	138.2	133.5	143.4
Stokes Inlet	Females	160.2	153.8	165.5
	Males	156.0	149.9	163.1

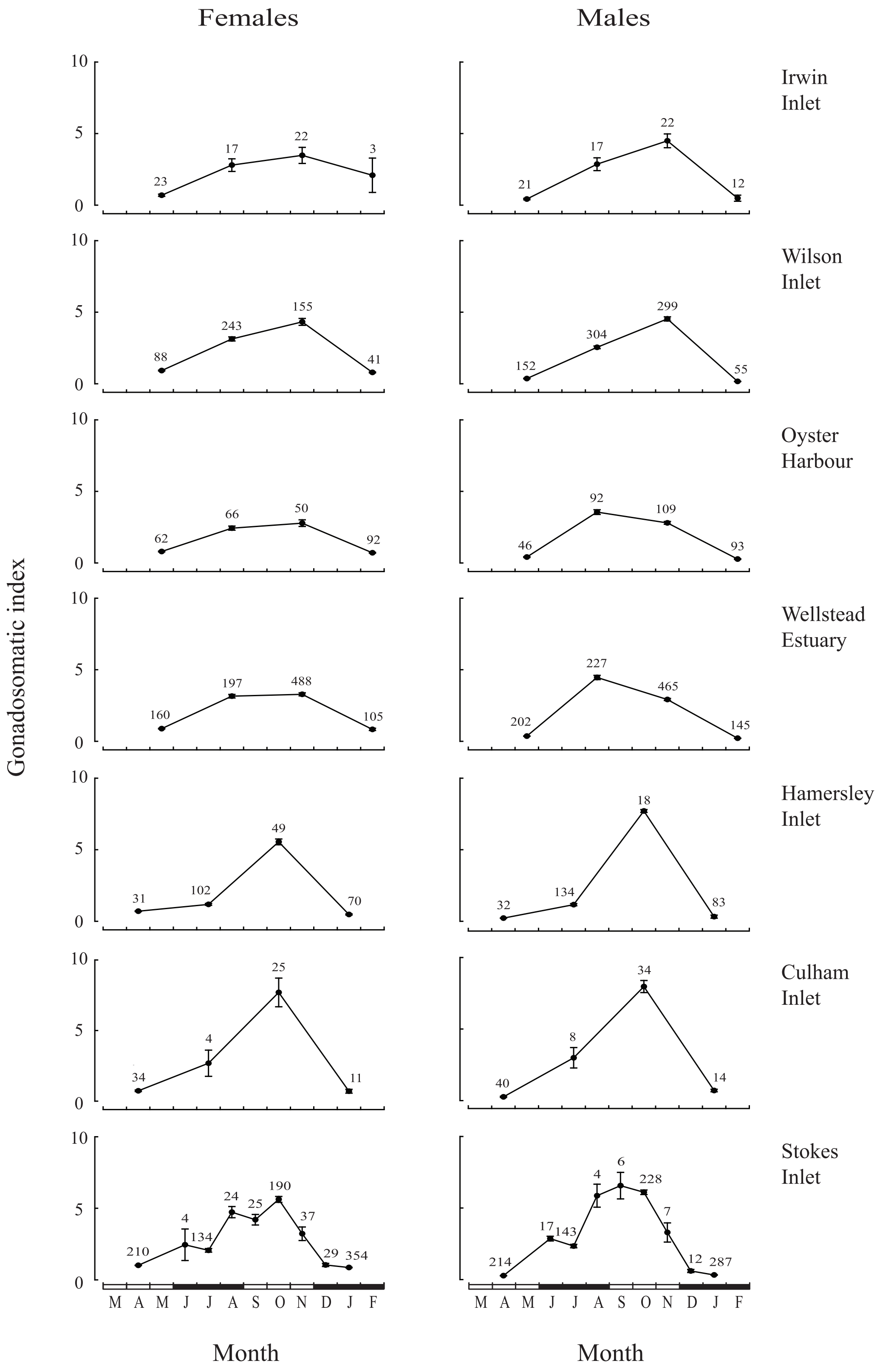

Fig. 4.6. Mean monthly gonadosomatic indices ± 1SE for female and male *Acanthopagrus butcheri* > L_{50} at sexual maturity from Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary in 2005-2007 and Hamersley, Culham and Stokes inlets in 2002-2004. Numbers of fish are shown above each mean. Closed bars on the x-axes represent winter and summer months and open bars represent autumn and spring months.

The ovaries of either the vast majority or all of the females of *A. butcheri* (> L_{50} at first maturity) in each estuary were at stage II (immature/resting) in autumn (Fig. 4.7). Fish with stage III (developing) ovaries were found in autumn and winter and those with stage IV (maturing) ovaries were recorded in winter and spring. Fish with stage V/VI (prespawning/spawning) ovaries were found in winter and spring and, collectively, were the dominant category in each estuary during spring. Stage VII (spent) and stage VIII (recovering/spent) ovaries were first recorded in fish in spring and, by summer, the ovaries of essentially all fish were either at stages VIII or II. The frequency of occurrence of the different stages in testicular development during the year followed essentially the same trend as that of ovaries (Fig. 4.7).

As the mean GSIs of female *A. butcheri* in each estuary, derived from data collected in one month in each season for three years, peaked in spring, spawning presumably also peaked in that season. Such a view is consistent with the fact that fish with ovarian stages V and VI dominated the October samples from Hamersley, Culham and Stokes inlets and fish with stage VII (spent) and VIII gonads dominating the November samples from Wilson Inlet, Oyster Harbour and Wellstead Estuary. Indeed, those latter data on the prevalence of ovarian stages suggest that spawning of this species typically peaks in October in south coast estuaries. It was thus considered appropriate to assign 1 October as the birth date for *A. butcheri* in these estuaries.

4.3.4. Gill net selectivity and mortality estimates

Analyses of mesh selectivity for *A. butcheri* indicated that stretched meshes of 35, 51, 63, 76, 89, 102 and 115 mm predominantly (*i.e.* for values of relative selectivity > 0.2) caught fish with lengths of 80-189, 100-239, 130-269, 170-309, 210-349, 250-379 and 290-409 mm, respectively (Fig. 4.8a). The values of θ_1 and θ_2, calculated using the method of Kirkwood and Walker (1986), were 3.10 and 1506.30, respectively. Trends in estimates of selectivity at age indicated that this species became fully susceptible to capture by the composite research gill net at an age of *ca* 2 years in each estuary and remained so throughout the rest of life, *e.g.* up to 15 years in Wilson Inlet (Fig. 4.8b), but declined slightly with age in some estuaries due to differences in growth.

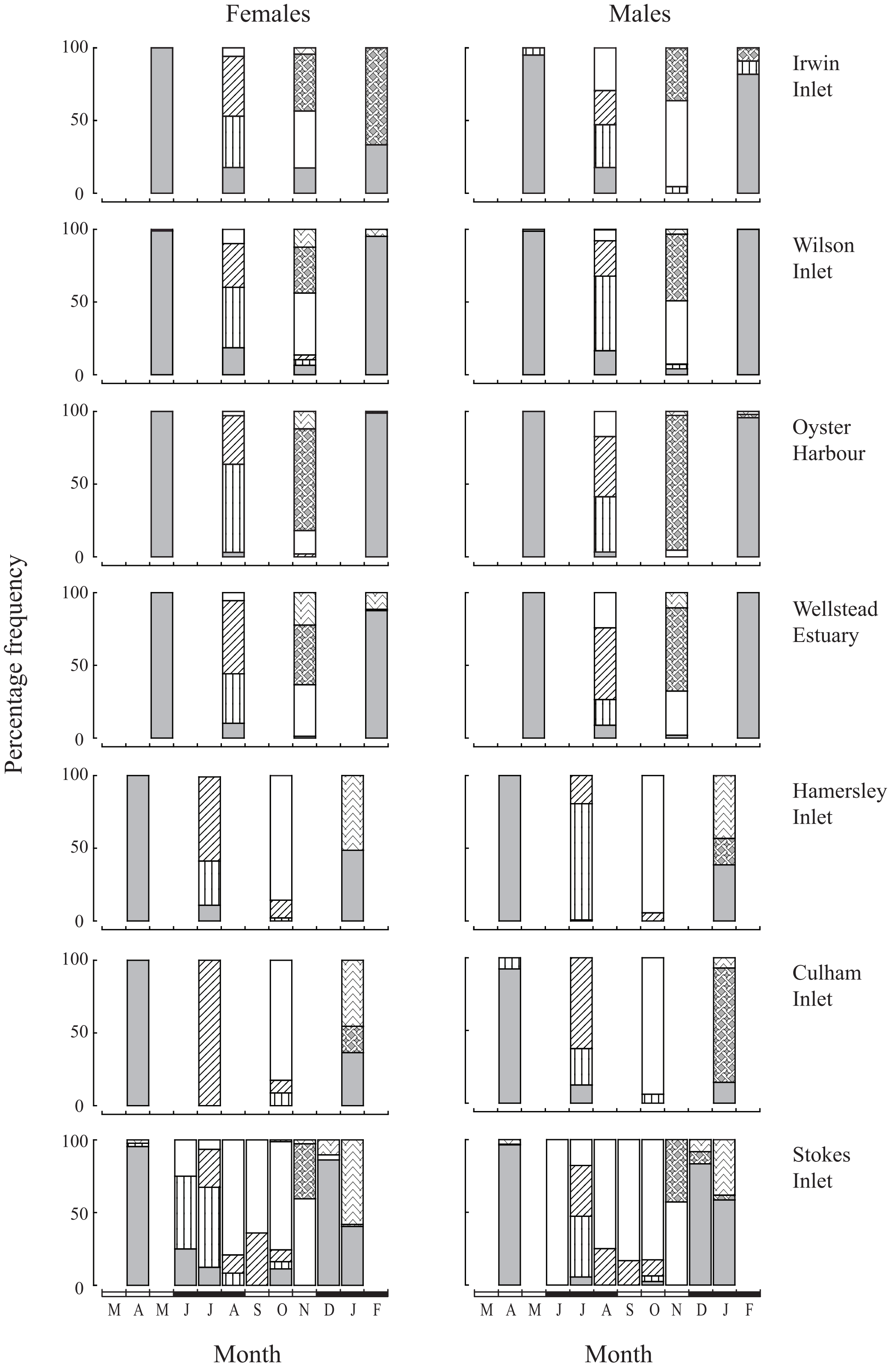

Fig. 4.7. Monthly percentage frequency of occurrence of gonads at different stages of development in female and male *Acanthopagrus butcheri* > L_{50} at sexual maturity from Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary in 2005-2007 and Hamersley, Culham and Stokes inlets in 2002-2004. The different stages of gonad development are ■ I&II ▥ III ▨ IV ☐ V&VI ▩ VII ▧ VIII

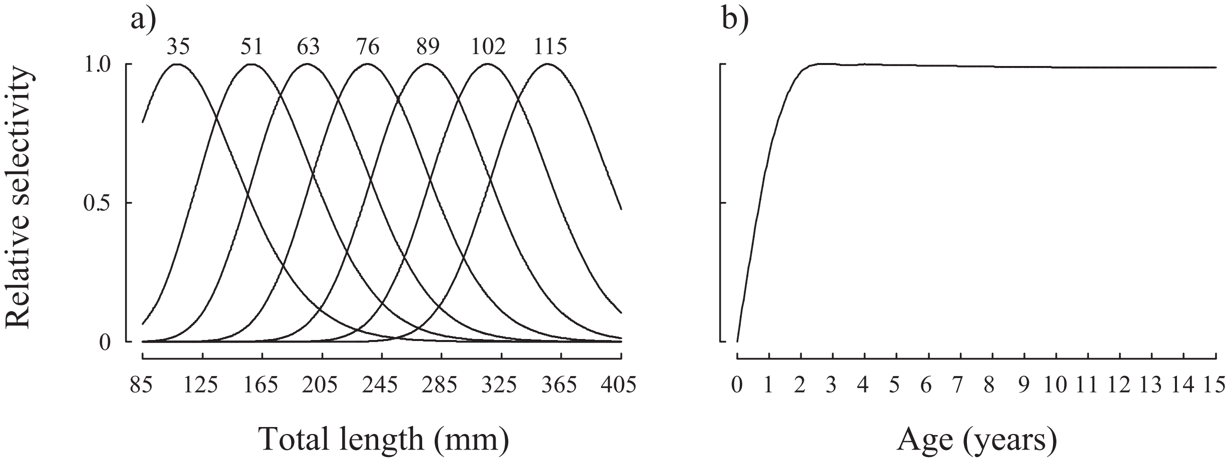

Fig. 4.8. Relative selectivity of *Acanthopagrus butcheri* captured by the composite research gill net in terms of a) total length in all estuaries combined and b) ages of fish in Wilson Inlet.

Although the estimates of fishing mortality (*F*) for *A. butcheri* were relatively imprecise, the point estimates suggested that this variable differed among estuaries. Thus, those estimates were highest for Oyster Harbour and Wilson Inlet and least for Wellstead Estuary, with that for Stokes Inlet being slightly higher than that for the latter estuary (Fig. 4.9, Table 4.4). Each of the point estimates for fishing mortality (*F*) was greater than that estimated for natural mortality (*M*) using Hoenig's (1983) equation, assuming that $Z = M + F$ and that *M* is constant, *i.e.* $M = 0.134$ year^{-1}, where the estimate of *M* is based on the maximum recorded age of 31 years for this species in Western Australia.

