**FRDC FINAL REPORT**

# ASSESSMENT OF INSHORE HABITATS AROUND TASMANIA FOR LIFE-HISTORY STAGES OF COMMERCIAL FINFISH SPECIES

*A.R. Jordan, D.M. Mills, G. Ewing and J.M. Lyle*

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# 94/037 Assessment of inshore habitats around Tasmania for life-history stages of commercial finfish species.

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## OBJECTIVES:

1. To determine the abundance and distribution of commercial fish species associated with selected inshore soft-bottom habitats around Tasmania.

2. To categorise the habitat types in these areas and determine the size/age structure of commercial fish species by habitat as a means of assessing the critical habitat requirements of such species.

3. To determine the fish community structure of inshore habitats and examine the associations between habitats and fish assemblages.

## NON TECHNICAL SUMMARY

In Tasmania, there is a paucity of information on the life-history, population parameters and habitat requirements of fish associated with inshore soft-sediment habitats, particularly seagrasses. Clearly, such information is needed before stock assessment models can be developed, recruitment processes understood, key habitats identified and appropriate management measures developed to minimise impacts on these habitats. In order to examine the structure of fish communities in coastal soft-sediment habitats around Tasmania, the demersal and larger mobile fish fauna were routinely sampled from three areas-Norfolk Bay, Georges Bay and Prosser Bay. In each area, representative unvegetated (mud and sand) and seagrass habitats between 1 and 12 m deep were sampled seasonally.

The abundance of demersal fish associated with subtidal *Heterozostera* seagrass beds were highest in Norfolk Bay and lowest in Prosser Bay, although seasonal variations in abundance differed across areas. In contrast, species richness was similar between areas, although it was consistently lowest in Prosser Bay. Different fish communities were

present in each area, although differences were also found between sites within Georges Bay, which is possibly related to the larger gradient of physical characteristics in this bay.

The temporal and spatial patterns of seagrass communities were examined in more detail in Norfolk Bay from bi-monthly sampling over two years. Two of the three most abundant species, and six of thirty two species overall showed a distinct preference for beds with the highest seagrass density. This appears to reflect that fact that all dominant species spawn within the bay, with behavioural selection taking place for dense beds. Abundance peaked in winter in both years, and was lowest in summer, which reflects the lack of seasonal transient species in the beds and winter die-back of beds reducing overall area of available habitat. While the community structure differed between sites, this pattern was not consistent between years. In general, seagrass beds in all three areas throughout Tasmania were found to be an important habitat for small, resident fishes, but not an important nursery habitat for economically important scalefish species.

The fish fauna associated with *Heterozostera* seagrass and unvegetated habitats were compared in all three areas. Seagrass sites had a significantly higher abundance of demersal fish and a distinct community compared to unvegetated sites in Norfolk Bay and Georges Bay. In contrast, neither abundance or community composition differed between habitats in Prosser Bay. This pattern may be attributed to the patchy distribution of seagrass beds that result from the higher degree of exposure of the bay and the significant loss of beds over the past 20-30 years. Demersal fish in seagrass beds were dominated by small resident species, while those in unvegetated habitats were dominated by juveniles of larger species. Few larger more mobile species showed a distinct habitat preference. Unvegetated habitats were found to be more important than seagrass as a nursery area for juveniles of commercially important finfish species.

Fish communities in distinct *Posidonia* and *Heterozostera* seagrass beds were examined from seasonal sampling in the Tamar River. While neither abundance or species richness differed between habitats, each species of seagrass had a distinct fish community, with a large number of species unique to each habitat. Most abundant fish species were small permanent residents caught seasonally as juveniles and throughout the year as adults. It is clear that while both species of seagrass may be present in a single estuary each species should be managed as individual habitats.

The main factors structuring inshore soft-sediment fish communities in Tasmania is the presence/absence of vegetation, and in vegetated areas, the type of seagrass present. Secondary to this are broad scale factors relating to differences between bays. Of lesser importance was variability within bays, including position of beds and differences in seagrass density. As fish communities in seagrass beds are more similar within an bay than between bays, management of seagrass beds should be at the scale of individual coastal bays, as each area can be described as having some 'unique' community. In general,

seagrass beds were found to be the least important soft-sediment habitat in coastal waters of Tasmania utilised by juveniles of commercially important species. In contrast, shallow unvegetated habitats were found to be an important nursery habitat for yellow-eye mullet, Eastern Australian salmon and greenback flounder, indicating that management should be directed as minimising impacts on both habitats throughout the coastal zone. An effective means of achieving this outcome is through the development of habitat management guidelines that aim to provide a basis in which sustainable management decisions can be made for the coastal zone. Such guidelines will also assist the integration of habitat considerations and conservation into fishery management plans.

Limited mapping of seagrass habitats along the north coast identified around 530 $km^2$ of previously undocumented seagrass beds, approximately doubling the known area around the State. This has highlighted the lack of information available on the distribution of such habitats around Tasmania and reflects the lack of habitat mapping at an appropriate spatial scale for effective management and monitoring.

The life-history ecology of sand flathead was examined in detail to determine the spatial and temporal patterns of spawning distribution, recruitment, abundance and distribution and size and age composition. Spawning occurred for up to six months between October and March in estuaries and coastal embayments, with settlement occurring over an extended period in summer. Size at maturity for males and females was 21 and 23 cm, respectively. While juveniles showed a preference for unvegetated habitats, mature sand flathead showed no preference between seagrass and unvegetated habitats. The low abundance of juveniles in the shallow nearshore beach habitats suggests that the unvegetated subtidal zone is a more significant nursery area for the species than the intertidal zone. Otolith annuli were validated by examining trends in marginal increments, with maximum ages of 17 years for males and 13 for females. The population was dominated by 4 to 7 years olds with evidence of recruitment variability.

The spawning, early life-history, size composition and age and growth of sea garfish was examined from research and commercial sampling throughout eastern Tasmania. Spawning in eastern Tasmania occurred between October and February and was concentrated over shallow unvegetated habitats. Eggs were demersal, attached to drift algae with filamentous hairs and hatched at around 28-30 days old. Egg and early larval developmental stages are described. Size compositions from the commercial fishery differed considerably between north and east coast regions. Maximum ages of 7 years for males and 8 years for females were estimated from the east coast. Further work is needed on spawning habitat requirements and age and growth from north coast regions.

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# 1. Background

Throughout coastal waters of Tasmania there are a large range of soft-sediment habitats that can be broadly classified into open, semi-closed and closed estuaries, and beach environments. While such inshore areas are extensive around Tasmania and are represented by seagrass, sand and mud habitats, very little work has been done to assess the significance of these habitats for life-history stages of economically important fish species. A considerable amount of commercial fishing activity occurs in coastal soft-sediment habitats around Tasmania, with total landings of scalefish species associated with such habitats at around 690 tonnes in 1996/97, representing 60% of all scalefish catches (Lyle 1998). A wide range of species are targeted, including tiger and sand flathead, jackass morwong, Australian salmon, school whiting, spotted and blue warehou, southern sea garfish, mullet and flounder. These species also form a significant component of the recreational scalefish catches throughout the state (Lyle and Smith 1998). In addition, a number of these species are caught in adjacent Commonwealth fisheries or are taken in other state fisheries.

Despite their commercial and recreational significance little information has been available to researchers or managers on the life-history, ecology and population parameters of key inshore demersal species in Tasmania. Basic information such as size at sexual maturity, spawning distribution, recruitment processes and age and growth is lacking for most species. Such information is necessary to understand the dynamics of populations and define many life-history parameters that are an essential part of developing age-based stock assessments. It is also an important part of establishing the age composition of a population in order to assess the extent of year-class variability, leading to a better understand the factors that cause such variability.

In addition, there has been little understanding of the relationship or dependence of life-history stages of scalefish species to inshore soft-sediment habitats in Tasmania. The only substantive research reported is by Last (1983) who found the inshore zone (typically <10m) to be utilised by adults and/or juveniles of many commercial species including jackass morwong, blue warehou, sand flathead, school whiting, southern sea garfish, Australian salmon and yellow-eye mullet. An understanding of the level of dependence these species have with inner continental shelf waters has recently been advanced with studies examining the significance of this region for life-history stages (Lyle and Ford 1993, Jordan 1997). Juveniles of several species including jackass morwong, tiger flathead, blue warehou, latchet and school whiting were restricted to inner-shelf indicating the importance of this part of the shelf as a nursery area for these species.

However, the main limitation of this previous work was the absence of quantitative sampling at depths of <15 m, an area which is both extensive, and represented by a range of soft-sediment habitat types, including seagrasses. Before the relative value of such habitats within coastal waters could be fully evaluated it was seen as critical to sample the

inshore zone to examine the use of this area by life-history stages of scalefish species. In addition, this is particularly important to the development of methods to determine indices of recruitment strength which relies on an understanding the full extent of nursery areas.

Throughout Australia, seagrasses are a dominant feature of inshore environments and are utilised by many fish species of commercial and recreational importance. Seagrasses are important contributors to coastal productivity, providing much of the basis for inshore fish production. While their significance as nursery areas for juvenile and sub-adult stages of economically important fish has been documented for mainland waters of southern Australia (see Bell and Pollard 1989), very little work has been conducted in Tasmania.

Recent work on the extent of seagrass habitats throughout Tasmania suggests that up to 500 $km^2$ of seagrass occurs around the state, with *Posidonia australis* and *Amphibolis antarctica* dominant along the north coast, and *Heterozostera tasmanica* common in estuaries and coastal embayments around the state (Rees 1993). Such seagrass beds were found to contain the highest fish diversity compared with unvegetated habitats, and were important nursery areas for several species including sand flathead, sea garfish and yellow-eye mullet (Last 1983). Despite their present distribution, a significant loss of seagrass beds appears to have occurred, particularly in areas close to centres of population and human activity (Rees 1993). The loss of these habitats results in not only a decrease in coastal productivity but can lead to a reduction in the settlement success of juveniles and a decrease in biodiversity.

While seagrass beds are widely recognised as an important nursery area for many economically important fishes by providing protection and increased food resources compared to bare substrata, unvegetated habitats are becoming increasing recognised as an important habitat for juvenile fishes (Ayvazian and Hyndes 1995, Edgar and Shaw 1995a, Gray *et al.* 1996, Jenkins *et al.* 1997). They are particularly important fish habitats when located adjacent to seagrass beds (Ferrell and Bell 1991), which has important implications in the management of such areas. While there have been many studies assessing the habitat requirements of juvenile fishes in mainland waters of southern Australia, the lack of studies in Tasmania have precluded an assessment of the significance of both vegetated and unvegetated habitats. In addition, few studies have examined the early life-history of the majority of key commercial species resulting in a poor understanding of habitat associations of different ontogenetic stages.

## 2. Need

Knowledge of the life-history, population parameters and habitat requirements of our fish resources is critical for well informed and scientifically based management decisions. While some research has been undertaken in southern Australia to evaluate the significance of inshore waters to life-history stages of commercial fish species, in Tasmania there is a paucity of information on the relationship between fish production and inshore habitats. In

addition, basic data such as the abundance, distribution and size/age structure of commercial species in inshore areas is either lacking or only preliminary for most species. Clearly, such information is needed before the significance of such areas as fish habitats can be evaluated, recruitment processes understood, key habitats identified and appropriate management measures developed to minimise impacts on these habitats.

There are a number of threats to inshore soft-sediment habitats around Tasmania which have the potential to adversely impact on the fish populations that are dependant upon them. In the nearshore, changes in seagrass beds are probably the most conspicuous indication of habitat change, and therefore attract a lot of attention. The significant decline of seagrass beds that has occurred around the state, with up to total loss in some areas, is an obvious example (Rees 1993). Increased nutrient levels and turbidity (from urban and industrial discharges, catchment usage) appear to play a prominent role in the decline of seagrass beds. These human induced changes also impact on other soft-bottom habitats through algal and dinoflagellate blooms and accumulation of wood pulp effluent. In addition, indirect effects such as the introduction of exotic species, particularly the northern Pacific seastar, *Asterias amurensis*, have the potential to significantly alter the structure of invertebrate communities (Davenport and McLoughlin 1993). This directly impacts on the productivity and biodiversity of such communities and ultimately the fish populations associated with these habitats. The habitat requirements of scalefish species needs to be identified before plans can be developed to minimise the direct and cumulative impacts to the key habitats.

A further outcome of this work will be to provide valuable information on recruitment processes for key commercial species caught in both Tasmanian fisheries and adjacent Commonwealth fisheries. The identification of nursery grounds could also indicate the feasibility of developing recruitment indices for such species that could provide early warning signal of year-class strength variability and possible recruit overfishing.

## 3. Objectives

1. To determine the abundance and distribution of commercial fish species associated with selected inshore soft-bottom habitats around Tasmania.

2. To categorise the habitat types in these areas and determine the size/age structure of commercial fish species by habitat as a means of assessing the critical habitat requirements of such species.

3. To determine the fish community structure of inshore habitats and examine the associations between habitats and fish assemblages.

# 4. General methods

## 4.1 Main survey areas

The demersal and larger mobile fish fauna from inshore (<15 m) soft-sediment habitats were routinely sampled in four areas around the coast of Tasmania - Norfolk Bay, Georges Bay Prosser Bay and the Tamar River (Fig. 4.1). Such areas were chosen to represent a range of coastal environments throughout Tasmania in order to examine the significance of several habitat types at a range of spatial scales.

Norfolk Bay is a large marine dominated bay situated on the south-east coast of Tasmania linked by a wide entrance (~3.7 km) to Storm Bay via Frederick Henry Bay (Fig. 4.1). It has a small tidal range (~1.3 m) with little estuarine influence. The bay is characterised by a rocky shore composed of sandstone or dolerite and shallow sand embayments, most containing discrete beds of the seagrass *Heterozostera tasmanica* in the 2 to 7 m depth range. However, a more extensive bed of *H. tasmanica* exists in a broader area of shallow ground (~5 m) in the north-west section of the bay (Rees 1993). Small amounts of the seagrass, *Halophila australis* are also present in some embayments, while small, sparse beds of *Zostera muelleri* exist in the intertidal zone in several sand embayments. The center of the bay is dominated by soft mud sediments and is mainly between 10 and 15 m deep.

Georges Bay is a large coastal lagoon situated on the north-east coast of Tasmania linked by a narrow entrance and extensive barway to the Tasman Sea (Fig. 4.1). It has a small tidal range (~1.3 m) and experiences strong tidal flows in the entrance channel. The bay is characterised by wide sandy embayments separated by rocky headlands composed of sandstone or granite and intertidal mudflats on the northern shore. Seagrass in the bay is predominantly *H. tasmanica* occurring in a wide bed along the southern shore, in narrow patchy bed on the north-western shore, and on intertidal sand banks adjacent to the entrance channel. The centre of the bay is dominated by soft mud sediments and is predominantly between 10 and 20 m deep.

Prosser Bay is a semi-exposed marine embayment situated on the central east coast of Tasmania linked to the Tasman Sea via Mercury Passage (Fig. 4.1). The bay has a small tidal range (~1.3 m), experiences little estuarine influence and is characterised by moderately exposed sandy beaches separated by rocky headlands composed of sandstone or dolerite. Seagrass is predominantly *H. tasmanica* occurring in patchy beds in depths of 1 to 5 m. The mouth of Prosser Bay is sand at around 12 m deep.

The Tamar River is a large estuary linked by a narrow entrance (~2 km) to the Bass Strait (Fig. 4.1). The mouth of the estuary has a large tidal range (~3 m) and experiences strong tidal flows. The lower estuary is characterised by exposed and semi-exposed sandy beaches (at the mouth), intertidal flats and extensive rocky and cobbled shores. The Tamar

River supports bulk shipping facilities and heavy industry and has suffered impacts that include shipping, industrial operations, sewage treatment, storm water run-off and agriculture. The upper estuary offers a variety of soft-sediment habitats including mud, sand and beds of *Heterozostera tasmanica* and *Posidonia australis* (in more exposed areas around the mouth of the estuary).

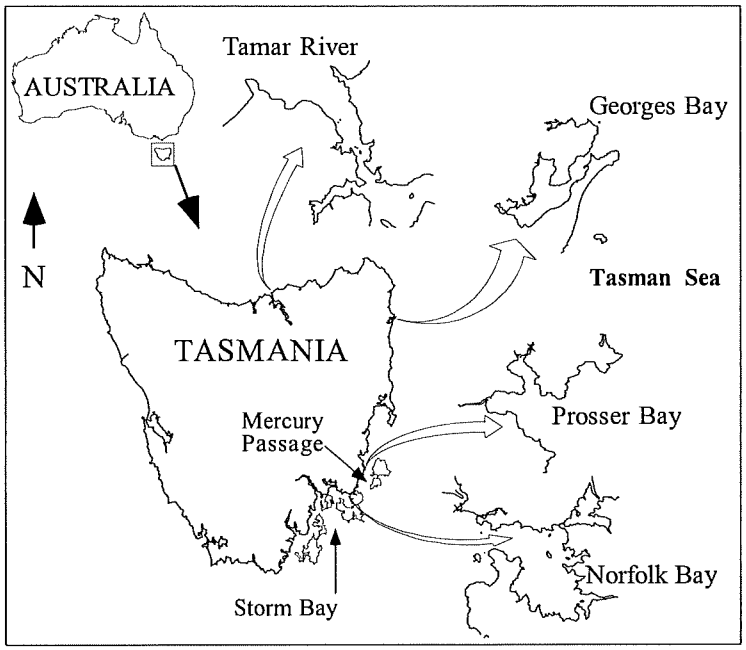



Fig.4.1 Location of the four main inshore survey areas on the north and east coast of Tasmania.

## 4.2 Sampling gear and regime

### 4.2.1 Beam trawl and gillnet surveys

A range of sampling gears have traditionally been used in surveys of inshore fish communities including beam trawls (eg. Young 1981, Bell *et al.* 1992, Warburton and Blaber 1992, Ferrell *et al.* 1993) and seine nets (eg. Ferrell and Bell 1991, Connolly 1994, Edgar and Shaw 1995a, Clarke 1997). Other techniques such as throw (box) traps and poisoning have been utilised in a few studies.

The choice of primary sampling gear for this study was dictated by the depth distribution of soft-sediment habitats around Tasmania, particularly deep (8-12 m) unvegetated areas and seagrass beds. Intertidal seagrass beds are extremely limited in distribution, with the exception of a few sparse, seasonally transient *Zostera muelleri* beds in very sheltered estuaries and embayments (eg. Tamar River). The inner margins of *Heterozostera tasmanica* and *Posidonia australis* beds mostly occur greater than 1 to 2 m below the low water mark, and often in depths considerably greater. For example, the shallow margin of *Posidonia* beds around Waterhouse Island (north-east Tasmania) is approximately 4 m. In many embayments *Heterozostera* beds also start outside an inner margin of sand that is often ~100 m wide. The deep (outer) margin of *Heterozostera* beds in southern Tasmania

is normally in depths of around 8 m, while *Posidonia* beds in northern Tasmania often extend to depths in excess of 20 m.

Deep seagrass beds pose particular sampling problems and so the use of seine nets, throw traps or poisoning was not considered practical. The use of beach hauled seine nets for research purposes is considered to be practical in depths of up to 2 m (Gray and Bell 1986), while boat hauled seine systems have been used successfully in depths of up to five metres (Edgar and Shaw 1995a). As a requirement for the present study was to take comparable samples across all depths at which seagrass occurs in Tasmania, the use of a beam trawl was considered the best option.

The demersal fish fauna was sampled at each site with a beam trawl with an opening of 2.0 x 0.9 m. The trawl consisted of a 2.0 m aluminium beam (with skids and ground chain) with a 5 m long net with the following specifications: headline length 2.6 m, panel mesh 13 mm, codend liner mesh 7 mm. At each site three non-overlapping 3 min trawls were conducted at a tow speed of 2 knots. All sampling was conducted within 2 hours of high tide. Beam trawl catch rates were calculated as the number of fish per tow.

Beam trawl efficiency is likely affected by a wide range of physical and biological variables. Catch efficiency will vary between species depending on several factors including swimming speed, habit and escape behaviour. No attempt was made to calibrate catch efficiency for different species, to do so with an acceptable degree of accuracy across the depth range sampled in this study was considered impractical. Faster moving pelagic species are not captured reliably by beam trawl (Gray and Bell 1986), and consequently results from beam trawl samples reflect relative abundances of demersal species only. Schooling pelagic species are often discounted in community analysis when looking at inshore habitat associations due to patchy distribution, and lack of direct association with the habitats being sampled.

Catchability of many species associated with seagrass increases at night (Gray and Bell 1986). McNeil and Bell (1992) found that significantly more species and individuals of invertebrates were trawled at night. While a greater number of fish were caught at night, the results were not as clear cut as for invertebrates. Some fish species were more abundant during the day, while some were restricted to night or day samples only. Due to the remote nature, difficult access, and navigation hazards associated with most of the sampling sites in our study night trawling was considered unsafe and impractical.

Larger and more mobile fishes were sampled with 30 m long multi-panel gillnets comprising three randomly placed 10 m panels of different gillmesh size (64, 89 and 108 mm). Two multi-panel gillnets were set overnight at each gillnet site with nets set as close to dusk and retrieved as close to dawn as practical. Details of net specifications are given in Table 4.1. The gillnets were buoyed at both ends and anchored at one end with a 1.5 kilogram lead weight. Gillnet catch rates were calculated as the number of fish per hour.