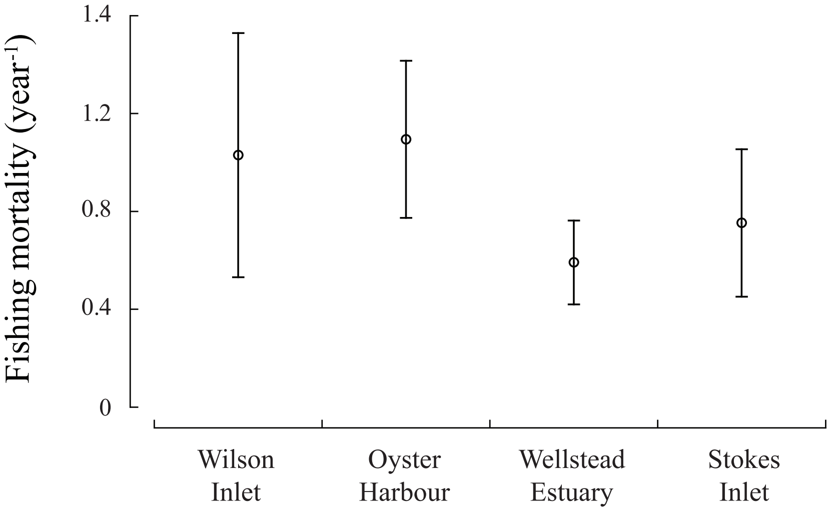

Fig. 4.9. Estimates of fishing mortality (*F*) and their 95% confidence intervals for *Acanthopagrus butcheri* in Wilson Inlet, Oyster Harbour, Wellstead Estuary and Stokes Inlet.

Table 4.4. Estimates of fishing mortality (*F*) ± 2 SE (given in parenthesis) and total mortality (*Z*) derived from relative abundance analyses for *Acanthopagrus butcheri* in Wilson Inlet, Oyster Harbour, Wellstead Estuary and Stokes Inlet.

	F (year^{-1})	Z (year^{-1})
Wilson Inlet	1.03 (0.53-1.53)	1.17
Oyster Harbour	1.09 (0.77-1.42)	1.23
Wellstead Estuary	0.59 (0.42-0.76)	0.73
Stokes Inlet	0.75 (0.45-1.05)	0.89

4.3.5. Variability in annual recruitment

The differences between the numbers of the different year classes in gill net samples from the various estuaries, together with differences in the estimates of the recruitment factor for those estuaries in which the numbers of individuals in the different year classes were sufficient to estimate this variable, demonstrated that recruitment varied markedly among years and estuaries (Fig. 4.10). Thus, on the basis of data from samples collected in 2005-2007, annual recruitment peaked in 1997 and 2002 in Wilson Inlet, 2001 in Oyster Harbour and 1998, 2002 and 2005 in Wellstead Estuary and, on the basis of data from sampling in 2002-2004, peaked in 2000 in Hamersley Inlet, 2001 in Culham Inlet and 1993 and 1998 in Stokes Inlet (Fig. 4.10).

Catches of 0+ *A. butcheri* obtained in 2005-2007 by seine netting in nearshore, shallow waters provided very strong evidence that the annual recruitment strength in Oyster Harbour and Wellstead Estuary varied (Table 4.5). Thus, individuals from the 2006 year class were more numerous than those from the 2005 year class in Oyster Harbour, whereas those of 2005 were most numerous in Wellstead Estuary. No 0+ individuals were captured in Wilson Inlet, indicating that no recruitment of this species occurred in that estuary in 2005 or 2006 and very low densities of 0+ *A. butcheri* from both of those year classes were recorded in Irwin Inlet. In the case of Stokes Inlet, sampling in 2002, 2003 and 2004 indicated that, while very large numbers of juvenile *A. butcheri* were recruited in 2003, no such recruitment occurred in 2002 (Table 4.5). No 0+ juveniles were recorded in any year in Hamersley Inlet, and while relatively high densities of the 2001 year class were recorded in Culham Inlet, recruitment of *A. butcheri* did not occur in that estuary in 2002 or 2003 (Table 4.5).

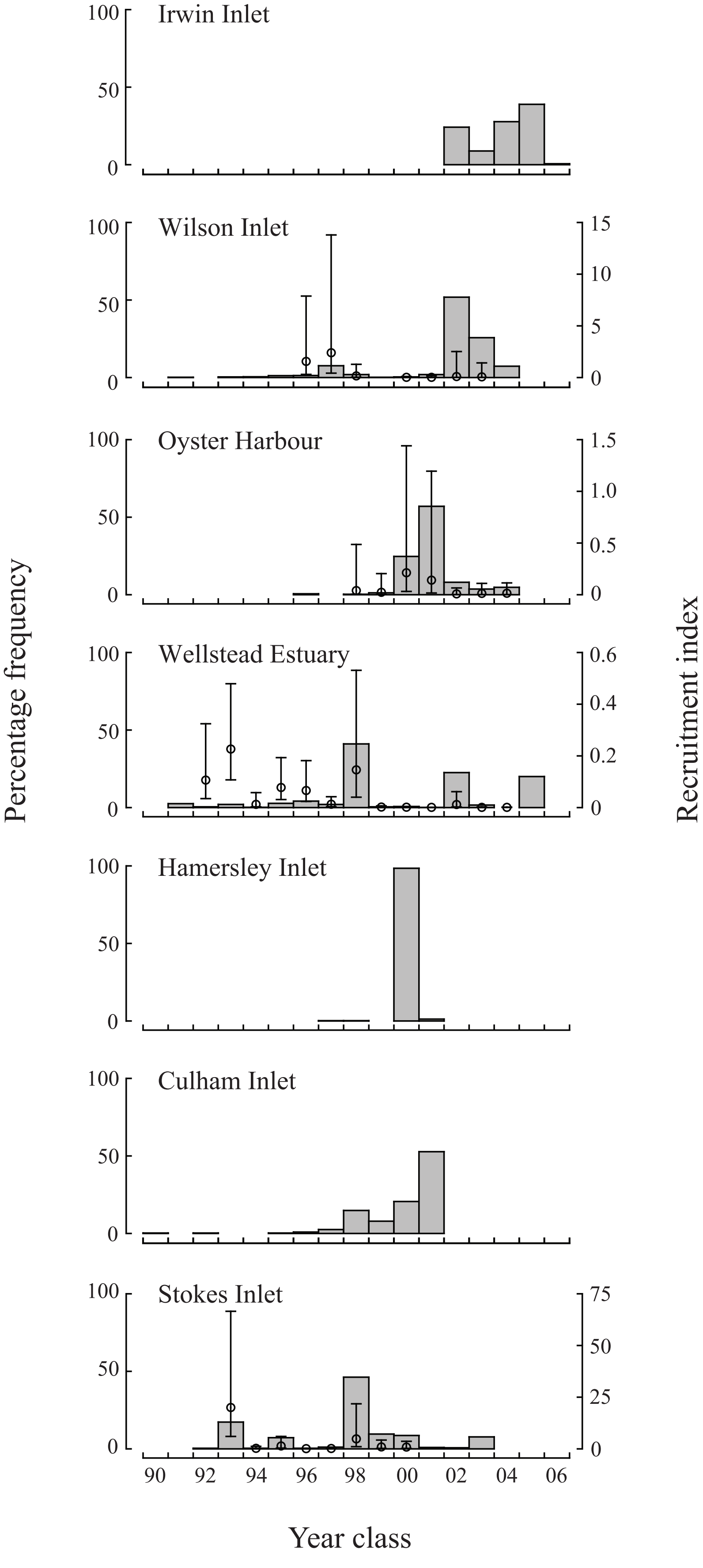

Fig. 4.10. Total numbers of each year class of *Acanthopagrus butcheri* captured with gill nets in each estuary (columns) and estimated recruitment indices (± 2 SE) for the various year classes in Wilson Inlet, Oyster Harbour, Wellstead Estuary and Stokes Inlet.

Table 4.5. Mean densities of 0+ *Acanthopagrus butcheri* in nearshore, shallow waters (fish 100 m^{-2}) representing the 2005 and 2006 year classes in the Broke, Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary and the 2001, 2002 and 2003 year classes in Hamersley, Culham and Stokes Inlets.

	2001	2002	2003	2005	2006
Broke Inlet	-	-	-	0	0
Irwin Inlet	-	-	-	0.1	0.1
Wilson Inlet	-	-	-	0	0
Oyster Harbour	-	-	-	0.2	2.9
Wellstead Estuary	-	-	-	20.5	3.5
Hamersley Inlet	0	0	0	-	-
Culham Inlet	19.0	0	0	-	-
Stokes Inlet	3.4	0	58.9	-	-

4.4. Discussion

4.4.1. Growth comparisons

This study on *Acanthopagrus butcheri* in estuaries on the south coast of Western Australia complements that of Sarre and Potter (2000) and further emphasises that the growth of this species varies markedly among estuaries in south-western Australia. An earlier laboratory experiment provided very strong circumstantial evidence that the marked difference between the early growth of *A. butcheri* in two estuaries was related to the very different environments of those estuaries rather than to genetic variability between those populations (Partridge *et al.*, 2003).

Sarre and Potter (2000) suggested that variations in the growth of *A. butcheri* among estuaries might be related to differences in water temperature and/or the density and/or dietary composition of this species. As the estuaries that were the subject of the current study were all located at a latitude of *ca* 34° S and thus have similar water temperature regimes, this environmental variable would appear to be an unlikely candidate for accounting for the marked differences in the growth of this species among south coast estuaries.

As the growth of *A. butcheri*, particularly during the first three years of life, was greatest in the estuaries in which the densities of this sparid were least, *i.e.* Irwin and Culham inlets, the faster growth in those two estuaries may be related to the presence of less intraspecific competition for either spatial and/or food resources. The possibility that the growth of *A. butcheri* is influenced by density is consistent with it being far less in the

Wellstead Estuary during the present study than it was *ca* 10 years earlier (see Sarre and Potter, 2000) when, during a previous study (Young and Potter, 2002), the catch rates of this species were far lower (data not shown).

In the context of the possibility that the quality and/or quantity of food are related to growth, it is relevant that the dietary composition of *A. butcheri* varies markedly among estuaries. This strongly suggests that this species is highly opportunistic in its feeding and indicates that it targets food sources that are abundant and/or of high nutritional value (Sarre *et al.*, 2000; Chuwen *et al.*, 2007). For example, the volumetric contributions of macrophytes to the diets of *A. butcheri* were far higher in the Moore River Estuary than in the Swan River Estuary, while the reverse was true for molluscs (Sarre *et al.*, 2000). The latter authors considered that these differences could have contributed to the growth in the early years of life being much slower in the former estuary. Likewise, *A. butcheri* consumed far greater volumes of macrophytes in Stokes Inlet than in Culham Inlet, in which greater volumes of benthic macroinvertebrates, teleosts and insects were ingested and the length at age was greater throughout the first six years and main part of life (Chuwen *et al.*, 2007).

Partridge and Jenkins (2002) recorded the growth of *A. butcheri* during the first four to six months of life in a range of salinities in the laboratory. Their results demonstrated that growth was greatest at a salinity of 24 and least at 60. These indications that salinity would be likely to influence the growth of *A. butcheri* in its natural habitat are consistent with the finding that growth was slowest overall in Stokes Inlet, in which the salinities remained elevated for a prolonged period but did not exceed 65, which approaches the lethal level of this species (Hoeksema *et al.*, 2006a). Although the growth of this species in Culham Inlet was particularly rapid early in life (Fig. 4.3), it should be recognised that these growth data were based on that recorded for samples collected during the first two seasons of the study, when salinities were lower than 65.

Given the confounding effects of the different factors affecting growth, the relative effects of those variables on the growth of *A. butcheri* are difficult to disentangle. The above comparisons indicate, however, that the growth of *A. butcheri* in an estuary is strongly influenced by one or more of the following variables; density of this species and the composition and abundance of the available food sources and the salinity regime of that estuary.

4.4.2. Otolith growth in different years

The results of the otolith-growth analyses were broadly consistent with the hypothesis that the greatest differences in inter-annual otolith growth would be detected in normally-closed estuaries in which interannual differences in environmental characteristics are most pronounced. Thus, inter-annual differences in the extent of otolith growth were particularly marked in the normally-closed Stokes Inlet, with growth being greatest in the growth year of 1998-99, *i.e.* between successive birth dates in 1998 and 1999, which was characterised by relatively high rainfall (Commonwealth Bureau of Meteorology) and thus presumably also relatively low salinities (< 35). It is not clear why year did not have a similar influence on the growth of *A. butcheri* in the normally-closed Wellstead Estuary, which was studied in a different period. Insufficient numbers of *A. butcheri*, due to massive mortalities resulting from exceptionally high salinities (Hoeksema *et al.*, 2006a), were available in the other normally-closed estuaries (Culham and Hamersley inlets) for analysis. The decline in the fourth annual growth increment for both sexes for 2004 in the permanently-open Oyster Harbour could not be explained.

Although the relationship between otolith radius and fish length appeared to vary among estuaries and sexes of *A. butcheri*, and thus constrained prediction of fish length from otolith radius and exploration of the direct influence of environmental factors on the growth of fish, it is highly likely that such growth will vary with those factors in a manner that corresponds to the variation exhibited by the growth of otoliths, *i.e.* years in which otolith growth is greatest are also likely to be those in which fish growth is greatest.

4.4.3. Reproductive biology

The reproductive data for *A. butcheri* in the estuaries for which sample sizes were substantial, *i.e.* Wilson Inlet, Oyster Harbour, Wellstead Estuary and Stokes Inlet, demonstrated that, while this species never matured at the end of the first year of life in any of those estuaries, between 73 and 100% did attain maturity by the end of their second year of life. Furthermore, the L_{50}s at first maturity in Wilson Inlet, Oyster Harbour, Wellstead Estuary and Stokes Inlet were not significantly different. It is thus relevant that, although the overall growth of *A. butcheri* varied markedly among the populations in the different estuaries, growth in the above four estuaries was similar in the first few years of life. Thus, the lengths at age 2 years, estimated from the von Bertalanffy growth equations for *A. butcheri* in each of those four estuaries, ranged only from 139 to 155 mm in the case of females and 133 to 164 mm for males and, the minimum recorded lengths for females and males after *ca* 2 years of growth in each estuary ranged only from 135 to 143 mm. In the Swan River Estuary on the lower west coast of Australia, maturity was likewise attained by the majority of individuals (85%) at the end of the second year of life (Sarre and Potter, 1999).