Table 4.1 Gillnet specifications for three mesh sizes used in the study

| Stretched mesh size (mm) | 64 | 89 | 108 |
|---|---|---|---|
| Mesh drop (no.) | 50 | 40 | 33 |
| Hanging ratio (%) | 50 | 50 | 50 |
| Hang length (m) | 10 | 10 | 10 |
| Hang depth (m) | 1.6 | 1.8 | 1.8 |
| Monofilament gauge (mm) | 0.38 | 0.45 | 0.52 |

### 4.2.2 Specific sampling sites

In each inshore area, sites in the 1 to 12 m depth range were chosen to be representative of unvegetated (mud and sand) and seagrass habitats. Seagrass sites in Norfolk Bay, Georges Bay and Prosser Bay consisted almost exclusively of *Heterozostera tasmanica*, although small amounts of *Halophila australis* were present at some sites. Seagrass sites in the Tamar River consisted of *H. tasmanica* and *Posidonia australis*. Site characteristics and sampling gear used for all sites in all inshore areas are presented in Table 4.2.

Four *Heterozostera* and three unvegetated sites were sampled in Norfolk Bay once every two months between February 1995 and December 1996 (Fig. 4.2). Sommers Bay faces south and has a small sand embayment inside a continuous dense bed of *Heterozostera* running parallel to the shoreline (approximately 100 m wide). Prices Bay faces north-east and has a broad intertidal sandflat and a bed of *Heterozostera* running parallel to the shoreline (approximately 150 m wide) that begins as a patchy bed around halfway across the bay and extends to the eastern end of the bay where it is broader and denser. Lime Bay is close to the entrance of Norfolk Bay, faces north and has a broad shallow sub-tidal sandflat inside a bed of *Heterozostera* which runs parallel to the shoreline (approximately 200 m wide). Smooth Island has an area of shallow ground (~5 m) on the eastern shore supporting a more extensive bed of *Heterozostera* than elsewhere in Norfolk Bay (approximately 1 km wide).

Two *Heterozostera* and two unvegetated sites were sampled in Georges Bay seasonally between February 1995 and February 1996 (Fig. 4.3). Steiglitz Beach faces west with a wide bed of *Heterozostera* occurring along the entire beach while Moulting Bay faces east and contains a narrow patchy bed along the shore.

Two *Heterozostera* and one unvegetated site were sampled in Prosser Bay (Fig. 4.4) seasonally from February 1995 to February 1996. Paddys Point faces west with *Heterozostera* occurring in a narrow continuous bed fringing the shore. Shelley Beach faces north with small patchy beds of *Heterozostera*. The mouth of Prosser Bay is sand at around twelve metres deep, often with large quantities of drift filamentous algae.

Four sites in the Tamar River representative of *H. tasmanica* and *P. australis* habitats were sampled seasonally from February 1995 to February 1996 (Fig. 4.5). Lagoon Bay and NW Bank were located in extensive beds of *Posidonia* that were essentially continuous along both sides of the entrance channel near the mouth. Kelso and Sandy Beach were located further up the estuary and consisted of small patchy beds of *Heterozostera*.

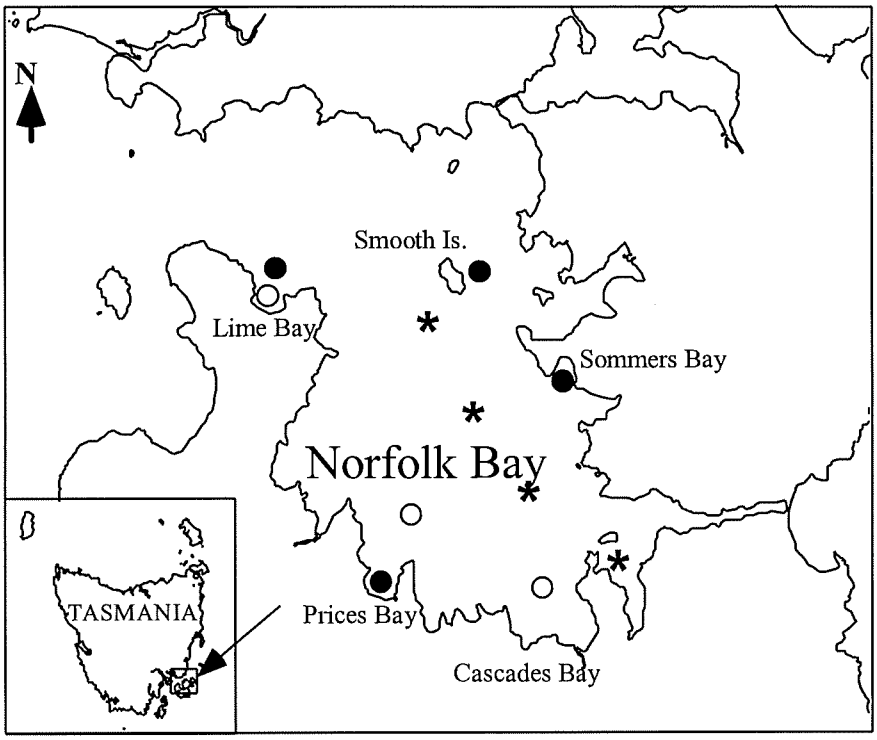



Fig. 4.2 Position of beam trawl and gillnet sampling sites in Norfolk Bay, south-east Tasmania. Dark circles represent *Heterozostera tasmanica* and open circles unvegetated sites. Stars represent ichthyoplankton sampling sites.

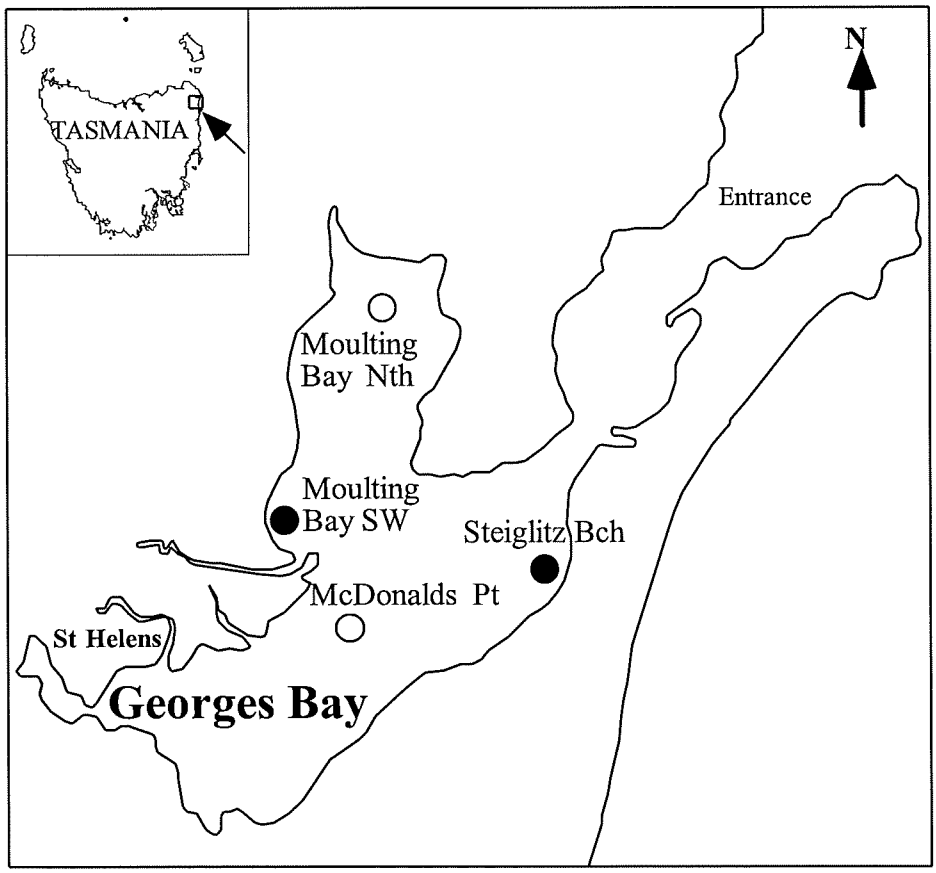



Fig. 4.3 Position of beam trawl and gillnet sampling sites in Georges Bay, north-east Tasmania. Dark circles represent *Heterozostera tasmanica* sites and open circles unvegetated sites.

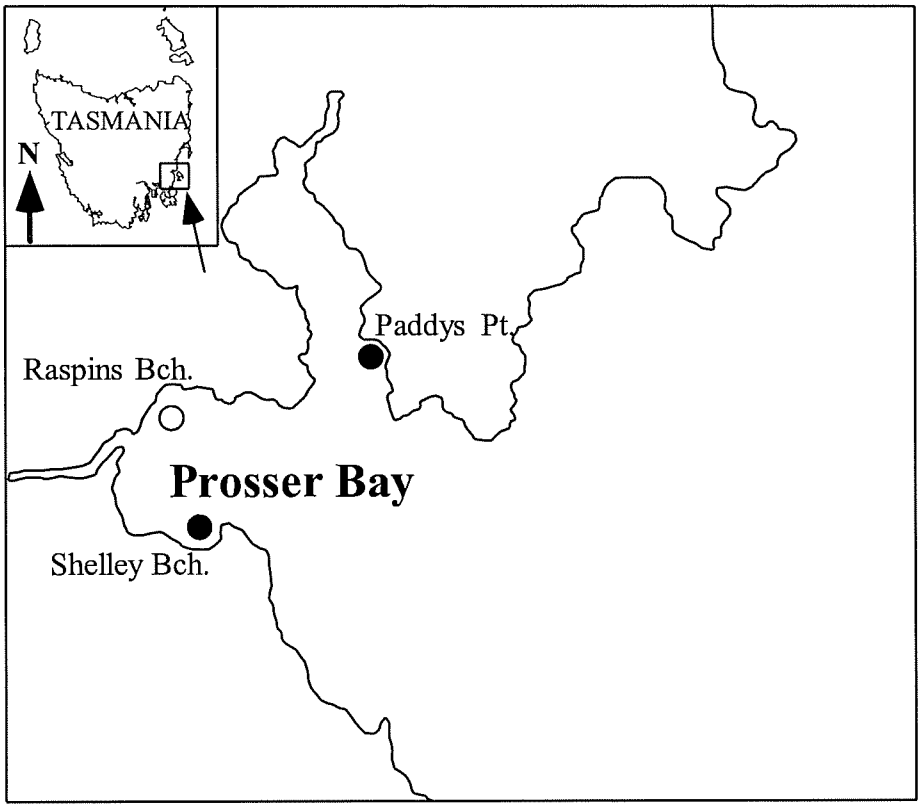



Fig. 4.4 Position of beam trawl and gillnet sampling sites in Prosser Bay, eastern Tasmania. Dark circles represent *Heterozostera tasmanica* sites and open circles unvegetated sites.

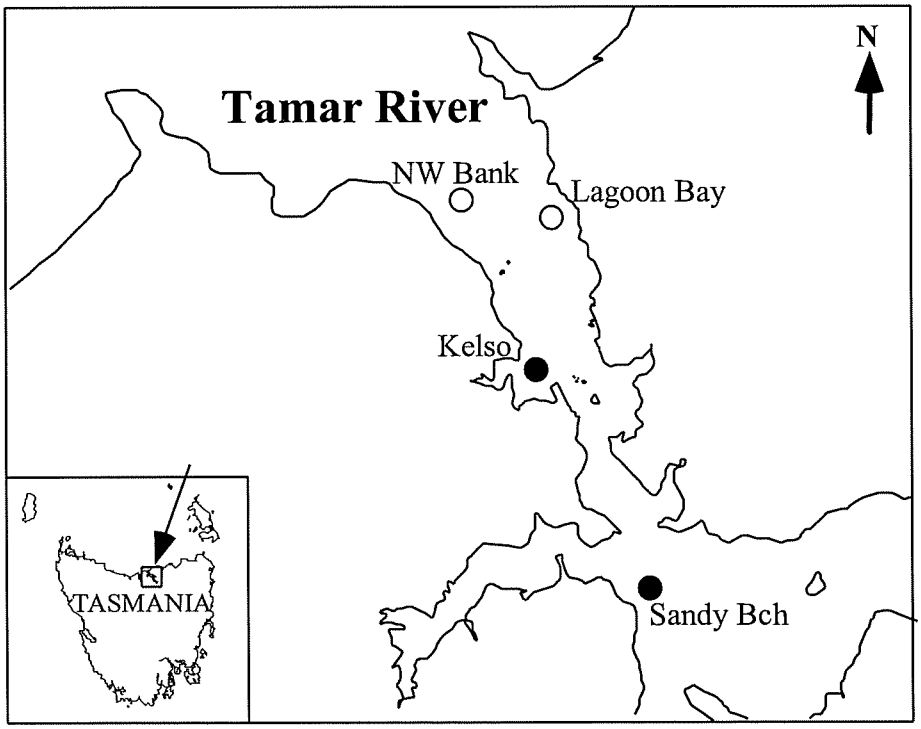



Fig. 4.5 Position of beam trawl sampling sites in the Tamar River, northern Tasmania. Dark circles represent *Heterozostera tasmanica* and open circles *Posidonia australis* sites.

Table 4.2 Habitat characteristics of routine sites sampled in Norfolk Bay, Georges Bay, Prosser Bay and the Tamar River.

| Area/Site | Habitat | Seagrass Density | Depth (m) | Gear deployed | Fetch |
|---|---|---|---|---|---|
| **Norfolk Bay** | | | | | |
| Cascade Bay | Mud | | 8 - 12 | BT, GN | |
| Prices Bay | *H. tasmanica* | Medium | 3 - 6 | BT | 20km NW. |
| Prices Bay | Mud | | 8 - 12 | BT | |
| Lime Bay | *H. tasmanica* | Medium | 3 - 6 | BT, GN | 6.9km Nth. |
| Lime Bay | Sand | | 1 - 3 | BT | 7.0km Nth. |
| Smooth Island | *H. tasmanica* | Low | 4 - 6 | BT, GN | 11.5km Sth. |
| Sommers Bay | *H. tasmanica* | High | 2 - 7 | BT | 9km Sth. |
| **Georges Bay** | | | | | |
| Steiglitz Beach | *H. tasmanica* | High | 2 - 5 | BT, GN | 3km NW. |
| McDonalds Pt. | Mud | | 8 - 12 | BT | |
| Moulting Bay Nth | Mud | | 3 - 5 | BT, GN | |
| Moulting Bay SW. | *H. tasmanica* | Low | 2 - 4 | BT | 3km East |
| **Prosser Bay** | | | | | |
| Paddys Point | *H. tasmanica* | Low | 3 - 5 | BT | 3.7km SW. |
| Raspins Beach | Sand | | 2 - 4 | BT, GN | 15km SE. |
| Shelley Beach | *H. tasmanica* | Low | 3 - 6 | BT | 2.3km Nth. |
| **Tamar River** | | | | | |
| Sandy Beach | *H. tasmanica* | Low | 2 - 6 | BT | 3km East |
| Kelso Bay | *H. tasmanica* | Low | 2 - 5 | BT | 1.5km NE. |
| Lagoon Bay | *P. australis* | High | 2 - 5 | BT | 1.8km NE. |
| NW. Bank | *P. australis* | High | 2 - 4 | BT | 1.3km West |

In addition, in order to monitor temporal patterns of recruitment of juvenile *P. bassensis* and *N. macropterus*, beam trawl sampling was conducted at a single site in North West Bay, located in south-east Tasmania once every two months between March and July 1996, and then monthly until May 1997 (Fig. 4.6). In addition, ad hoc beam trawl sampling was conducted at a single site at Nutgrove Beach in the Derwent River (Fig. 4.6). In both areas sampling sites was unvegetated and consisted of soft-mud between 3-10 m deep.

### 4.2.3 Plankton surveys

The inshore distribution of platycephalid larvae was assessed during ichthyoplankton sampling conducted in October, November and December 1996 at four stations in Norfolk Bay (Fig. 4.2). Samples were collected with a 1 m diameter ring net with 500 µm mesh. Each station consisted of a surface and oblique tow to a maximum depth of 15 m (bottom depth permitting), at a tow speed of ~3 knots. Filtered volume was estimated using calibrated flowmeters. Sampling was restricted to daylight hours (~0600 to 2000 hrs). During inshore surveys, surface and bottom temperatures were recorded with a temperature/depth probe ($^{\pm}$0.1°C, 0.1ppt).

### 4.2.4 Beach seine survey

The fish fauna of nearshore beach habitats were sampled monthly from December 1996 to February 1997 at 27 sites throughout south-eastern Tasmania. The distribution of sampling sites is presented in Fig. 4.6 and site characteristics detailed in Table 4.3. Sites were chosen to be representative of the nearshore intertidal zone with varying levels of exposure. Sampling was conducted with a 25 m beach seine with a 3 m drop and mesh size of 20 mm. At each site three seine net hauls were conducted parallel to shore sampling an area of 40 $m^2$ in each haul. Sampling at all sites was conducted within one hour of high tide and restricted to daylight hours (~0600 to 2000 hrs).

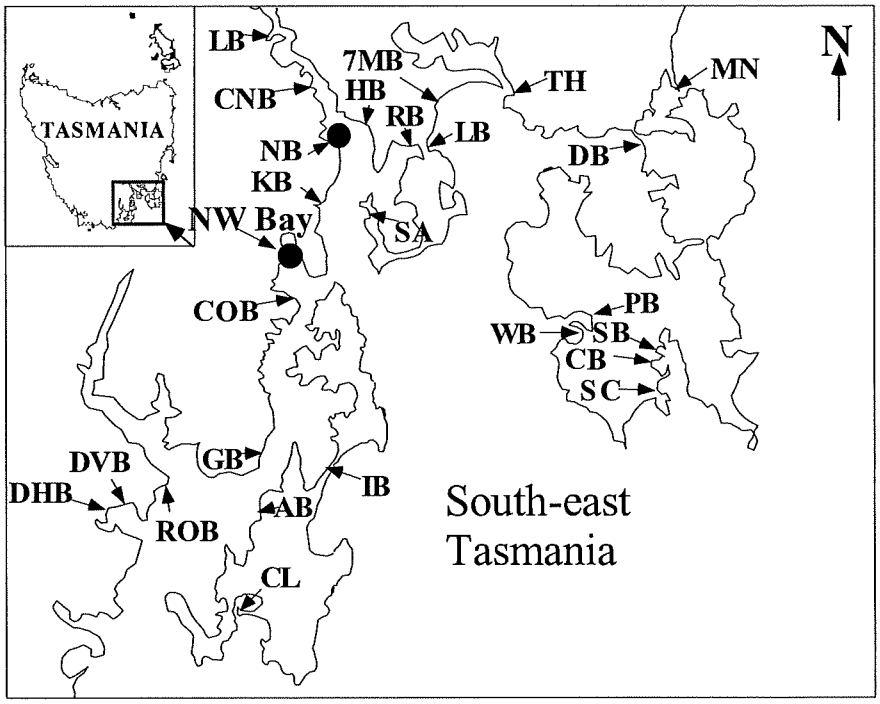



Fig. 4.6 Position of beach seine sampling sites in south-east Tasmania. Details of site codes are presented in Table 4.3. Dark circles represent sites additionally sampled by beam trawl in North West Bay (NW Bay) and Nutgrove Beach (NB).

Table 4.3 Site and habitat characteristics of beach seine sampling sites around south-eastern Tasmania. Site codes are in parentheses.

| Site | Substrate | Exposure | Maximum Depth (m) |
|---|---|---|---|
| Coningham Beach (COB) | Sand | Moderate | 1.5 |
| Gordon Beach (GB) | Sand / *H. tasmanica* | Moderate | 1.0 |
| Roaring Beach (ROB) | Sand | V. high | 1.5 |
| Dover Beach (DVB) | Sand / *H. tasmanica* | Moderate | 1.5 |
| Dover Hotel Beach (DHB) | Silt / Sand | Low | 0.5 |
| Alonnah (AB) | Sand | High | 1.0 |
| Cloudy Lagoon (CL) | Sand / *H. tasmanica* | Moderate | 1.0 |
| Cloudy Lagoon Entrance (CLE) | Sand | Low | 1.5 |
| Isthmus Bay (IB) | Sand | High | 0.5 |
| Lowecroft Bay (LB) | Silt / *H. tasmanica* | Low | 0.5 |
| Cornelian Bay (CNB) | Silt / Sand | Low | 0.5 |
| Nutgrove Beach (NB) | Sand | Moderate | 1.5 |
| Kingston Beach (KB) | Sand | High | 1.5 |
| North West Bay (NW Bay) | Mud | Low | 1.5 |
| Howrah Beach (HB) | Sand | High | 1.5 |
| Rokeby Beach (RB) | Silt / Sand | Moderate | 1.0 |
| South Arm Beach (SA) | Sand | High | 1.5 |
| Lauderdale Beach (LB) | Sand | Moderate | 1.5 |
| Seven Mile Beach (7MB) | Sand | High | 1.5 |
| Tiger Head Ramp (TH) | *H. tasmanica* | Low | 1.0 |
| Dunalley Beach (DB) | Sand | High | 0.5 |
| Marion Bay Narrows (MN) | Sand / *H. tasmanica* | Moderate | 0.5 |
| Parsons Bay (PB) | Silt / Sand | Low | 0.5 |
| White Beach (WB) | Sand | High | 1.5 |
| Stewarts Bay (SB) | Sand | Low | 1.5 |
| Carnarvon Bay (CB) | Sand / *H. tasmanica* | Low | 1.0 |
| Safety Cove (SC) | Sand | Moderate | 1.5 |

### 4.2.5 Snapshot survey

A broader scale snapshot survey of fishes associated with coastal soft-sediment habitats around Tasmania was conducted between January and March 1996. The aim was to describe the fish assemblages associated with these habitats at a broader scale around the state and establish the level of geographic variation within the assemblages. Beam trawl sampling was conducted at twenty-four unvegetated and seagrass sites (Fig. 4.7, Table 4.4). The state was divided up into five regions with up to six sites chosen in each region representative of the available subtidal soft-sediment habitat present in the 2-8 m depth range. As *Posidonia australis* habitat on Flinders Is and on the north east coast occurs in water as deep as 20 m, several deeper seagrass sites were sampled in those areas.