The L_{50}s at maturity of the females (145-161 mm) and males (138-164 mm) of *A. butcheri* in the four estuaries on the south coast of Western Australian for which there were substantial sample sizes, were far lower than the minimum legal length (MLL) of 250 mm for retention of this species in south-western Australia. Indeed, all *A. butcheri* that had reached 250 mm in these systems were mature and this even applied to the vast majority of those that had reached 200 mm. Thus, the current MLL for *A. butcheri* in Western Australia potentially allows all *A. butcheri* in south coast estuaries the opportunity to spawn before being legally retained. However, since the fecundity of *A. butcheri* increases exponentially with increasing body size (Sarre and Potter, 1999), the larger fish contribute more to egg production than smaller fish and thus heavy fishing pressure could have an impact on the abundance of this species in an estuary. This point is particularly relevant as *A. butcheri* are confined to their natal estuaries in south-western Australia (Chaplin *et al.*, 1998), and thus the stock in an estuary cannot be replenished naturally from another system.

4.4.4. Juvenile recruitment

The age compositions of gill net samples collected over two or three years demonstrated that the recruitment of *A. butcheri* in south coast Western Australian estuaries varies markedly among years. Since recent work in eastern Australia demonstrated that hypoxic conditions, particularly when accompanied by low salinities, lead to increased mortality of the eggs and early larvae of this species (Hassell *et al.*, 2008a, b; Nicholson *et al.*, 2008), low dissolved oxygen concentration and low salinity are also likely to influence the recruitment success of *A. butcheri* in south-western Australian estuaries.

The riverine regions of estuaries in the western part of the south coast, and particularly those that are seasonally open and thus have restricted tidal influence, are stratified throughout most of the year, with salinities being lower at the surface and dissolved oxygen concentrations being lower at the bottom of the water column (see Chapter 3). Thus, since *A. butcheri* spawns predominantly in the upper estuary (river), the presence of extremes in these conditions would presumably inhibit the survival of the eggs and early larvae of *A. butcheri* (see Newton, 1996; Haddy and Pankhurst, 2000; Hassell *et al.*, 2008a, b). It therefore appears relevant that the year classes of this species in Wilson Inlet were strongest in those years when rainfall was relatively low and thus stratification was less and salinities and dissolved oxygen concentrations were consequently higher. Such a relationship between reduced fluvial discharge and increased recruitment of *A. butcheri* in seasonally-open south-western Australian estuaries contrasts with the relationship between those variables in intermittently-open estuaries in Victoria, where decreased flows resulted in increased stratification and anoxia and thus reduced survival of early-life stages of this species (Nicholson *et al.*, 2008). Those authors attributed increased survival of those life stages to increased dissolved oxygen concentrations in the bottom water layer as a result of increased flow and subsequent breaching of the sand bar and tidal replenishment of oxygen reduced saline water.

In contrast to seasonally-open estuaries, the rivers in the normally-closed estuaries of the eastern region of the south coast of Western Australia are shallower and can remain hypersaline for extended periods. In these latter estuaries, high levels of recruitment of *A. butcheri* occurred in years when rainfall increased just prior to the spawning period or there were successive years of average rainfall and salinities were thus presumably not excessively high. Such a view is consistent with the fact that, following heavy rainfall and reduction in salinities to below that of full-strength sea water in 2005 in Wellstead Estuary, seine net catches of the juveniles of this species that hatched in that year were particularly high. Although the sand bar at the mouth of that estuary was breached in the autumn of 2005 and stayed open for the remainder of that year, riverine flow was not particularly strong during the spawning period of this species and would thus not have flushed the eggs and larvae of this species out to sea, as would potentially be the case if the sand bar was open and flow was particularly high during the spawning period (see Sarre and Potter, 2000).

The marked differences between the years in which recruitment of *A. butcheri* peaked among the normally-closed Hamersley, Culham and Stokes inlets are probably related, in part, to the highly localised rainfall that can occur in the drier eastern region of the south coast. However since massive mortalities of this species occurred in the former two estuaries (Hoeksema *et al.*, 2006a), direct comparisons of the year class compositions in those estuaries is problematic.

4.4.5. Mortality estimates

As annual recruitment of *A. butcheri* varied markedly in each estuary and some estuaries contained relatively few year classes, it was not appropriate to use traditional catch curve analyses to estimate the total mortality (Z) of this species in south coast Western Australian estuaries. Thus, relative abundance analyses of the various year classes were used for estimating F. Although, because of data limitations, this method provided relatively imprecise estimates of F, the point estimates for this variable suggested that fishing mortality was greatest in those estuaries in which commercial fishing is most intense (Wilson Inlet and Oyster Harbour) and least in Wellstead Estuary, which has been closed to commercial fishing for more than 20 years. Furthermore, although *A. butcheri* occurs mainly in the lower riverine reaches that constitute the upper estuary (*e.g.* Hindell *et al.*, 2008; Chuwen *et al.*, 2009b), and commercial fishing is prohibited in the rivers of Wilson Inlet and Oyster Harbour, it is relevant that the rivers of Wilson Inlet and more particularly Oyster Harbour are subjected to high levels of recreational fishing.

4.4.6. Conclusions

The results of this component of the study provide further evidence of the “plasticity” of *A. butcheri* and this is particularly well reflected in the marked variations in the growth of this species in the different estuaries. Such differential growth is likely to be attributable to one or more of the following factors; density of *A. butcheri*, quality/quantity of food and salinity. The ability of *A. butcheri* to adapt to a range of environmental conditions accounts for the success of this sparid in very different types of estuaries. Moreover, since the recruitment strengths of *A. butcheri* are also likely to be related to environmental factors, including salinity and dissolved oxygen concentrations, effective management of this species requires the integration of environmental and fisheries plans to maintain environmental conditions that will enhance the conservation of this iconic species.

5. Estuary Cobbler *Cnidoglanis macrocephalus*

5.1. Introduction

The Estuary Cobbler *Cnidoglanis macrocephalus*, which is represented by both estuarine and marine populations in south-western Australian waters (Laurenson *et al*., 1993a; Ayvazian *et al*., 1994), makes the greatest contribution to the overall value of the commercial estuarine fishery on the south coast of Western Australia (Smith and Brown, 2008). Although this plotosid (eeltail catfish) is endemic to Australia southwards of 28-29° S, it is far more abundant in the western than eastern part of Australia (Hutchins and Swainston, 1996; Hoese *et al*., 2006). It attains maximum total lengths (TL) and weights of over 900 mm and 2600 g (Hutchins and Swainston, 1996) and can live for 10 to 12 years (Laurenson *et al*., 1994). Studies during the late 1980s in Wilson Inlet, a large seasonally-open estuary on the south coast of Western Australia in which *C. macrocephalus* is the most important commercial fish species (Laurenson *et al*., 1993b; Smith and Brown, 2008), demonstrated that this plotosid rarely becomes mature at lengths less than 425 mm TL and three years of age and that its eggs and larvae are guarded by the males in excavated burrows until they have absorbed their yolk sac, which occurs at a TL of about 40 mm (Laurenson *et al*., 1993a, 1994). Despite marked differences between the environments in the seasonally-open Wilson Inlet and the permanently-open Swan River Estuary on the lower west coast of Australia, the growth of *C. macrocephalus* in these two estuaries in the past was very similar (Nel *et al*., 1985; Laurenson *et al*., 1994).

The trends exhibited by commercial catches indicate that the abundance of *C. macrocephalus* in some permanently-open estuaries on the lower west coast of Australia declined between the 1970s and 1990s (Steckis *et al*., 1995). It is thus almost certainly relevant that, during that period, the minimum legal length for retention (MLL) of this species was equivalent to a TL of 318 mm and thus *ca* 100 mm below the TL at which, at least in Wilson Inlet, the first individuals of this species begin to mature (Laurenson *et al*., 1993a). The appreciable catches of *C. macrocephalus* in Wilson Inlet in the late 1980s were assumed to be attributable, in part, to the beneficial effects to the stock of this species of the closure to commercial fishing of an area at the seaward end of the 48 km^2 basin of this estuary in the

1950s and within which the individuals were larger than in open fishing waters (Laurenson *et al.*, 1993a).

As the commercial fishery in Wilson Inlet is multi-species and multi-gear, and fishers shift their effort among species according to the availability and value of the various species, fishery-dependent catch per unit of effort (cpue) data cannot be used as a reliable measure of the relative abundance of individual species, such as *C. macrocephalus*. Although the annual commercial catches of *C. macrocephalus* in Wilson Inlet, which are obtained exclusively by gill netting, have varied markedly over the last 20 years, they have not shown a conspicuous overall decline or rise during that period (Smith and Brown, 2008). However, two reliable commercial fishers, who frequently target *C. macrocephalus* in Wilson Inlet, report that, during the past 20 years, they have had to increase their effort to maintain their catches of this species and have ceased to use the larger mesh sizes in their nets because net panels constructed using those mesh sizes no longer catch substantial numbers of *C. macrocephalus* (O. McIntosh and W. Miller, pers. comm.). The fact that *C. macrocephalus* is represented by discrete populations in south-western Australian estuaries (Ayvazian *et al.*, 1994) provides an excellent opportunity to establish a sampling regime that would obtain representative samples of this plotosid in these estuaries and thereby, through inter-estuarine and inter-period comparisons, enable the relationship between the biological characteristics of this species and selected environmental factors and fishing pressure to be explored.

A number of studies have shown that intense harvesting of a stock of a species can lead to rapid evolution of key life-cycle traits. For example, heavy exploitation of certain fish species in the north-western Atlantic was accompanied by declines in the age and length of the males and the age of the females of the porbeagle *Lamna nasus* at maturity (Cassoff *et al.*, 2007) and in the age and length of the cod *Gadus morhua* at maturity (Olsen *et al.*, 2004). The former workers also presented data that showed that heavy exploitation of *L. nasus* was accompanied by increased growth (Cassoff *et al.*, 2007). In contrast, a meta-analysis of 73 commercially-fished stocks, Hilborn and Minte-Vera (2008) found no evidence that the intensity of fishing was accompanied by changes in growth rate. They did recognise, however, that, as found by Brander (2007) using a meta-analysis of a number of cod stocks, environmental factors are probably very important in determining weight at age of fish.

Although their model suggested that heavy exploitation is unlikely to have a significant evolutionary impact on growth rates, they accepted that such impacts on other life-cycle traits, and particularly the size and age at maturity, are much more likely.

Sites throughout the permanently-open Oyster Harbour, seasonally-open Broke, Irwin and Wilson inlets and normally-closed Wellstead Estuary and Stokes Inlet on the south coast of Western Australia were sampled seasonally for either two or three years during 2005-2008 by using baited traps, seine nets and gill nets. The gill net catch rates and growth of *C. macrocephalus* in the various estuaries were compared to elucidate which environmental conditions are likely to be most suitable for this plotosid. In addition, the length (L_{50}) and age (A_{50}) of females at maturity and both fishing and total mortality were estimated for *C. macrocephalus* in each estuary in which it was abundant. The values for these mortalities were then used to ascertain whether the populations of this species in the different estuaries were likely to be experiencing deleterious effects due to fishing pressure. Our data on the catch rates and length and age compositions in the areas closed and open to fishing in Wilson Inlet, which are fishery-independent, were next compared to explore whether the closed area is still benefiting the stock of *C. macrocephalus* in this estuary. The catch rates, size and age compositions and mortality estimates, derived from our data for *C. macrocephalus* in Wilson Inlet in 2005-08, were compared with those derived for this species using the same methods in that estuary during 1987-89 (Laurenson *et al*., 1993a, 1994). These comparisons were used to test the hypothesis that, in Wilson Inlet, the abundance and prevalence of larger and older individuals in the population of *C. macrocephalus* has declined between these two periods and that the total mortality of this species has increased. The results of these inter-period comparisons and those between such variables as the L_{50} and A_{50} at maturity will be considered in the context of paradigms regarding the impact of heavy fishing pressure on fish stocks.

5.2. Materials and methods

The materials and methods that cover various general aspects of the study are given in the *General materials and methods* section, while details of those that relate specifically to *Cnidoglanis macrocephalus* are given below.

5.2.1. Catch rates among estuaries and between periods and regions in Wilson Inlet

The catch rate of *C. macrocephalus* at each site on each sampling day was expressed as the number of fish caught per night (*ca* 12 h) in the case of Irwin and Wilson inlets and Oyster Harbour, in which substantial numbers of this species were caught during the study. The catch rates at each site in Irwin and Wilson inlets and Oyster Harbour in each season in each of the two years between summer 2006 to spring 2007 were subjected to three-way analysis of variance (ANOVA). These analyses were used to determine whether catch rates were significantly related to estuary, year and/or season, each of which was considered a fixed factor. Prior to undertaking these and all subsequent ANOVAs involving catch rates, those rates were $\log_{10}(x+1)$ transformed. This form of transformation was shown to be appropriate from the relationships between the $\log_{10}$ of the standard deviation and the $\log_{10}$ of the mean catch rates of the replicate samples obtained seasonally from the basin of each estuary in each year (see Clarke and Warwick, 2001). All mean catch rates were back-transformed before being plotted or reported after being subjected to the bias correction of Rothery (1988). When there were significant interactions ($p < 0.05$), Scheffé's multiple comparison tests were used to determine whether the means were significantly different.

ANOVA was used to compare the catch rates of *C. macrocephalus* in replicate gill net samples from Wilson Inlet in each season between winter 2005 and autumn 2008 with those derived for the same mesh sizes and net length using the data collected by Laurenson *et al.* (1994) between winter 1987 and autumn 1989. Note that these comparisons were restricted to the data recorded for same six sites that were sampled in both periods (see Fig. 2.1). These analyses were used to determine whether the gill net catch rates in Wilson Inlet were significantly related to period, season and region, which were each considered fixed factors, and years (winter to autumn), where year was considered a random factor nested within period.

ANOVA was next used to examine the relationships between the catch rates of both all *C. macrocephalus* and of those > MLL in waters open and closed to fishing in Wilson Inlet in 2005-08. These analyses employed data for the three and seven sites sampled in closed and open waters, respectively (Fig. 2.1), in each 12 month period (year) between winter 2005 and autumn 2008. ANOVA was finally used to examine whether the TL of *C. macrocephalus* was related to region (open *vs* closed fishing waters) using measurements made on the same fish that were used to derive the above catch rates for all fish. Note that plots of the $\log_{10}$ of the standard deviation and the $\log_{10}$ of the mean TLs in the replicate samples obtained seasonally in each year demonstrated that the TLs should be square root transformed prior to subjection to ANOVA (see Clarke and Warwick, 2001). For all of the above analyses, region, season and year were considered fixed factors.