Table 4.4 Site and habitat characteristics of snapshot sampling sites in five survey areas around the coast of Tasmania. Site codes are in parentheses.

| Site | Area | Habitat | Seagrass Density | Depth (m) |
|---|---|---|---|---|
| Dru Point (DP) | Sth. East Coast | *H. tasmanica* | Medium | 1 - 3 |
| Trial Bay (TB) | Sth. East Coast | *H. tasmanica* | Low | 1 - 5 |
| Simpsons Bay (SB) | Sth. East Coast | Sand | | 1 - 4 |
| Cloudy Lagoon (CL) | Sth. East Coast | *H. tasmanica* | High | 2 |
| Lime Bay (LB) | Sth. East Coast | *H. tasmanica* | Medium | 3 - 6 |
| Lime Bay (LB) | Sth. East Coast | Sand | | 1 - 3 |
| Booming Bay (BB) | East Coast | *A. antarctica* | Low | 2 - 5 |
| Little Swanport (LS) | East Coast | *H. tasmanica* | High | 2 - 3 |
| Promise Bay (PB) | East Coast | *H. tasmanica* | Low | 4 - 5 |
| Promise Bay (PB) | East Coast | Sand | | 4 - 5 |
| Franklin Sound (FS) | Flinders Island | *P. australis* | | 2 - 3 |
| Franklin Sound (FS) | Flinders Island | *P. australis* | | 1 - 2 |
| Kent Bay (KB) | Flinders Island | *P. australis* | | 6 - 10 |
| Prime Seal Island (PSI) | Flinders Island | *P. australis* | | 16 - 18 |
| Robbins Island (RIV) | NW Coast | *P. australis* | | 1 - 3 |
| Robbins Island (RIU) | NW Coast | Sand | | 1 - 4 |
| Stanley Beach (ST) | NW Coast | Sand | | 2 - 3 |
| West Inlet (WI) | NW Coast | *H. tasmanica* | Low | 1 - 2 |
| West Inlet (WI) | NW Coast | Sand | | 1 - 3 |
| Tomahawk (TH) | NE Coast | Sand | | 1 - 2 |
| Little Musselroe Bay (LMB) | NE Coast | *P. australis* | Medium | 4 - 6 |
| Waterhouse Island (WHI) | NE Coast | *P. australis* | Medium | 6 - 9 |
| Port Sorell (PS) | NE Coast | *H. tasmanica* | Low | 2 - 4 |
| Port Sorell (PS) | NE Coast | Mud | | 2 - 4 |

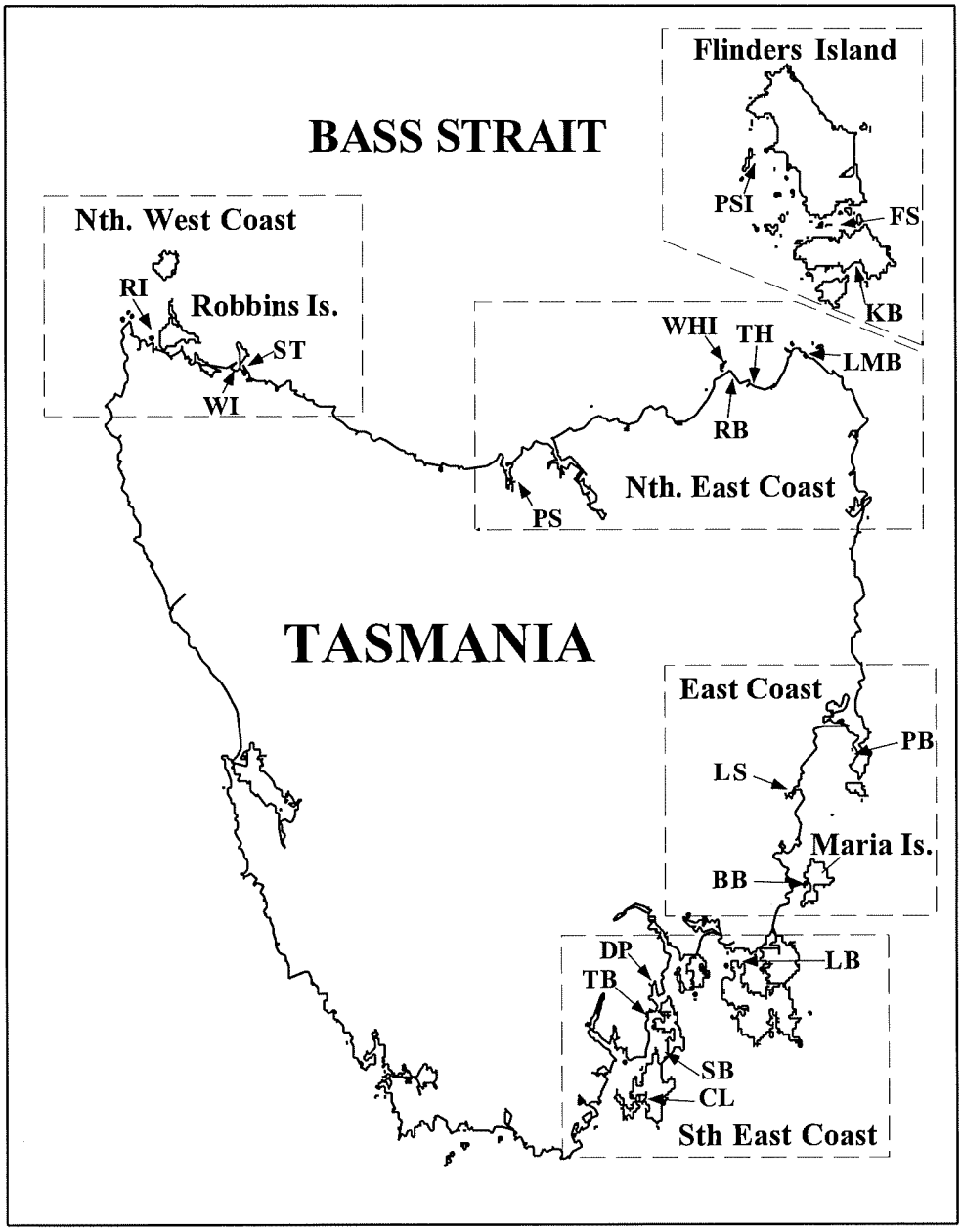



Fig. 4.7 Distribution of snapshot sampling sites around the coast of Tasmania. Details of site codes are presented in Table 4.4.

### 4.3 Laboratory analysis

During routine surveys, a maximum of 20 individuals of each species from each sample were measured for fork length (FL) (length of the shortest caudal ray) in the laboratory. All commercially and recreationally important species were retained and processed for biologicals including fork length (to the nearest millimetre), total weight (to the nearest gram), sex, gonad stage and gonad weight (to the nearest gram). Gonads were staged macroscopically according to the criteria modified from Blackburn and Gartner (1954) (Table 4.5). For *P. bassensis* and *H. melanochir,* sagittal otoliths were removed from all fish, cleaned, dried and stored in envelopes prior to processing.

Gonosomatic index (GSI) was calculated using the formula:

$$GSI = \frac{\text{gonad weight}}{\text{somatic weight}} \times 100$$

Table 4.5 Macroscopic gonad staging criteria used for males and females of all finfish species.

| Stage Category | Macroscopic criteria |
|---|---|
| FEMALES | |
| 1. Virgin | Small strap, less than 3/4 of body cavity. Firm texture. |
| 2. Maturing Virgin | Virgin - Small strap with rounded edge at least 3/4 of body length, pink and transparent.<br>Recovering - as long as body cavity, bloodshot and flabby at posterior. |
| 3. Developing | Almost length of body cavity, opaque and becoming yellow. Ova not discernible. |
| 4. Late Developing | Full length of body cavity, opaque and yellowish pink. Ova discrete. |
| 5. Ripe | Full length of body cavity and swollen occupying all available space. Ovary and ova become translucent. |
| 6. Running ripe | Eggs expressed with slight pressure. Ovary pinkish, clear and granular. |
| 7. Spent | Slack and bloodshot. Few residual oocytes present. |
| MALES | |
| 1. Virgin | Small strap, less than 3/4 of body cavity. Firm texture. |
| 2. Maturing Virgin | Virgin - Small strap with sharp edge at least 3/4 of body length, pink and opaque.<br>Recovering - as long as body cavity, bloodshot and flaccid at posterior. |
| 3. Developing | Almost length of body cavity, opaque and becoming larger. |
| 4. Late Developing | Full length of body cavity and larger. |
| 5. Ripe | Full length of body cavity and swollen occupying all available space. No milt expressed with slight pressure. |
| 6. Running ripe | Milt expressed with slight pressure. Testes granular. |
| 7. Spent | Flaccid and bloodshot. |

## 4.4 Statistical analysis

Variation in fish abundance and number of species per tow between areas, sites and sampling period was assessed using analysis of variance (ANOVA). As the experimental design is specific for each chapter, full details of ANOVA model designs are presented in each relevant section. In general, however, data were tested for conformity to the assumptions of ANOVA using the $F_{max}$ test for heteroscedascity (Sokal and Rohlf 1981) and by examining residual plots, and transformed when necessary. Where no significant interaction terms were detected, Ryans Q test was used to identify significant differences among means for main effects. Ryans Q test is considered to be the most powerful post-hoc test which allows the user to control experiment-wise error rate (Day and Quinn 1989). Calculations were performed with the Peritz FORTRAN program (Martin and Toothaker 1989).

Differences in fish community structure between areas, and between sites within areas were analysed by non-metric multi-dimensional scaling (MDS) and analysis of similarities (ANOSIM) (Clarke 1993, Clarke and Warwick 1994). A matrix of ranked similarities was generated using the Bray-Curtis similarity index (Bray and Curtis 1957) applied to $x^{0.25}$ transformed abundance data. This transformation has the effect of down-weighting the influence of highly abundant species when generating a measure of distance between two samples (Clarke and Green 1988); a desirable effect where abundances are dominated by a small number of species.

Two-dimensional MDS plots were generated from the similarity matrix. Non-metric MDS is a highly flexible ordination technique that allows the user to employ a distance measure, transformation and standardisation that is appropriate to the questions being asked, and the distributions within the data. Marine communities are often characterised by a large suite of species with sparse, highly skewed abundances and as such, distributions are unlikely to ever meet the assumptions of normality required in parametric multi-variate analysis. Non-metric MDS constructs a configuration of samples in low-dimensional space which preserves the rank order of distances between samples. By basing the analysis on rank order only, the method is free from distributional assumptions. A 'stress' value is generated, which describes the degree of distortion involved in reducing the data to the required number of dimensions. Where stress values were high (>0.18), arrangements of points were checked against groups generated by group-average clustering. Where grouping were similar by both methods, MDS plots were accepted as an appropriate representations of the data (see Clarke and Warwick 1994).

To test whether fish communities were significantly different between samples, habitats, areas, and sites within areas, analysis of similarities (ANOSIM) was applied to the ranked similarity matrix. The algorithm for ANOSIM involves the calculation of a global test statistic which compares variability among replicates within groups selected on an *a-priori* basis, to variability between groups. A simulated distribution for this statistic is generated by repeatedly and randomly reassigning group labels to samples within the similarity matrix and recalculating the test statistic. The significance level is calculated by referring the observed value of this statistic to its permutation distribution. If the observed value appears unlikely to have come form the permutation distribution, there is strong evidence to reject the null hypothesis (Clarke and Warwick 1994). Similarity percentage (SIMPER) analysis was used to calculate species contributions to the average dissimilarity between groups.

# 5. Detailed Results - Community composition

## 5.1 Broad scale spatial and temporal patterns in *Heterozostera* communities

### 5.1.1 Introduction

Within coastal soft-sediment habitats, seagrass beds are widely recognised as an important habitat for fishes by providing protection and increased food resources compared to bare substrates (see Bell and Pollard 1989). Despite the extensive distribution of seagrass habitats in coastal waters throughout Tasmania, there have been few baseline studies describing seagrass fish assemblages, particularly at a range of spatial scales. The recent finding that a significant decline has occurred in the extent of seagrass beds throughout estuarine and coastal waters of Tasmania (Rees 1993) has highlighted the need to examine the dynamics of seagrass fish assemblages throughout the state. Given the significance of seagrass beds as nursery areas for commercial and recreationally important species throughout southern Australia (Bell and Pollard 1989), there is a clear need to assess the importance of such areas throughout Tasmania.

To meaningfully interpret any data on changes in fish abundance or community composition, it is important to first understand natural population fluctuations, both spatial and temporal, at a range of scales. A considerable number of studies have examined variability in seagrass fish assemblages on the scale of kilometres or tens of kilometres (Burchmore *et al.* 1984, Bell *et al.* 1988, De Ben *et al.* 1990, Gray *et al.* 1990). Few studies, however, have examined variability in seagrass fish assemblages across estuaries or embayments on the scale of hundreds of kilometres. Fish assemblages associated with seagrass in three estuaries along the coast of New South Wales were found to be significantly different and related to hydrographic and bathymetric differences (Ferrell *et al.* 1993). In contrast, Gray *et al.* (1996) found no consistent inter-estuary differences along the coast of New South Wales, although there were large variations in the abundance of individual species.

The aim of this chapter is therefore to examine the spatial and temporal variations in abundance and community composition of fishes associated with *Heterozostera tasmanica* beds located in the three areas along the east coast of Tasmania - Norfolk Bay, Prosser Bay and Georges Bay. This was done in order to compare the significance of *Heterozostera* habitats across a large section of coastline, and therefore provide information at a spatial scale relevant to management of such habitats throughout Tasmania.

### 5.1.2 Methods

Comparisons of the demersal fish fauna associated with *Heterozostera tasmanica* beds were made across three areas along the east coast of Tasmania, chosen to be representative of coastal environments with available *Heterozostera* habitats. Two distinct *Heterozostera* beds were chosen within each area to allow estimates of variability within and between areas. Fish were sampled with a beam trawl on five occasions between summer 1995 and summer 1996. Full details of sampling gear and regime is presented in Chapter 4.

Table 5.1.1 Site characteristics of *Heterozostera tasmanica* beds sampled in Norfolk Bay, Prosser Bay and Georges Bay. Site codes are in parentheses.

| Area | Site | Seagrass Density | Depth (m) | Fetch |
|---|---|---|---|---|
| **Norfolk Bay** | Sommers Bay (SB) | High | 3 - 6 | 9km Sth. |
| | Lime Bay (LB) | Medium | 3 - 6 | 6.9km Nth. |
| **Prosser Bay** | Paddys Point (PP) | Low | 3 - 5 | 3.7km SW. |
| | Shelley Beach (SHB) | Low | 3 - 6 | 2.3km Nth. |
| **Georges Bay** | Steiglitz Beach (STB) | High | 2 - 5 | 3km NW. |
| | Moulting Bay SW. (MB) | Low | 2 - 4 | 3km East |

Variation in fish abundance and number of species per tow between areas, sites and sampling period was assessed using a three-way nested ANOVA. Area and site were considered to be random factors, as they were chosen to be indicative of seagrass habitats on the east coast of Tasmania. Date of sampling was also considered random, as there was no *a-priori* reason for choosing sampling dates; they were chosen to give an even spread of samples throughout the year; (samples did, however, fall within quarters, and for ease of reference these have been referred to by season names). The resulting ANOVA model, with three random factors (one nested), provides no appropriate test of main effects of area, as no denominator for calculation of the variance ratio which includes all appropriate error terms is available. Main effects for area were therefore estimated using a *quasi-F* calculation (Winer 1971), developed according to the protocol presented by Zar (1996). Where no significant interaction terms were detected, Ryans Q test was used to identify significant differences among means for main effects.

Differences in fish community structure between the three areas, and between sites within areas, were analysed by non-metric multi-dimensional scaling (MDS) and analysis of similarities (ANOSIM). A matrix of ranked similarities was generated using the Bray-Curtis similarity index (Bray and Curtis 1957) applied to $x^{0.25}$ transformed abundance data. Two-dimensional MDS plots were generated from the similarity matrix. Significance of differences between fish communities from the three areas, and two sites within each area, was tested by analysis of similarities (ANOSIM) applied to the ranked similarity matrix. Similarity percentage (SIMPER) analysis was used to calculate species contributions to the average dissimilarity between groups.

### 5.1.3 Results

#### *5.1.3.1 Environmental variability*

All sites were predominantly marine, with greatest range in bottom salinities being observed at Moulting Bay in Georges Bay (minimum 29 ppt). Bottom temperature varied seasonally from averages of 16° - 18° C in February to 8° - 10° C in August, but site and area differences were small. Visual inspection during the period of the study revealed distinct seasonal changes in seagrass standing stock. In late autumn and early winter, large

amounts of seagrass detritus were caught in the beam-trawl indicating die-back at this time of year. Seagrass standing stock at all sites was lowest in winter. However, site differences were evident with seasonal die-back most noticeable at sites initially assessed in summer as having low seagrass biomass. At Prosser Bay, die-back was almost total, with very few standing blades apparent during winter. In contrast, all sites in Norfolk Bay and Steiglitz Beach in Georges Bay showed lower levels of seasonal change.

*5.1.3.2 Catch composition*

A total of 9,312 fish from 49 species and 20 families were captured in 90 beam trawl tows (Table 5.1.1). A full list of scientific and common names is presented in Appendix 1. Thirteen species and six families were represented by a single individual, and were excluded from statistical analysis. The catch was dominated by bridled leatherjacket (*Acanthaluteres spilomelanurus)*, spotted pipefish (*Stigmatopora argus)* and rock whiting (*Neoodax balteatus)*, comprising 90% of total fish abundance, although their relative dominance varied between areas. For example, *A. spilomelanurus* were not evenly distributed between areas making up over 60% of the total catch at Norfolk Bay, while at Georges Bay they only made up around 2%.

There was considerable overlap in the abundant species across areas. Sixteen species were common to all areas, and these made up 97.5% of total fish abundance. Georges Bay and Norfolk Bay shared 20 species, Georges Bay and Prosser Bay shared 19 species and Norfolk Bay and Prosser Bay shared 18 species. Of the 36 species represented by more than a single individual, 11 species were caught at only one of the three areas; 5 species were unique to Georges Bay, 4 species to Prosser Bay and 2 species were unique to Norfolk Bay. These species make up only 0.3% of the total catch. A further 10 species were absent from one area. Most abundant species unique to one area was *Arenogobius bifrenatus* of which 4 individuals were caught.

*5.1.3.3 Fish abundance and species richness*

Abundance of demersal fishes in *Heterozostera* beds varied significantly between all areas (Table 5.1.3, Fig. 5.1.1A). Post-hoc tests indicate that abundance was higher in Norfolk Bay compared to Georges Bay, which was higher than Prosser Bay (Table 5.1.4). Differences between sites within area are significant ($p=0.045$, Table 5.1.3), but only just. Differences were detected between sites within Georges Bay, but not within Norfolk Bay and Prosser Bay (Table 5.1.4).

Patterns of seasonal variability in total fish abundance differ across areas (Fig. 5.1.1A). While seasonal differences in abundance are not significant, some seasonality is evident in Norfolk Bay and Georges Bay with abundances highest in autumn. No seasonal trend is evident at Prosser Bay. There was no significant difference in the number of species between area or season (Table 5.1.3), but tends to be lower in Prosser Bay (Fig. 5.1.1B).

Table 5.1.2 Total number of individuals and % of total individuals for fish collected by beam trawl between summer 1995 and 1996 on seagrass sites at Norfolk Bay, Georges Bay and Prosser Bay.