5.2.2. *Growth*

Since previous work on *C. macrocephalus* indicated that the Schnute growth curve provided significantly better statistical fits than the von Bertalanffy growth curve to the lengths at age of this species (Laurenson *et al.*, 1994), Schnute growth curves were also fitted to the length-at-age data collected during the present study in Irwin and Wilson inlets and to those collected by Laurenson *et al.*, (1994) in Wilson Inlet in 1987-89. The Schnute growth equation has four possible forms depending on the value of the parameters *a* and *b*, *i.e.*

1. $a \neq 0, b \neq 0$ $$L_t = \left[y_1^b + \left(y_2^b - y_1^b\right)\frac{1-\exp(-a(t-T_1))}{1-\exp(-a(T_2-T_1))}\right]^{1/b},$$

2. $a \neq 0, b = 0$ $$L_t = y_1 \exp\left[\log\left(\frac{y_2}{y_1}\right)\frac{1-\exp(-a(t-T_1))}{1-\exp(-a(T_2-T_1))}\right],$$

3. $a = 0, b \neq 0$ $$L_t = \left[y_1^b + \left(y_2^b - y_1^b\right)\frac{t-T_1}{T_2-T_1}\right]^{1/b},$$

4. $a = 0, b = 0$ $$L_t = y_1 \exp\left[\log\left(\frac{y_2}{y_1}\right)\frac{t-T_1}{T_2-T_1}\right],$$

where y_1 and y_2 are the total lengths at the specified ages T_1 and T_2 (*i.e.* ages 1 and 4 years, which encompass the majority of the length-at-age data collected in this study). The same likelihood-ratio test that was described in the *General materials and methods* section was used to compare the von Bertalanffy and Schnute growth curves for *C. macrocephalus* in Irwin and Wilson inlets, noting that the former curve is nested within the latter curve (*i.e.* when $a > 0$, $b = 1$).

5.3. Results

5.3.1. *Comparisons between estuaries*

The catches of *Cnidoglanis macrocephalus* in Irwin and Wilson inlets far exceeded those obtained from Oyster Harbour and Wellstead Estuary, while only two individuals were caught in Stokes Inlet and none was collected from Broke Inlet. Three-way ANOVA, using data for the first three estuaries, demonstrated that the catch rates differed significantly among those estuaries and with season and that there was a significant interaction between estuary and year (Table 5.1). The mean square was greater for estuary than for both season and the interaction (Table 5.1). Overall, the mean catch rate (after back transformation) in Irwin Inlet (9.3 fish 12 h^{-1}, CI = 6.8-12.6 fish 12 h^{-1}) was greater than in Wilson Inlet (5.0 fish 12 h^{-1}, CI = 3.7-6.5 fish 12 h^{-1}), which in turn was greater than in Oyster Harbour (3.2 fish 12 h^{-1}, CI = 2.1-4.7 fish 12 h^{-1}). The interaction reflected the fact that, whereas the mean catch rate in Irwin Inlet was far greater in 2007 than 2006, they were similar in each year in both Wilson Inlet and Oyster Harbour (Fig. 5.1). The mean catch rates in the three estuaries declined from their maxima in summer to their minima in winter and rose again in spring (Fig. 5.1).

Table 5.1. *F*-statistics, mean squares (given in parenthesis) and significance levels for three-way ANOVAs of the catch rates of *Cnidoglanis macrocephalus* in composite gill nets in Irwin and Wilson inlets and Oyster Harbour in each season of 2006 and 2007. ** $p < 0.01$, *** $p < 0.001$.

	Main effects			
	Estuary (E)	Year (Y)	Season (S)	Residual
df	2	1	3	160
Catch rate	9.928*** (2.019)	3.542 (0.720)	3.997** (0.809)	 (0.203)
	Interactions			
	E x Y	E x S	Y x S	E x Y x S
df	2	6	3	6
Catch rate	5.424** (1.103)	1.362 (0.277)	1.560 (0.317)	1.684 (0.342)

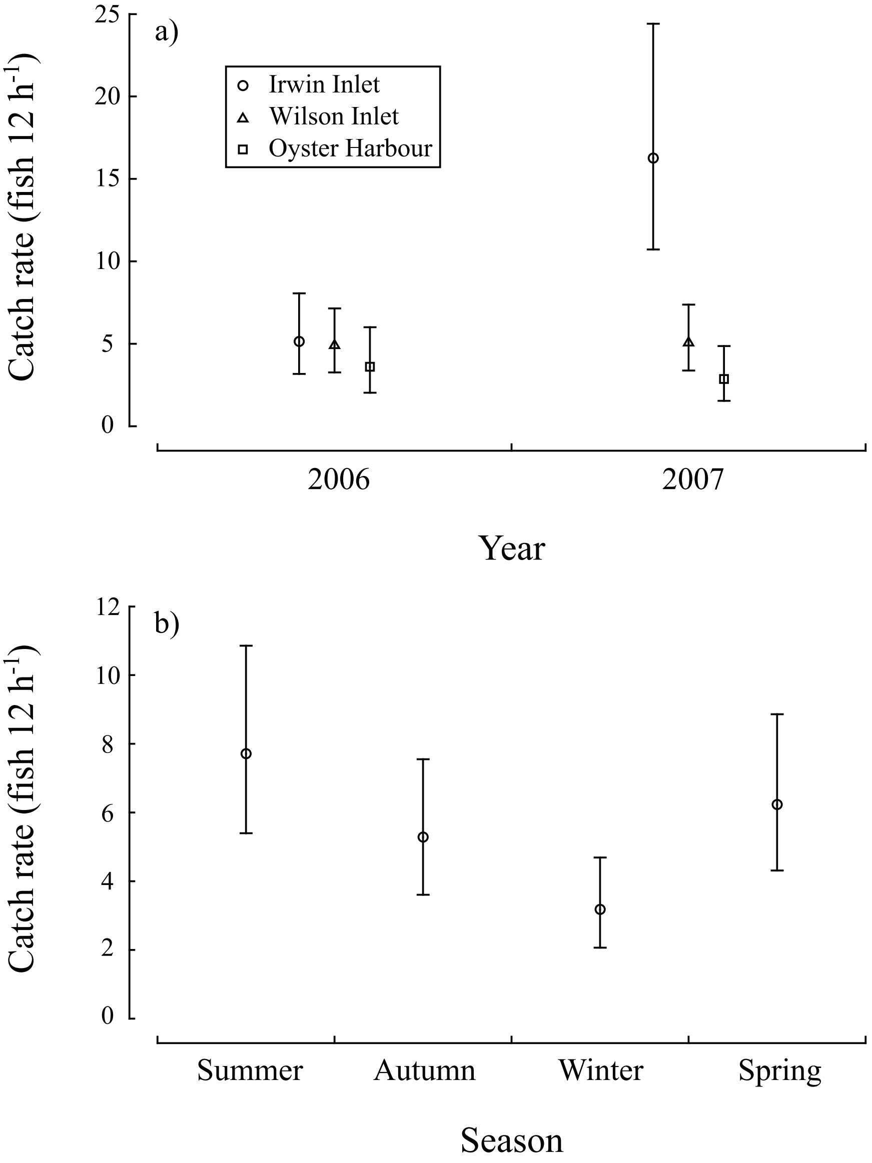

Fig. 5.1. Back-transformed mean catch rates ± 95% confidence intervals for *Cnidoglanis macrocephalus* in Irwin and Wilson inlets and Oyster Harbour in a) 2006 and 2007 and b) each season collectively for all estuaries.

The *C. macrocephalus* caught with gill nets in Irwin and Wilson inlets covered a similar length range, *i.e. ca* 100 to 700 mm, and contained a similar percentage of fish > 430 mm, the MLL for retention of this species, *i.e.* 26 and 30%, respectively (Fig. 5.2). The length-frequency distributions tended to peak more sharply in Wilson Inlet than in Irwin Inlet. The fish in both estuaries belonged predominantly to the 1+ to 3+ age classes and none was older than 5+ and 6+ years, respectively (Fig. 5.2). However, the modal age class was 1+ in Irwin Inlet, but with substantial numbers of 2+ fish, whereas the modal age class in Wilson Inlet was 2+ and that age class dominated the samples to the extent that it contributed more than 2.5 times the number of individuals of either the 1+ or 3+ age classes (Fig. 5.2). The individuals from the far smaller samples caught in Oyster Harbour and Wellstead Estuary ranged in length from *ca* 160-580 and 260-660 mm, respectively, and in age class from 1+ to 5+ (Fig. 5.2).

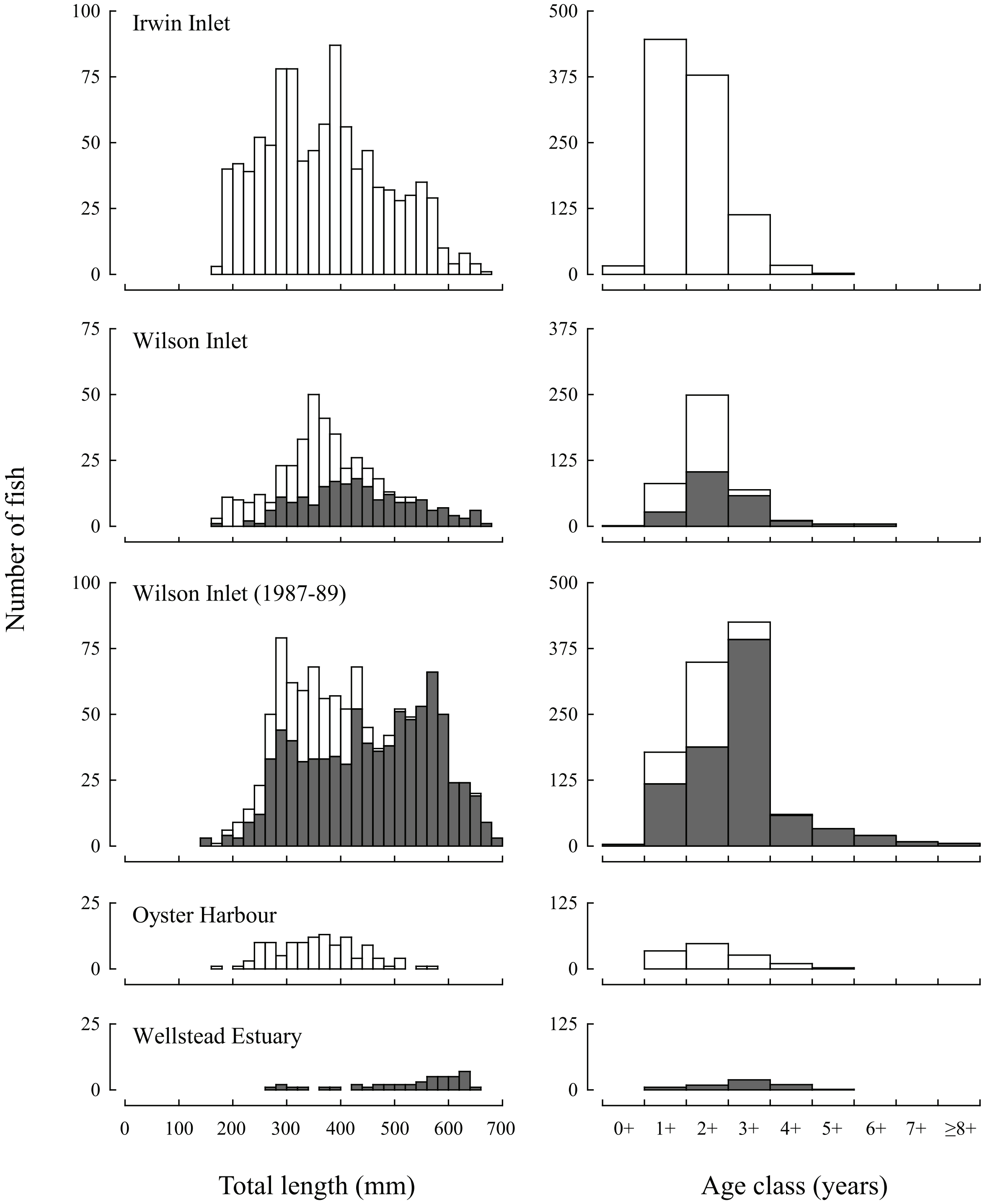

Fig. 5.2. Length and age class frequency distributions for *Cnidoglanis macrocephalus* caught with gill nets in Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary in 2005-08 and in Wilson Inlet in 1987-89. White and grey bars denote fish caught in waters open and closed to commercial fishing, respectively.

Comparisons of the von Bertalanffy and Schnute growth curves for the females and males of *C. macrocephalus* demonstrated that those curves were significantly different (each $p < 0.05$) in each of the Irwin and Wilson inlets, for which there were sufficient data to fit separate von Bertalanffy and Schnute curves for the females and males. However, as those differences were not considered biologically significant, *i.e.* the estimated lengths at each integer age (> 0) never exceeded 5%, the length-at-age data for the two sexes in each system were pooled. Although the likelihood-ratio test demonstrated that the von Bertalanffy and Schnute growth curves for the pooled sexes were significantly different in the Irwin and Wilson inlets (each $p < 0.001$), the differences were also not considered biologically significant. Both growth curves thus provided good fits to the lengths at age of *C. macrocephalus* in the Irwin and Wilson inlets, while the von Bertalanffy growth curve also provided good fits to the length-at-age data for Oyster Harbour and Wellstead Estuary, for which sample sizes were only 123 and 45, respectively (Fig. 5.3, Table 5.2). The growth of *C. macrocephalus* in each of the four estuaries was significantly different from that in each other estuary (all $p < 0.001$). The lengths at each age > *ca* 1.5 years were greatest in Irwin Inlet and Wellstead Estuary, followed by Wilson Inlet and then Oyster Harbour (Fig. 5.3). This point is emphasised by the fact that, on the basis of the von Bertalanffy growth curves, the estimated respective lengths at ages 3 and 4 years, for which there were by far the most data, were 493 and 600 mm in Irwin Inlet, 451 and 539 mm in Wilson Inlet and only 384 and 442 mm in Oyster Harbour (Fig. 5.3). These differences each exceeded 5% and were thus considered biologically significant.