| | Georges Bay | | | | Norfolk Bay | | | | Prosser Bay | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | STB | | MB | | SB | | LB | | SHB | | PP | |
| Species | n | % | n | % | n | % | n | % | n | % | n | % |
| **Urolophidae** | | | | | | | | | | | | |
| *Urolophus cruciatus* | 1 | 0.1 | 0 | 0 | 0 | 0 | 0 | 0 | 8 | 5.2 | 0 | 0 |
| **Moridae** | | | | | | | | | | | | |
| *Pseudophycis bachus* | 1 | 0.1 | 1 | 0.1 | 36 | 0.8 | 14 | 0.5 | 5 | 3.3 | 1 | 0.8 |
| **Sygnathidae** | | | | | | | | | | | | |
| *Urocampus carinirostris* | 0 | 0 | 2 | 0.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Hippocampus abdominalis* | 1 | 0.1 | 5 | 0.7 | 3 | 0.1 | 1 | 0.04 | 0 | 0 | 0 | 0 |
| *Mitotichthys semistriatus* | 0 | 0.0 | 0 | 0 | 5 | 0.1 | 6 | 0.2 | 2 | 1.3 | 0 | 0 |
| *Stigmatopora argus* | 700 | 57.5 | 528 | 69.6 | 768 | 17.8 | 231 | 8.4 | 21 | 13.7 | 23 | 18.5 |
| *Stigmatopora nigra* | 5 | 0.4 | 108 | 14.2 | 0 | 0 | 0 | 0 | 3 | 2.0 | 10 | 8.1 |
| *Vanacampus phillipi* | 45 | 3.7 | 5 | 0.7 | 2 | 0.05 | 14 | 0.5 | 5 | 3.3 | 1 | 0.8 |
| **Scorpaenidae** | | | | | | | | | | | | |
| *Helicolenus barathri* | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.1 | 0 | 0 | 0 | 0 |
| *Gymnapistes marmoratus* | 58 | 4.8 | 12 | 1.6 | 38 | 0.9 | 20 | 0.7 | 7 | 4.6 | 4 | 3.2 |
| **Platycephalidae** | | | | | | | | | | | | |
| *Platycephalus bassensis* | 2 | 0.2 | 6 | 0.8 | 5 | 0.1 | 2 | 0.1 | 11 | 7.2 | 1 | 0.8 |
| **Apogonidae** | | | | | | | | | | | | |
| *Vincentia conspersa* | 64 | 5.3 | 10 | 1.3 | 39 | 0.9 | 33 | 1.2 | 6 | 3.9 | 0 | 0 |
| **Carangidae** | | | | | | | | | | | | |
| *Pseudocaranx dentex* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 1.3 | 0 | 0 |
| **Odacidae** | | | | | | | | | | | | |
| *Haletta semifasciata* | 0 | 0 | 0 | 0 | 1 | 0.02 | 0 | 0 | 0 | 0 | 1 | 0.8 |
| *Neoodax balteatus* | 244 | 20.0 | 5 | 0.7 | 310 | 7.2 | 650 | 23.7 | 22 | 14.4 | 9 | 7.3 |
| **Bovichtidae** | | | | | | | | | | | | |
| *Pseudaphritis urvillii* | 0 | 0 | 3 | 0.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Clinidae** | | | | | | | | | | | | |
| *Cristaceps australis* | 12 | 1.0 | 0 | 0.0 | 10 | 0.2 | 39 | 1.4 | 16 | 10.5 | 29 | 23.4 |
| *Heteroclinus puellarum* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 2.4 |
| *H. perspicillatus* | 6 | 0.5 | 1 | 0.1 | 0 | 0 | 6 | 0.2 | 2 | 1.3 | 1 | 0.8 |
| **Gobiidae** | | | | | | | | | | | | |
| *Callogobius mucosus* | 1 | 0.1 | 0 | 0 | 1 | 0.02 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Favonigobius tamarensis* | 0 | 0 | 3 | 0.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nesogobius pulchellus* | 7 | 0.6 | 22 | 2.9 | 2 | 0.05 | 1 | 0.04 | 3 | 2.0 | 7 | 5.6 |
| *Arenigobius bifrenatus* | 1 | 0.1 | 3 | 0.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Pseudogobius olurum* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 1.3 | 0 | 0 |
| *Nesogobius* sp.1 | 4 | 0.3 | 7 | 0.9 | 5 | 0.1 | 5 | 0.2 | 23 | 15.0 | 3 | 2.4 |
| *Nesogobius* sp.3 | 1 | 0.1 | 1 | 0.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Pleuronectidae** | | | | | | | | | | | | |
| *Ammotretis rostratus* | 0 | 0 | 2 | 0.3 | 2 | 0.05 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Taratretis derwentensis* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 1.6 |
| *Rhombosolea tapirina* | 0 | 0 | 14 | 1.8 | 2 | 0.05 | 0 | 0 | 1 | 0.7 | 0 | 0 |
| **Monocanthidae** | | | | | | | | | | | | |
| *Acanthaluteres vittiger* | 2 | 0.2 | 0 | 0 | 13 | 0.3 | 8 | 0.3 | 1 | 0.7 | 2 | 1.6 |
| *A. spilomelanurus* | 43 | 3.5 | 8 | 1.1 | 3055 | 70.9 | 1703 | 62.2 | 12 | 7.8 | 8 | 6.5 |
| *Meuschenia australis* | 1 | 0.1 | 4 | 0.5 | 1 | 0.02 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Brachaluteres jacksonianus* | 7 | 0.6 | 3 | 0.4 | 0 | 0 | 1 | 0.04 | 0 | 0 | 2 | 1.6 |
| *Eubalichthys gunnii* | 1 | 0.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 15 | 12.1 |
| *Meuschenia freycineti* | 11 | 0.9 | 6 | 0.8 | 8 | 0.2 | 1 | 0.04 | 1 | 0.7 | 2 | 1.6 |
| **Diodontidae** | | | | | | | | | | | | |
| *Diodon nicthemerus* | 0 | 0 | 0 | 0 | 1 | 0.02 | 1 | 0.04 | 0 | 0 | 0 | 0 |

Table 5.1.3 Three way ANOVA (area, site and date random factors, site nested in area) of log transformed abundances of fish and number of species ($N.tow^{-1}$) caught at *Heterozostera tasmanica* sites in Norfolk Bay, Georges Bay and Prosser Bay between summer 1995 and 1996.

| | | | Log abundance | | | Number of species | | |
|---|---|---|---|---|---|---|---|---|
| Factor | Hypothesis | DF | MS | F | Prob. | MS | F | Prob. |
| Area, *A* | *quasi F* | 2 | 70.6 | 19.4 | 0.009 | 89.5 | 10.7 | 0.085 |
| Site(Area), *B(A)* | *B(A)/B(A)C* | 3 | 2.7 | 3.6 | 0.045 | 7.6 | 1.6 | 0.250 |
| Season, *C* | *C/AC* | 4 | 4.3 | 2.6 | 0.119 | 7.8 | 1.4 | 0.323 |
| Area*Date, *AC* | *AC/B(A)C* | 8 | 1.7 | 2.2 | 0.102 | 5.7 | 1.2 | 0.389 |
| Site(Area)*Season, *B(A)C* | *BC/E* | 12 | 0.8 | 1.0 | 0.460 | 4.9 | 1.2 | 0.325 |
| Error, *E* | | 60 | 0.8 | | | 4.2 | | |

Table 5.1.4 Ryans Q-test of abundances of fish ($N.tow^{-1}$) between areas and sites from ANOVA presented in Table 5.1.3. Bold underlining indicates no significant difference.

| **Prosser Bay** | | **Georges Bay** | | **Norfolk Bay** | |
|---|---|---|---|---|---|
| PP | SHB | MB | STB | LB | SB |
| 2.2 | 1.9 | 3.2 | 4.1 | 4.8 | 5.4 |

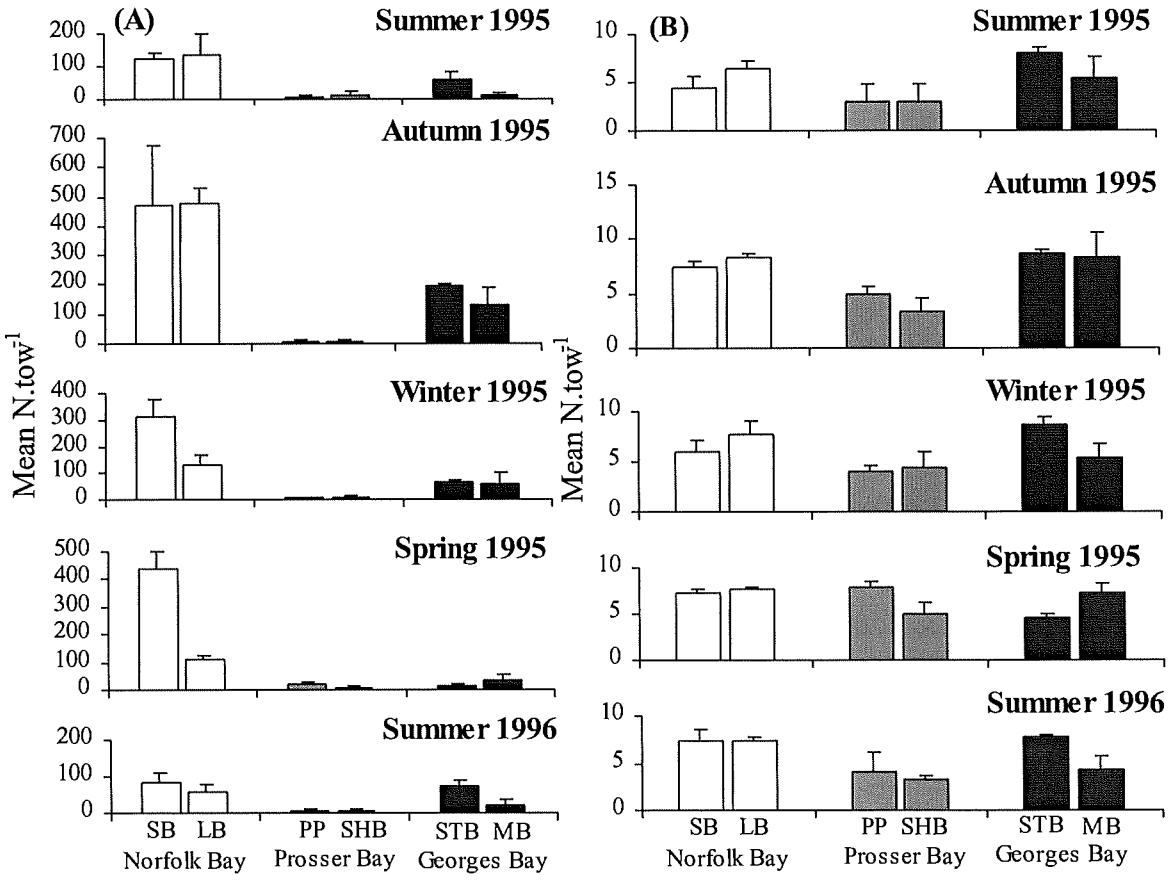



Fig. 5.1.1 (A) Mean seasonal abundance ($N.tow^{-1}$) of fish and (B) mean number of species ($N.tow^{-1}$) in *Heterozostera tasmanica* sites in Norfolk Bay, Georges Bay and Prosser Bay between summer 1995 and 1996. Error bars are standard error.

### *5.1.3.4 Community composition*

Multi-dimensional scaling reveals a clear separation between fish communities from the three areas (Fig. 5.1.2). Pairwise comparison (ANOSIM) shows this separation to be highly significant ($p<0.001$). Similarity percentage analysis identified the 10 species contributing most to the separation of fish assemblages, and their percentage contribution to the separation (Table 5.1.5). The percentage difference between assemblages from areas described by these 10 species varied from 53.4% (Prosser Bay / Georges Bay) to 66.5% (Norfolk Bay / Prosser Bay).