In the two estuaries from which substantial numbers and a wide length range of *C. macrocephalus* were obtained during the spawning period, the minimum length of mature females was 443 mm in Irwin Inlet and 410 mm in Wilson Inlet and all females > 520 and 537 mm in these two estuaries, respectively, were mature (Fig. 5.4). The estimated L_{50} for female *C. macrocephalus* at maturity in Irwin Inlet (470 mm) was significantly different from the corresponding estimate (417 mm) for this species in Wilson Inlet ($p < 0.001$) (Table 5.3). The number of larger males caught during the spawning period was very low because, at this time, males tend to occupy burrows, within which they brood eggs and yolk-sac larvae under their pelvic fins (Laurenson *et al.*, 1993a). It was thus not possible to derive reliable estimates

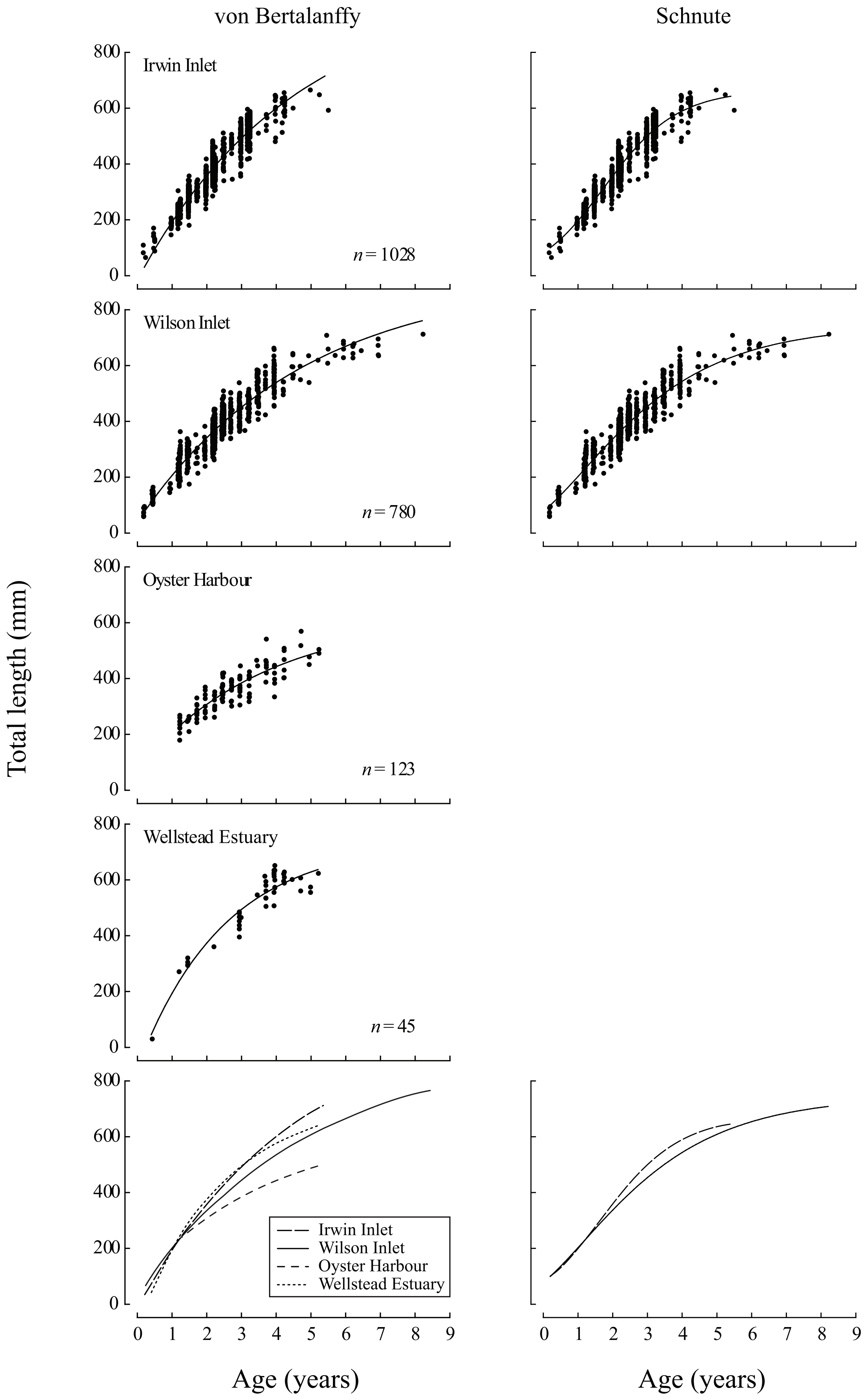

Fig. 5.3. von Bertalanffy growth curves fitted to lengths at age of *Cnidoglanis macrocephalus* from Irwin and Wilson inlets, Oyster Harbour and Wellstead Estuary and Schnute growth curves fitted to lengths at age of fish from Irwin and Wilson inlets. *n* = number of fish.

for the L_{50} for males. None of the 1+ or 2+ females of *C. macrocephalus* caught collectively in Irwin and Wilson inlets in 2005-07 was mature during the spawning period of this plotosid (Fig. 5.4). The percentages of females that were mature at the end of their third and fourth years of life were slightly greater in the samples from Irwin Inlet than Wilson Inlet during the same period (2005-07). All of the limited numbers of 5+ and 6+ female fish that were caught during the spawning period in Irwin and Wilson inlets were mature (Fig. 5.4). The estimated A_{50} for female *C. macrocephalus* at maturity in Irwin Inlet (3.1 years) was not significantly different from the corresponding estimate (3.0 years) for this species in Wilson Inlet ($p > 0.05$) (Table 5.3).

Table 5.2. von Bertalanffy growth parameters and their 95% confidence intervals (CI) derived from lengths at age of *Cnidoglanis macrocephalus* caught in Irwin Inlet, Wilson Inlet, Oyster Harbour and Wellstead Estuary. L_∞, asymptotic length (mm); *k*, growth coefficient (y^{-1}); t_0, hypothetical age at which fish would have zero length (y); R^2, coefficient of determination; *n*, sample sizes; L_{max}, maximum lengths (mm); W_{max}, maximum weights (g) and A_{max} maximum ages (y).

						von Bertalanffy parameters			
		n	L_{max}	W_{max}	A_{max}	L_∞	*k*	t_0	R^2
Irwin Inlet		1028	665	1584	5	1031.5	0.22	0.06	0.91
	Lower CI					921.0	0.18	-0.01	
	Upper CI					1142.0	0.23	0.14	
Wilson Inlet		780	712	2576	8	915.1	0.21	-0.21	0.90
2005-08	Lower CI					849.1	0.18	-0.29	
	Upper CI					981.1	0.24	-0.13	
Wilson Inlet		1289	722	2188	12	808.6	0.25	0.01	0.76
1987-89	Lower CI					762.4	0.22	-0.13	
	Upper CI					854.9	0.28	0.13	
Oyster Harbour		123	569	858	5	627.2	0.27	-0.45	0.78
	Lower CI					453.3	0.09	-1.12	
	Upper CI					801.1	0.46	0.22	
Wellstead Estuary		45	651	1793	5	736.5	0.40	0.24	0.90
	Lower CI					617.1	0.23	-0.06	
	Upper CI					855.9	0.57	0.53	

Analysis of the mesh selectivity of the composite gill net for *C. macrocephalus* indicated that stretched meshes of 35, 51, 63, 76, 89, 102 and 115 mm predominantly caught fish with lengths of 160-379, 190-479, 200-549, 270-619, 410-679, 490-709 and 560-709 mm, respectively, *i.e.* those with relative selectivity values of > 0.2, (Fig. 5.5a). The values of θ_1 and θ_2, calculated using the method of Kirkwood and Walker (1986), were 6.08 and 6801.95, respectively. Trends in estimates of selectivity at age indicated that, in both Irwin and Wilson inlets, this species became fully susceptible to capture by the above suite of mesh sizes at *ca* 2 years of age and essentially remained so in the immediately ensuing years (Fig. 5.5b).

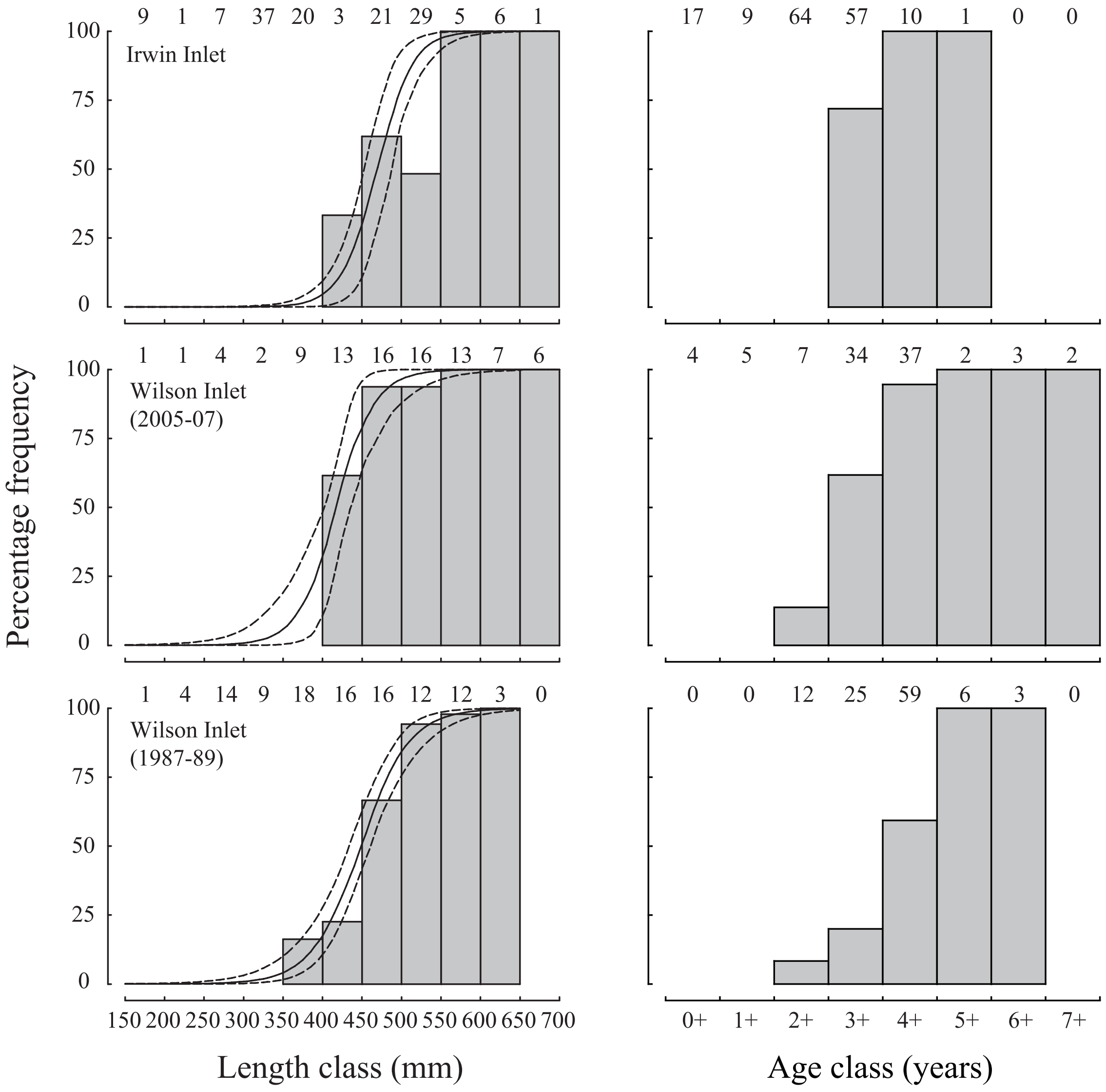

Fig. 5.4. Percentage frequencies of occurrence of females with ovaries at one of stages III to VIII in sequential 50 mm length classes and sequential age classes of *Cnidoglanis macrocephalus* during the spawning period. Logistic curves (solid lines) and their 95% confidence intervals (dotted lines) were derived from logistic regression analyses that described the relationship between total length and the probability that an individual fish possessed gonads at one of stages III to VIII. Sample sizes in each length class are shown.

Table 5.3. Lengths (L_{50} and L_{95}) and ages (A_{50} and A_{95}) at first maturity and their 95% confidence intervals (in parenthesis) of female *Cnidoglanis macrocephalus* in Irwin Inlet in 2005-07 and Wilson Inlet in 2005-07 and 1987-89.

	L_{50} (mm)	L_{95} (mm)	A_{50} (years)	A_{95} (years)
Irwin Inlet	470 (453-488)	535 (507-558)	3.0 (2.9-3.1)	4.0 (3.9-4.1)
Wilson Inlet 2005-07	417 (402-436)	490 (448-538)	3.1 (2.9-3.2)	4.1 (4.0-4.8)
Wilson Inlet 1987-89	449 (434-461)	540 (516-571)	3.5 (3.4-3.6)	4.7 (4.5-5.1)

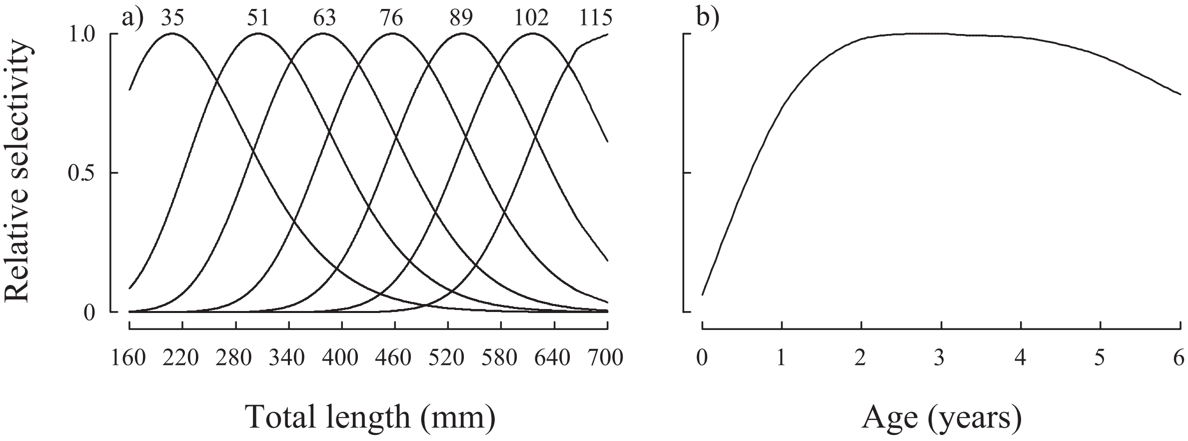

Fig. 5.5. Relative selectivity of *Cnidoglanis macrocephalus* caught by the composite gill net for a) total length in all estuaries combined and b) ages of fish in Wilson Inlet.

The estimates of fishing mortality (*F*), derived for *C. macrocephalus* in 2005-08, and taking into account differences in annual recruitment, were 0.77 year^{-1} for Irwin Inlet and 1.47 year^{-1} for Wilson Inlet (Fig. 5.6, Table 5.4). Natural mortality (*M*) was calculated to be 0.35 year^{-1}.

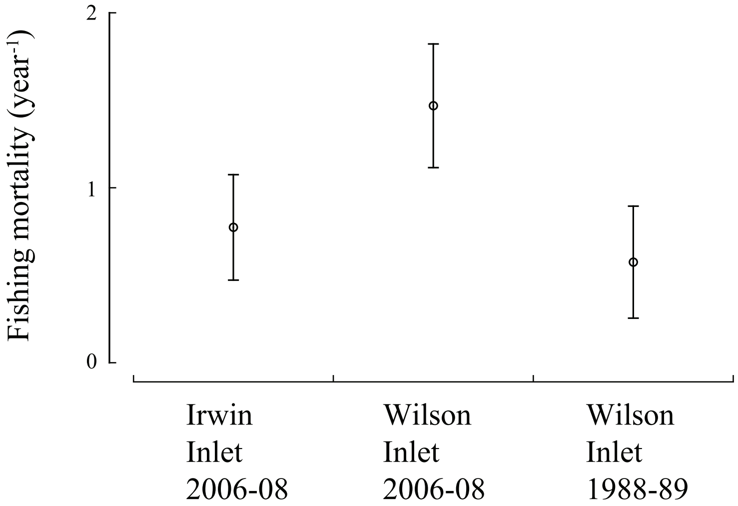

Fig. 5.6. Fishing mortality estimates (*F*) ± 2 SE derived from relative abundance analyses for *Cnidoglanis macrocephalus* in Irwin and Wilson inlets in 2006-08 and Wilson Inlet in 1988-89.

Table 5.4. Estimates of fishing mortality (*F*) ± 2 SE (given in parenthesis) and total mortality (*Z*) derived from relative abundance analyses for *Cnidoglanis macrocephalus* in Irwin and Wilson inlets in 2006-08 and Wilson Inlet in1988-89.

	F (year^{-1})	*Z* (year^{-1})
Irwin Inlet	0.77 (0.47-1.07)	1.12
Wilson Inlet (2006-08)	1.47 (1.11-1.82)	1.82
Wilson Inlet (1988-89)	0.57 (0.25-0.89)	0.92

5.3.2. Comparisons between waters open and closed to commercial fishing in Wilson Inlet

Three-way ANOVA demonstrated that the catch rates of *C. macrocephalus* in Wilson Inlet in 2005-08 were significantly related to region (*i.e.* waters open and closed to fishing) and season and that there was a significant interaction between region and year (Table 5.5). The mean square was far greater for region than for either of the other two effects (Table 5.5). The mean overall catch rate was greater in the region of the estuary closed to fishing (7.6 fish 12 h^{-1}, CI = 5.0-11.4 fish 12 h^{-1}) than open to fishing (2.2 fish 12 h^{-1}, CI = 1.5-3.1 fish 12 h^{-1}). The interaction reflected, in part, the fact that, in the third year, the mean catch rate rose in closed waters but declined slightly in open waters, with the catches in that year strongly influencing the overall difference between the catch rates of the two regions (Fig. 5.7a). The mean catch rate was greatest in summer and least in winter (Fig. 5.7b).

ANOVA demonstrated that the catch rates of *C. macrocephalus* with TLs > the MLL for retention (430 mm) in open and closed waters (region) differed significantly and that there were significant two-way interactions between year and season and between region and year and a significant three-way interaction between region, year and season (Table 5.5). The mean square for region was far higher than for any of the interactions. These results reflected the fact that the mean catch rates were greater in closed than open waters in 10 of the 12 consecutive seasons sampled and markedly so in some seasons (Fig. 5.8a), a trend that accounts for the overall mean catch rate in closed waters (3.1 fish 12 h^{-1}, CI = 2.9-4.5 fish 12 h^{-1}) being far greater than in open waters (0.5 fish 12 h^{-1}, CI = 0.3-0.7 fish 12 h^{-1}).

ANOVA further demonstrated that the TL of *C. macrocephalus* differed significantly with region and year and that there was a significant three-way interaction between region, year and season (Table 5.5). The mean square was far higher for region than for year, which in turn was greater than for the interaction. This reflects the fact that the mean seasonal TLs were greater in closed than open waters in all but one season and consequently that the overall mean TL in closed waters (438 mm, CI = 417-460 mm) was far greater than in open waters (351 mm, CI = 324-379 mm) (Fig. 5.8b).

Table 5.5. *F*-statistics, mean squares (given in parenthesis) and significance levels for three-way ANOVAs of overall catch rates, catch rates of individuals > minimum legal length for retention (MLL) and total lengths of *Cnidoglanis macrocephalus* in composite gill nets in the regions open and closed to fishing in Wilson Inlet in each season of 2006 and 2007. ** $p < 0.01$, *** $p < 0.001$.

	Main effects			
	Year (Y)	Season (S)	Region (R)	Residual
df	2	3	1	48
Catch rate (overall)	0.495	5.315**	20.760***	
	(0.072)	(0.770)	(3.008)	(0.145)
Catch rate (> MLL)	2.897	0.846	59.193***	
	(0.154)	(0.045)	(3.144)	0.053
Total length	6.704***	2.594	22.894***	
	(27.637)	(10.692)	(94.382)	(4.123)
	Interactions			
	Y x S	Y x R	S x R	Y x S x R
df	6	2	3	6
Catch rate (overall)	1.535	6.077**	1.094	0.700
	(0.222)	(0.881)	(0.158)	(0.101)
Catch rate (> MLL)	4.461***	6.415**	0.899	4.204**
	(0.237)	(0.341)	(0.048)	(0.223)
Total length	1.195	2.802	2.541	3.599**
	(4.928)	(11.551)	(10.475)	(14.837)

5.3.3. Historical comparisons in Wilson Inlet

Analyses of catch rates for each region and season in the two periods (2005-08 *vs* 1987-89) demonstrated that catch rate was significantly related only to period, recognising that, although the effect of region was not significant ($p = 0.052$), the mean square value for this factor was relatively high (Table 5.6). The mean overall catch rate of 4.4 fish 12 h^{-1} (CI = 3.3-5.8 fish 12 h^{-1}) for *C. macrocephalus* in Wilson Inlet in 2005-08 was far lower than the 14.9 fish 12 h^{-1} (CI = 11.0-19.9 fish 12 h^{-1}) for the same species in 1987-89.

While the length range of *C. macrocephalus* caught in Wilson Inlet in 2005-08 and 1987-89 was similar, the length-frequency distribution in the earlier period tended to be bimodal, with a second strong modal length class in waters closed to commercial fishing occurring at 560-579 mm and thus at a length exceeding that of the vast majority of fish caught in the later period (Fig. 5.2). While the 2+ age class dominated the samples in 2005-07, the 3+ age class, followed closely by the 2+ age class, made the greatest contribution to those collected in 1987-89 (Fig. 5.2). The contribution of *C. macrocephalus* ≥ 4+ years in 1987-89 far exceeded that recorded in 2005-08 (Fig. 5.2).

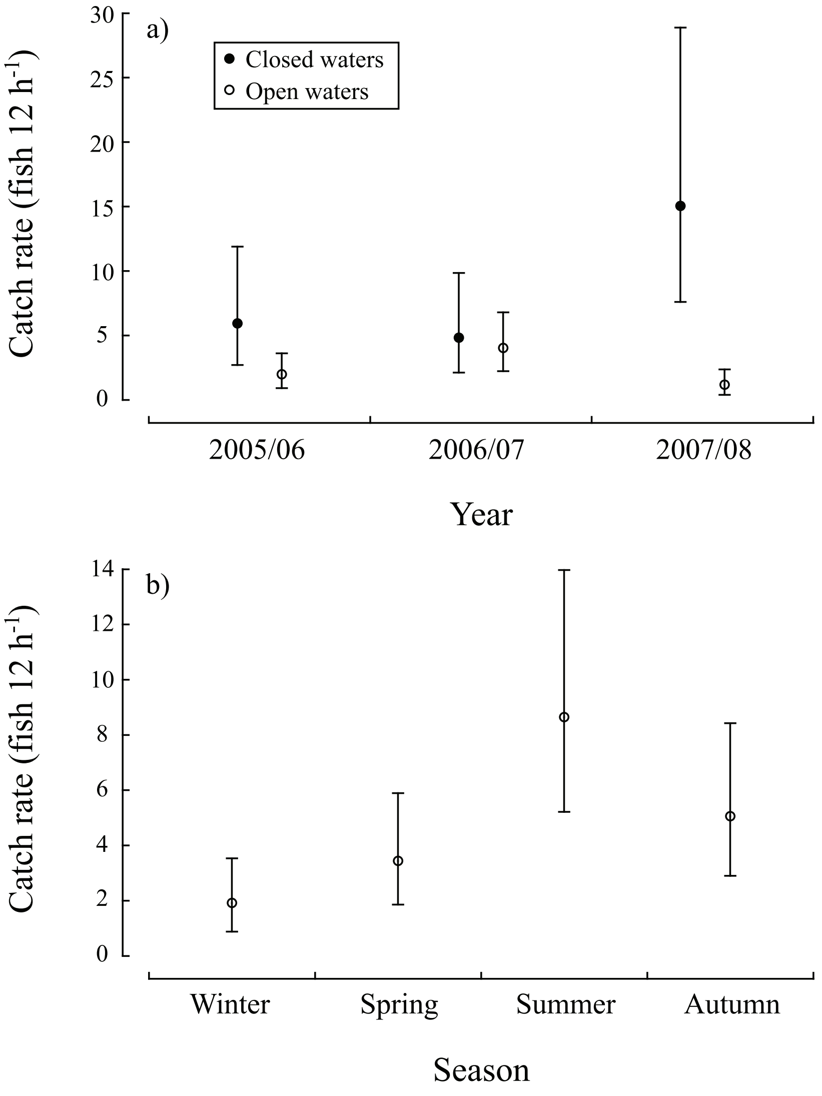

Fig. 5.7. Back-transformed mean catch rates ± 95% confidence intervals of *Cnidoglanis macrocephalus* for a) closed and open fishing waters in Wilson Inlet in 2005/06, 2006/07 and 2007/08, and b) for each season collectively in Wilson Inlet.

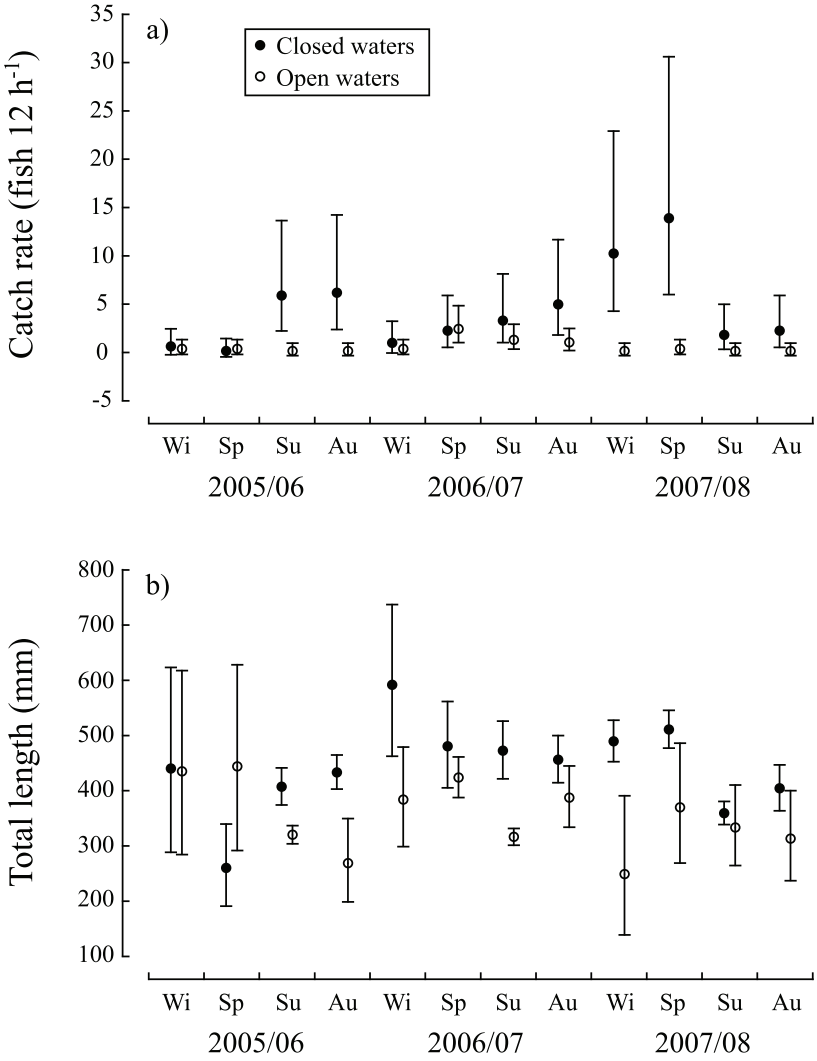

Fig. 5.8. Back-transformed mean values ± 95% confidence intervals for a) catch rates of *Cnidoglanis macrocephalus* > MLL and b) total lengths of *Cnidoglanis macrocephalus* caught in closed and open fishing waters in each season of 2005/06, 2006/07 and 2007/08.

The likelihood-ratio test demonstrated that the von Bertalanffy and Schnute growth curves for the females and males of *C. macrocephalus* were significantly different in Wilson Inlet in both 2005-08 and 1987-89 (each $p < 0.05$). However, those differences were not considered biologically significant and the length-at-age data for the two sexes in each period were subsequently pooled. Although the Schnute and von Bertalanffy growth curves were significantly different in each of 2005-08 and 1987-89 ($p < 0.001$), the differences were not considered biologically significant. The von Bertalanffy and Schnute growth curves for *C. macrocephalus* in Wilson Inlet in 2005-08 were both significantly different (both $p < 0.001$) from the corresponding curves derived for this species in 1987-89 (Fig. 5.9, Table 5.2). However, since the differences in the estimated lengths at age never exceeded 5%

at any integer age in the range of dominant ages, the differences in growth between the two periods were not considered biologically significant.

Table 5.6. *F*-statistics, mean squares (given in parenthesis) and significance levels for three-way ANOVAs of the catch rates of *Cnidoglanis macrocephalus* in composite gill nets in the regions open and closed to fishing in Wilson Inlet in each season of each year nested within the periods between winter 1987 - autumn 1989 and winter 2005 - autumn 2008. * $p < 0.05$.

	Main effects			
	Period (P)	Season (S)	Region (R)	Residual
df	1	3	1	79
Catch rate	15.509*	1.815	9.746	
	(5.523)	(0.497)	(6.155)	(0.156)
	Interactions			
	P x Y	P x S	P x R	S x R
df	3	3	1	3
Catch rate	0.528	0.701	0.131	0.315
	(0.357)	(0.192)	(0.082)	(0.073)
	P x S x Y	P x R x Y	P x S x R	P x S x R x Y
df	9	3	3	9
Catch rate	1.187	2.741	0.946	1.480
	(0.274)	(0.633)	(0.218)	(0.231)

The 350-399 mm length class of *C. macrocephalus* collected from Wilson Inlet in 1987-89 contained only three mature females. The percentage contributions of mature females to all females in the 400-449 and the 450-499 mm length classes were less in that earlier period than in 2005-07 (Fig. 5.4), which helps account for the L_{50} of females at maturity being far greater ($p < 0.001$) in the earlier period, *i.e.* 449 mm, than in the later period, *i.e.* 417 mm (Table 5.3). A few females became mature at the end of their second year of life in 1987-89, but not 2005-07 (Fig. 5.4). The percentage of females that were mature at the end of their third year of life was, however, far greater in 2005-07 (62%) than in 1987-89 (15%) and the same was true for females at the end of their fourth year of life, *i.e.* 100 *vs* 73% (Fig. 5.4). This accounts for the A_{50} for female *C. macrocephalus* at maturity in Wilson Inlet in 1987-89 (3.5 years) being significantly greater ($p < 0.001$) than in 2005-07 (3.1 years).

Fishing mortality (*F*) in Wilson Inlet in 2005-08, which was estimated taking into account interannual recruitment variability, was 1.47 year^{-1} and thus far greater than the 0.57 year^{-1} estimated for the population in this estuary in 1987-89 (Fig. 5.6, Table 5.4).

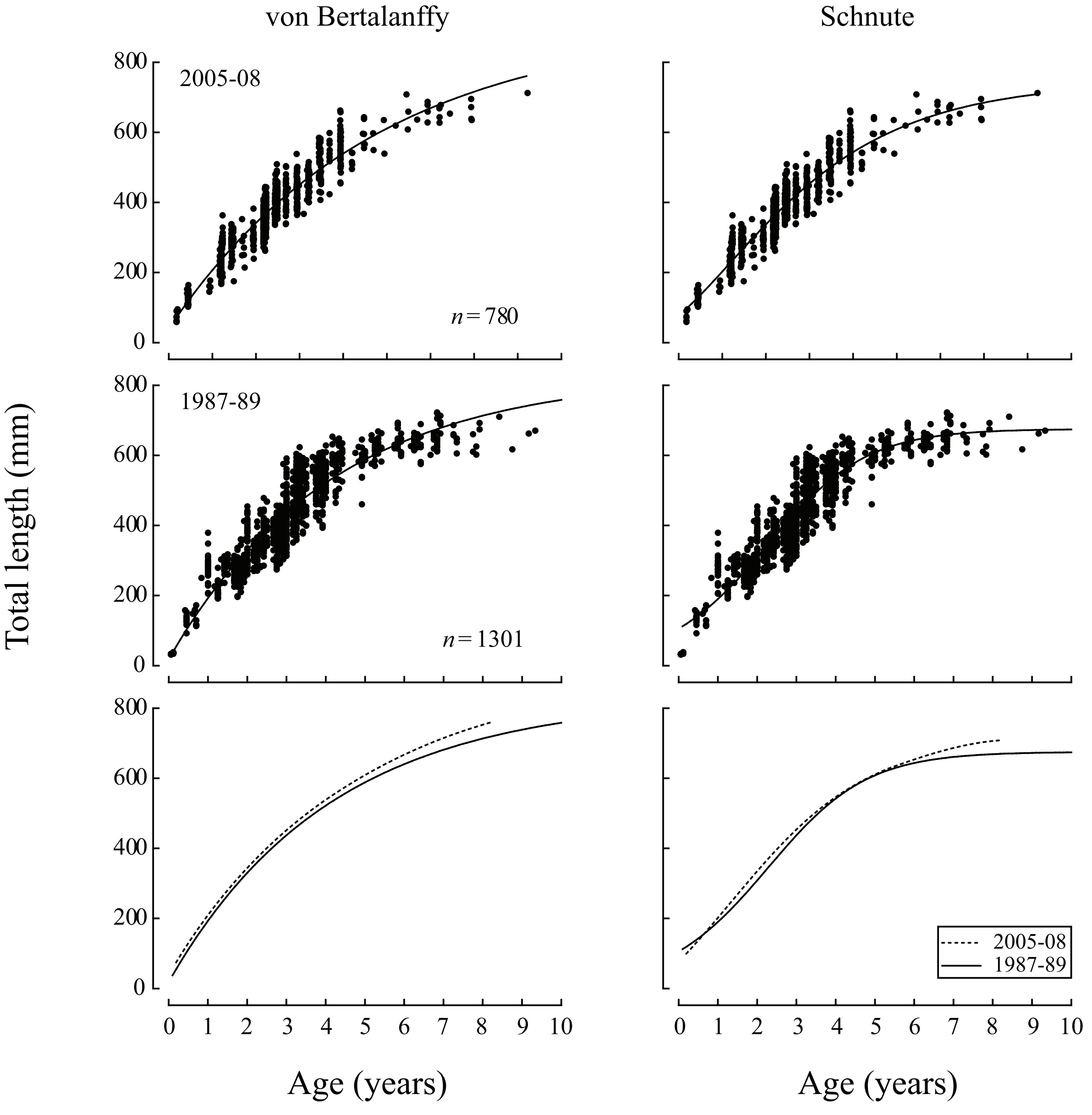

Fig. 5.9. Comparisons between von Bertalanffy growth curves for *Cnidoglanis macrocephalus* in Wilson Inlet in 2005-08 and 1987-89.

5.4. Discussion

5.4.1. Comparisons of the abundances and growth in the various estuaries

Seasonal sampling of several sites in six estuaries on the south coast of Western Australia demonstrated that the relative abundance of *Cnidoglanis macrocephalus* varied markedly among these estuaries. Thus, gill net catch rates were higher in Irwin and Wilson inlets and Oyster Harbour than in Wellstead Estuary, negligible in Stokes Inlet and zero in Broke Inlet. Substantial catches of *C. macrocephalus* were likewise obtained from Wilson Inlet during a study in 1987-89 (Potter *et al*., 1993; Laurenson *et* al., 1994) and were also low in Wellstead Estuary during a comparable study in 1996-98 (Young and Potter, 2002). The high catch rates in Irwin and Wilson inlets are consistent with the numbers of *C. macrocephalus* in these estuaries currently being able to support commercial fisheries for this species (and particularly Wilson Inlet, from which the vast majority of Western Australia's catch of *C. macrocephalus* is taken), as they have done for many years (Smith and Brown, 2008).

As *C. macrocephalus* was relatively abundant in the Irwin and Wilson inlets during the present study and was not recorded in Broke Inlet, which is also seasonally open to the ocean (Chuwen *et al.*, 2009a), the variation in abundance among estuaries is not directly related to estuary type. The success of *C. macrocephalus* in the coastal surf zone has been related to the presence of substantial amounts of macrophytes, which provide an important source of protection and invertebrate prey for the juveniles of this species (Lenanton *et al*., 1982; Robertson and Lenanton, 1984; Lenanton and Caputi, 1989). Furthermore, field and laboratory studies showed that the density and biomass of the juveniles of *C. macrocephalus* were positively related to the amount of detached macrophytes and that these juveniles do not occur over bare sand (Crawley *et al*., 2006). It thus appears highly relevant that Irwin and Wilson inlets, in which *C. macrocephalus* was most abundant, both contain very extensive growths of *Ruppia megacarpa*, which sloughs off an appreciable proportion of its volume each year to produce detached wracks, whereas this seagrass is very sparse in the oligotrophic Broke Inlet (Brearley, 2005). While Oyster Harbour contains appreciable growths of seagrass, the most abundant of its species belong to *Posidonia*, which does not produce wracks to the same extent as *R. megacarpa*. Thus, this seagrass does not produce such suitable conditions for juvenile *C. macrocephalus*, which would help account for the abundance of this plotosid in this estuary being lower than would have been the case in Irwin and Wilson Inlets if these latter estuaries had not been the focus of high levels of exploitation.

The fact that Wellstead Estuary contains very extensive growths of *R. megacarpa* raises the question as to why this normally-closed estuary does not also contain the type of substantial population of *C. macrocephalus* that is found in Irwin and Wilson inlets, in which profuse growths of this seagrass are also found. However, unlike the Irwin and Wilson inlets, the Wellstead Estuary becomes hypersaline, with salinities at some sites occasionally exceeding 70 (Brearley, 2005) and, on rare occasions, even exceeding 120 (Young and Potter, 2002). As laboratory studies showed that individuals of estuarine populations of *C. macrocephalus* survived better in salinities that were less, rather than greater than natural sea water (Kowarsky, 1973), the high salinities sometimes attained in the Wellstead Estuary would account for the catches of this species in this estuary being low. Similarly, the capture of only two *C. macrocephalus* in Stokes Inlet probably reflects the fact that this estuary also becomes hypersaline (Chuwen *et al.*, 2009a).

Our data demonstrated that the growth curves for *C. macrocephalus* in Irwin and Wilson inlets increasingly diverged after fish had reached *ca* 1.5 years. Thus, on the basis of growth equations, fish had attained, by 4 years of age, lengths that were 11% greater in Irwin Inlet than in Wilson Inlet. While such differences may partly reflect the genetic differences that are known to exist between the populations of this plotosid in the various estuaries of south-western Australia (Ayvazian *et al.*, 1994), they may also be related to differences in the amount of food available. Although there are no data on the relative amounts of detached macrophytes in either Irwin or Wilson inlets, the concentration of phosphorous in the water column of the former estuary is greater (Brearley, 2005), which would help account for our observations that the density of macrophytes is greater in this system and that, therefore, the density of the prey of *C. macrocephalus* would presumably also be greater. Despite the absence of young juveniles in Oyster Harbour, the length-at-age data demonstrate that larger and older fish do not grow as rapidly in this permanently-open system as in Wilson Inlet and even more particularly Irwin Inlet. It is thus relevant that, as pointed out earlier, the dominant seagrass in Oyster Harbour, *i.e. Posidonia* spp., does not produce wracks to the same extent as *R. megacarpa* and would therefore not contain the same rich source of prey.

5.4.2. Reproductive biology, mortality and management implications

This study indicates that the attainment of maturity by *C. macrocephalus* in estuaries along the south coast of Western Australia is related to both body size and age. Thus, the estimated L_{50} of females at maturity was *ca* 470 mm in Irwin Inlet and *ca* 417 mm in Wilson Inlet and 72% of females matured at the end of their third year of life in Irwin Inlet, compared with 62% in Wilson Inlet, in which growth was less. In an earlier study, Nel *et al*. (1985) also produced evidence that the attainment of sexual maturity of *C. macrocephalus* in the Swan River Estuary on the lower west coast of Western Australia was related to body size and age and that only the larger individuals of the younger age classes were mature during the spawning period.

From a management perspective, it is highly relevant that the MLL for retention of *C. macrocephalus* is currently 430 mm in Western Australia and that the L_{50}s for the females of this species are currently *ca* 40 mm greater than this value in Irwin Inlet but 13 mm less in Wilson Inlet, with both L_{50}s having tight confidence intervals. These comparisons imply that, while a substantial number of females would have had the opportunity to spawn once before becoming vulnerable to capture and retention by fishers in Wilson Inlet, few would have such an opportunity in Irwin Inlet. This raises the strong possibility that, in estuaries such as Irwin Inlet, the current MLL for retention of this species is not sufficiently high to protect this species from over-exploitation if the stock in that system becomes the target of heavy fishing pressure.

Although the estimated total mortality of *C. macrocephalus* in Wilson Inlet exceeded that for the stock in Irwin Inlet, the closed area in Wilson Inlet provides some protection from fishing mortality in that estuary. This is reflected by the fact that, although fish with ages from 3+ to 6+ were caught in the closed area, virtually no such fish were caught in the area open to fishing.

Total mortality Z for the (overall) Wilson Inlet population almost doubled between 1987-89 and 2005-08, while fishing mortality F increased from *ca* 0.57 to 1.47 year^{-1}, and thus represented a 255% increase. While fishing mortality in the earlier period was already *ca* 1.6 times that of natural mortality M, it had become 4.2 times that of M by 2005-08. The current level of F for *C. macrocephalus* in this estuary is thus well in excess of 0.75 or 0.8M, values which have been considered to be target reference points for *F* in data-poor fisheries (Gabriel and Mace, 1999). The current levels of exploitation are thus unlikely to be sustainable.

5.4.3. Inter-period comparisons for Wilson Inlet

As reported for 1987-89 (Laurenson *et al.*, 1993b), the catch rates of *C. macrocephalus* in waters closed to commercial fishing in Wilson Inlet in 2005-08 continued to exceed those for the open fishing waters. Fish collected from the closed area were both larger and older than those caught in the open area. The differences between the catch rates were more marked in the earlier period, however, reflecting the greater proportion of older fish in that earlier period. The closed area is continuing to serve as a refugium for the population, but current high levels of exploitation appear to have reduced the abundance of fish that survive to older ages and larger sizes.

The fishery in Wilson Inlet is multi-sector (commercial and, to a smaller extent, recreational), multi-gear and multi-species, of which *C. macrocephalus* is just one of the species targeted. Thus, the data reported on the commercial fishery to the Department of Fisheries Western Australia are greatly influenced by the variable extent to which the different species are targeted and it is not possible to identify the fishing effort that is directed towards a single species. Such fishery-dependent data thus provide little information on the status of the *C. macrocephalus* stock in Wilson Inlet and how it might be changing over time. At present, such information can only be obtained from fishery-independent data.

The fact that the mean catch rates of *C. macrocephalus* in Wilson Inlet, derived from fishery-independent studies, declined significantly and by a factor of nearly three between 1987-89 and 2005-08, very strongly indicates that the abundance of this species in Wilson Inlet has declined markedly between the two periods. It is also particularly relevant that the samples in 1987-89 were dominated by 2+ and even more particularly 3+ fish and contained appreciable numbers of fish older than 3 years and a few extending up to 12 years old, whereas those in 2005-08 were dominated by the 2+ age class and contained few fish older than 3 years and none older than 6 years. The fact that the age compositions were consistent among the years in each period in this estuary implies that the differences in age structures between periods represent a real change over time rather than just snapshots of two different periods. The lower number of older fish in the later period is reflected in the lower percentage of fish > the MLL of 430 mm. The implication that the age and length compositions have changed during the last two decades is consistent with the fact that, during this period, commercial fishers have reduced the gill net mesh size used to target this species in response to reduced catches of larger fish (O. McIntosh, pers. comm.). Furthermore, a decline in the relative abundance of larger and older fish accounts for estimates of total mortality increasing from 0.92 to 1.82 year^{-1}. The view that commercial fishing has led to heavy exploitation of this species during the last two decades is supported by the observation that, although fishery-independent catch rates have declined, the commercial catches of this species in Wilson Inlet exhibited no clear tendency to decline between these two periods (Smith and Brown, 2008) and thus have presumably been maintained by an increase in effort or fishing power.

Comparisons with reproductive data collected by Laurenson *et al.* (1993b) for *C. macrocephalus* in Wilson Inlet during 1987-89 demonstrate that this species matured at a far larger size and greater age in that period than in 2005-08, with respective L_{50}s of 449 and 417 mm and respective A_{50}s of 3.1 and 3.5 years. This shift towards the attainment of sexual maturity at smaller sizes and younger ages by the heavily-fished *C. macrocephalus*, which may reflect fisheries-induced evolution of these life-cycles traits, parallels the trends recorded for some other heavily exploited fish populations (*e.g.* Trippel, 1995; Olsen *et al.*, 2004).

Our results, which indicate that the growth of *C. macrocephalus* is not related to fishing mortality, support the findings of Hilborn and Minte-Vera (2008) who, on the basis of meta-analysis of 73 commercially-fished stocks, found no evidence for the growth of fish in heavily fished populations of a species to be related to fishing intensity.

6. Benefits and adoption

As identified in the original application, the Western Australian Fishing Industry Council, Department of Fisheries Western Australia, the Department of Water (previously Department of Environment), RecFishWest, South Coast Natural Resource Management Inc., South Coast Licensed Fisherman's Association and local commercial and recreational fishers will benefit from the outcomes of this project through it having achieved the following.

1. Improved environmental management will assist in maintaining the habitats occupied by Black Bream and Estuary Cobbler in south coast estuaries, and thereby assist in sustaining these stocks.
    a. Elucidation of the factors influencing the environmental characteristics of divergent south coast estuaries enables the development of appropriate environmental management plans for individual estuaries, such those recently completed by the Department of Water for the Culham and Stokes inlets.
    b. Expanded understanding of the factors affecting the spawning and/or recruitment success of Black Bream and Estuary Cobbler will further contribute to improved management of the environments of south coast estuaries.
2. The data on the biology and current status of the stocks of Black Bream and Estuary Cobbler produced by the study will facilitate the implementation of appropriate management plans, thereby assisting in sustaining the stocks of these two species in south coast estuaries.
    a. Determination of the current age and size compositions, growth and total mortality of Black Bream and Estuary Cobbler has provided useful information for recreational and commercial fishers and the implications of these findings will be discussed with Department of Fisheries' managers.
    b. Comparisons of the above biological attributes with those derived for Black Bream in Wellstead Estuary 10 years ago and Estuary Cobbler in Wilson Inlet *ca* 20 years ago have demonstrated marked shifts in catch and growth rates between periods.

3. Improved understanding and awareness by fishers and local communities will allow more informed discussion and understanding of management changes.
    a. Recreational and commercial fishers and the broader community have benefited directly through our dissemination of information during numerous public seminars that provided background and regular updates on the results and their implications.
4. Dissemination of our results at three international conferences has promoted the unique nature of south coast Western Australian estuaries to an international scientific audience.

7. Further development

The final report has been provided to managers in Western Australia at the Department of Fisheries, Department of Water, South Coast Natural Resource Management Inc. and Department of Environment and Conservation. Results of the study have been discussed with scientists from the Department of Fisheries, to facilitate communication of the implications of the project to fishery managers.

The final report has also been provided to RecFishWest and the South Coast Licensed Fisherman's Association. Articles will be produced for ProWest and for local newspapers on the south coast of Western Australia. A number of presentations have been given on the south coast to interested recreational and commercial fishers and the broader community to describe the outcomes of this research. Three papers are currently in preparation and will be submitted to relevant peer-reviewed international journals.

The results of this study highlight that key biological attributes of selected recreational and/or commercial fish species vary markedly among a number of environmentally divergent estuaries on the south coast of Western Australia. Although the collection of sound, quantitative data for Black Bream and Estuary Cobbler now includes an example of each type of estuary found on the south coast, further work is required to disentangle the relative effects of environmental change and fishing pressure on fish stocks in south coast estuaries.

8. Planned outcomes

The project has produced the following outputs that will facilitate the outcomes listed below:

1. Sound fisheries-independent data on the biology and contemporary status of the Black Bream and Estuary Cobbler in estuaries on the south coast of Western Australia.
2. The relationships between the status of stocks of Black Bream and Estuary Cobbler and environmental variables have been identified.
3. Strong evidence that the stock of Estuary Cobbler, in the main estuary (Wilson Inlet) in which it is commercially fished, has declined during the last 20 years.
4. The potential benefit of closing areas within estuaries to fishing for Estuary Cobbler has been confirmed.
5. Data to indicate that hypersalinity can have highly detrimental effects on the stocks of Black Bream in an estuary.
6. Information on how the age and size compositions, growth and total mortality of Black Bream and Estuary Cobbler in two estuaries on the south coast of Western Australia have changed over 10 and 20 years, respectively.

The following outcomes were or will be facilitated by above outputs:

1. Future management plans to facilitate the sustainability of the commercial and recreational fisheries for Black Bream and Estuary Cobbler and for the maintenance of the environments of estuaries on the south coast will be based on sound fisheries-independent data on the biology and contemporary status of those two species and knowledge of their relationships with the environment.
2. Understanding by managers, scientists and fishers of the implications of the changes in the age and size compositions, growth and total mortality of Black Bream and Estuary Cobbler in two south coast estuaries for the sustainability of the fisheries for those two important species in those estuaries.
3. Awareness by managers of the current status of stocks of Black Bream and Estuary Cobbler in south coast estuaries and, in particular, the decline in abundance of the latter species in Wilson Inlet over the last 20 years.

4. The implications of hypersalinity for the stocks of Black Bream within estuaries are now well understood by fishery managers and local communities.
5. Confidence by fishery managers that closure of certain areas within estuaries is an effective tool to protect the stocks of Estuary Cobbler in those estuaries.
6. Through their involvement in the study, two honours and a PhD student have been trained in contemporary techniques in fisheries science and population and community ecology.
7. A wide understanding by fishers and members of local communities of the significance and benefits of the study was created through their strong engagement with the research team during the study.

9. General conclusions

Sound quantitative data have been obtained for the biological attributes of *Acanthopagrus butcheri* and *Cnidoglanis macrocephalus* and also for the environmental conditions in permanently-open, seasonally-open and normally-closed estuaries on the south coast of Western Australia. These data have been used to meet all of the objectives in our original application. The major findings and conclusions are as follows.

- Environmental characteristics differ markedly among estuaries and estuary types reflecting variations in the relationship between the volume of fluvial discharge, which is determined by a combination of the amount of local rainfall, catchment size and extent of clearing of native vegetation, and the amount of intrusion by marine waters, which is largely controlled by the size and duration of the opening of the estuary mouth.
- In permanently and seasonally-open estuaries, pronounced haloclines and oxyclines were present in their narrow riverine regions, but not in their wide basins where the waters are subjected to wind-driven mixing.
- The maximum ages of *A. butcheri* ranged downwards from 13 to 15 years in Wilson Inlet, Wellstead Estuary, Culham Inlet and Stokes Inlet, 9 years in Oyster Harbour and only 5 years in Irwin Inlet and Hamersley Inlet.
- Growth of *A. butcheri* varied markedly among the various estuaries, probably reflecting differences in the density of *A. butcheri*, quality/quantity of food and/or salinity regime.
- The relationship between fish length and otolith radius varied between sexes and among estuaries. The width of the annual growth zones of otoliths was shown, however, to vary among years, particularly in Stokes Inlet, in which the growth zones were widest in years of relatively high rainfall and thus when salinities were presumably below that of full-strength sea water.
- The lengths at maturity (L_{50}s) of *A. butcheri* in the four estuaries from which it was possible to obtain substantial reproductive data were not significantly different (all $p > 0.05$), with the values for females, for example, ranging only from 146-161 mm.

- Recruitment of *A. butcheri* varied markedly among years and estuaries. Recruitment in the seasonally-open Wilson Inlet was greatest in years of below average rainfall and thus presumably also relatively elevated salinities and reduced stratification and associated deoxygenation of the bottom water layer in the rivers.
- Although massive mortalities of *A. butcheri* in two of the normally-closed estuaries prevented comparisons across this estuary type, it appears that strong recruitment in these estuaries is related to years of relatively high rainfall and presumably the lowering of salinities in these estuaries to below that of full-strength sea water.
- Total mortality (Z) of *A. butcheri* appeared to be slightly higher in estuaries with the greatest fishing pressure.
- *Cnidoglanis macrocephalus* was most abundant in Irwin and Wilson inlets followed by Oyster Harbour and Wellstead Estuary and only two individuals were caught in Stokes Inlet and none was taken in Broke, Hamersley or Culham inlets.
- Catch rates in each estuary in which *C. macrocephalus* was relatively abundant decreased from their maxima in summer to their minima in winter.
- Although the growth of *C. macrocephalus* was similar between the sexes in each estuary in which it was relatively abundant, growth did differ significantly among the estuaries, with the predicted lengths at ages after *ca* 2 years of age being greatest in Irwin Inlet and least in Oyster Harbour, reflecting differences in environmental characteristics such as macrophyte densities and species composition of macrophytes.
- The L_{50} at first maturity determined for females of *C. macrocephalus* was significantly different in Irwin and Wilson inlets, with the length at first maturity being greatest in the estuary in which the lengths at age were greatest, *i.e.* Irwin Inlet (470 mm) vs Wilson Inlet (417 mm).
- Total mortality of *C. macrocephalus* was very high in Wilson Inlet (1.82 year^{-1}) and relatively lower in Irwin Inlet (1.12 year^{-1}).
- In Wilson Inlet, catch rates and mean lengths of *C. macrocephalus* were generally greatest in the area closed to commercial fishing at the seaward end of the estuary, indicating that this area continues to act as a refugia for this species.

- Comparisons with data for *C. macrocephalus* collected at the same sites in Wilson Inlet in 1987-89 demonstrated that catches rates between those two periods have declined by almost 300% and the age structure of the population has shifted from predominately 2+ and 3+ year old individuals in 1987-89 to being dominated by 2+ individuals in 2005-08, with very few fish older than 3+ years of age in the later period.
- Growth of *C. macrocephalus* differed significantly between 1987-89 and 2005-08, with predicted lengths at ages > *ca* 4 years being greater in the earlier period.
- Analyses of reproductive data for *C. macrocephalus* demonstrated that the L_{50} at first maturity has decreased from 449 mm in 1987-89 to 417 mm in 2005-08 in Wilson Inlet and that the age at maturity has also decreased, with only 20% of females being mature at the end of their third year of life in 1987-89 compared with 62% during 2005-08.
- Estimates of total mortality for *C. macrocephalus* in Wilson Inlet increased from 0.92 year^{-1} in 1987-89 to 1.82 year^{-1} in 2005-08.

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11. Appendices

11.1. Appendix 1: Intellectual property

The value of the intellectual property will be 1% FRDC and 99% Murdoch University based on the project agreement.

11.2. Appendix 2: Staff

Prof. Ian Potter
Dr Ben Chuwen
Dr Steeg Hoeksema
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Miss Jo Bannister
Mr Xavier Launay