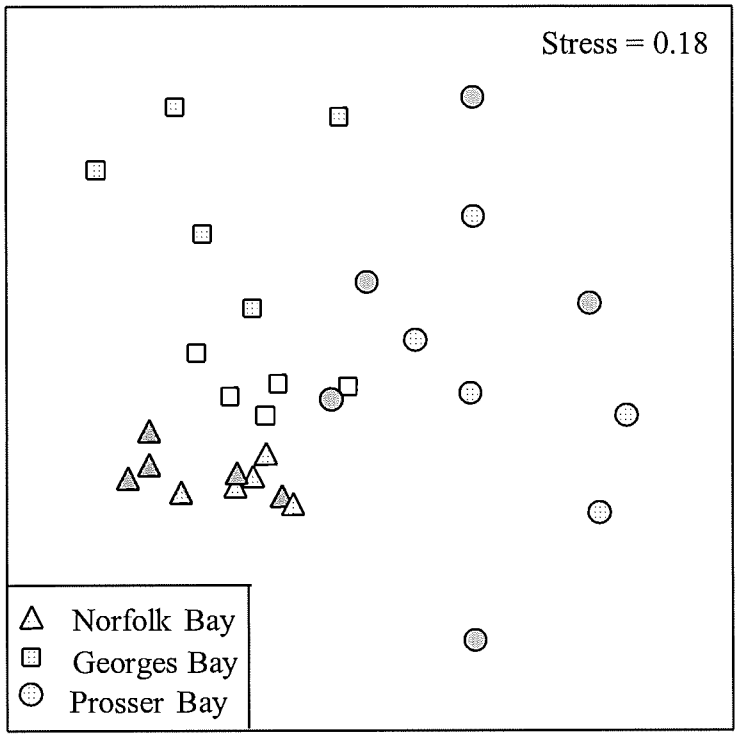



Fig. 5.1.2 Non-metric multi-dimensional scaling of fish communities sampled by beam trawl at *Heterozostera tasmanica* sites in Norfolk Bay, Georges Bay and Prosser Bay between summer 1995 and 1996.

Table 5.1.5 Similarity percentage pair-wise analysis of fish communities from three areas based on Bray-curtis similarities between 4th root transformed fish abundance. Percentage of the variability between fish communities explained by varying abundances of individual species are presented.

| | Average abundance | | | % variability explained | | |
|---|---|---|---|---|---|---|
| Species | Norfolk (1) | Prosser(2) | Georges (3) | 1 vs 2 | 1 vs 3 | 2 vs 3 |
| *Acanthaluteres spilomelanurus* | 475.8 | 2.0 | 5.1 | 17.5 | 15.4 | 4.3 |
| *Neoodax balteatus* | 96.0 | 3.1 | 24.9 | 10.4 | 7.9 | 6.7 |
| *Stigmatopora argus* | 99.9 | 4.4 | 122.8 | 7.0 | 4.8 | 8.2 |
| *Vincentia conspersa* | 7.2 | 0.6 | 7.4 | 5.9 | 3.2 | 6.6 |
| *Stigmatopora nigra* | 0.0 | 1.3 | 11.3 | 3.8 | 6.8 | 5.1 |
| *Gymnapistes marmoratus* | 5.8 | 1.0 | 7.0 | 5.9 | 3.7 | 5.9 |
| *Cristiceps australis* | 4.5 | 4.9 | 1.2 | 3.8 | 4.7 | 5.2 |
| *Pseudophycis bachus* | 5.0 | 0.6 | 0.2 | 5.1 | 5.9 | 2.5 |
| *Vanacampus phillipi* | 1.6 | 0.6 | 5.0 | 3.5 | 3.4 | 4.8 |
| *Nesogobius* sp. 1 | 1.0 | 2.6 | 1.1 | 3.6 | 3.2 | 4.1 |
| Total % | | | | 66.5 | 59.0 | 53.4 |

Inter-area differences were due to differences in abundance of the common species. *Acanthaluteres spilomelanurus* occurs in large numbers only at the two Norfolk Bay sites, and is present sporadically in small numbers at other sites (Fig. 5.1.3A). Abundance peaked in autumn at both Norfolk Bay sites. Highest abundances of *Neoodax balteatus* in Norfolk Bay occurred at Lime Bay, while Steiglitz Beach in Georges Bay had similar abundances to Sommers Bay in Norfolk Bay (Fig. 5.1.3B). Although abundance at Lime Bay follows the same seasonal pattern seen for *A. spilomelanurus*, there are no distinct seasonal changes at other sites.

*Stigmatopora argus* is present in similar abundance in Norfolk Bay and Georges Bay, and low abundance in Prosser Bay (Fig. 5.1.4A). Abundances of *S. argus* in Georges Bay sites and Lime Bay in Norfolk Bay peak in autumn, while high abundances occurs in winter and spring at Sommers Bay in Norfolk Bay. *Stigmatopora nigra* was absent from Norfolk Bay and most abundant in Georges Bay (Fig. 5.1.4B). *Vincentia conspersa* is abundant in Norfolk Bay and Georges Bay, but temporal trends, and distribution between sites within areas, are inconsistent (Fig. 5.1.5A). In contrast to these species which all occur in relatively low numbers in Prosser Bay, *Cristiceps australis* occurred in greatest numbers in Prosser Bay and Norfolk Bay, with the strongest peak occurring in Prosser Bay in summer 1995 (Fig. 5.1.5B).

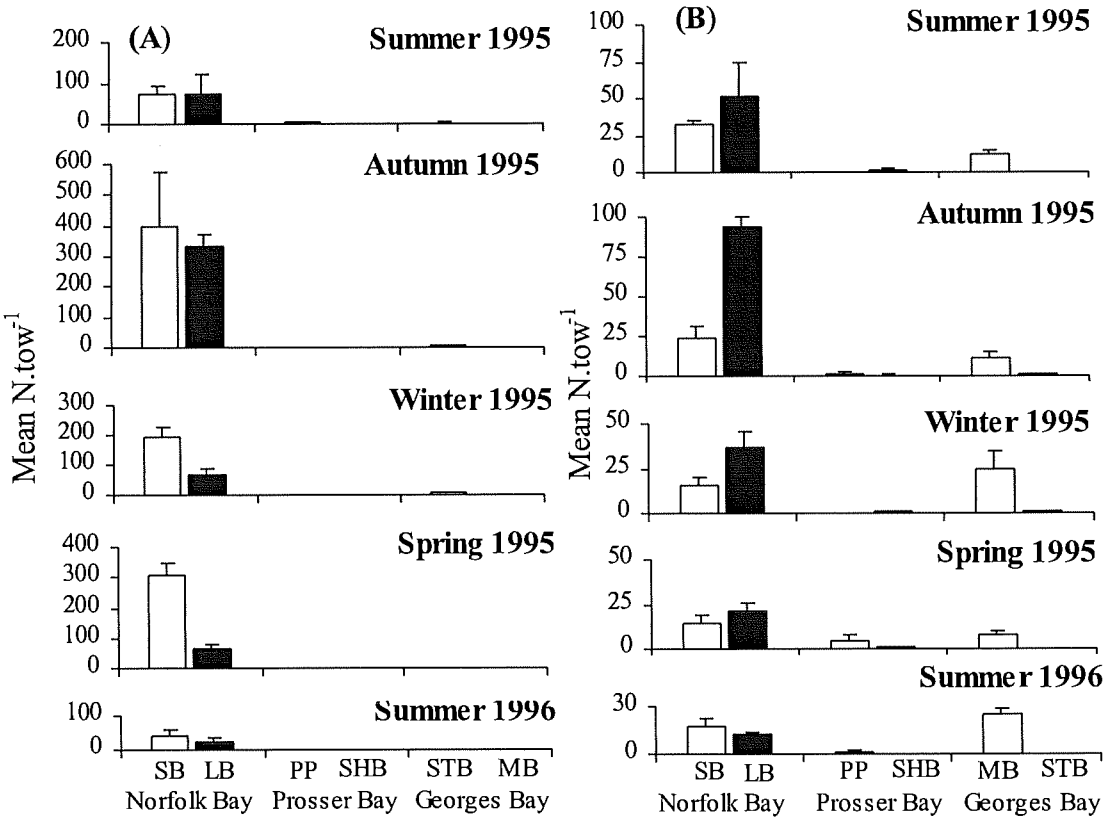



Fig. 5.1.3 Mean seasonal abundance ($N.tow^{-1}$) of (A) *Acanthaluteres spilomelanurus* and (B) *Neoodax balteatus* in Norfolk Bay, Prosser Bay and Georges Bay. Error bars are standard error.

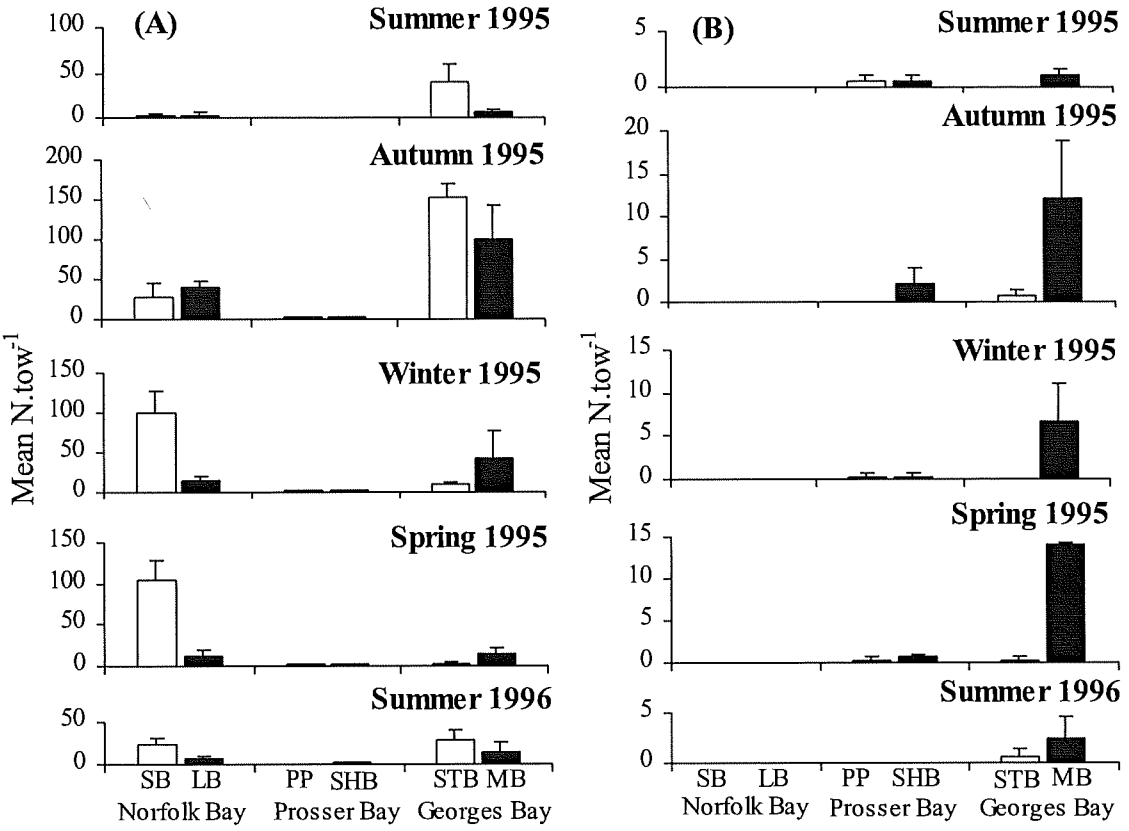



Fig. 5.1.4 Mean seasonal abundance (N.tow$^{-1}$) of (A) *Stigmatopora argus* and (B) *Stigmatopora nigra* in Norfolk Bay, Prosser Bay and Georges Bay. Error bars are standard error.

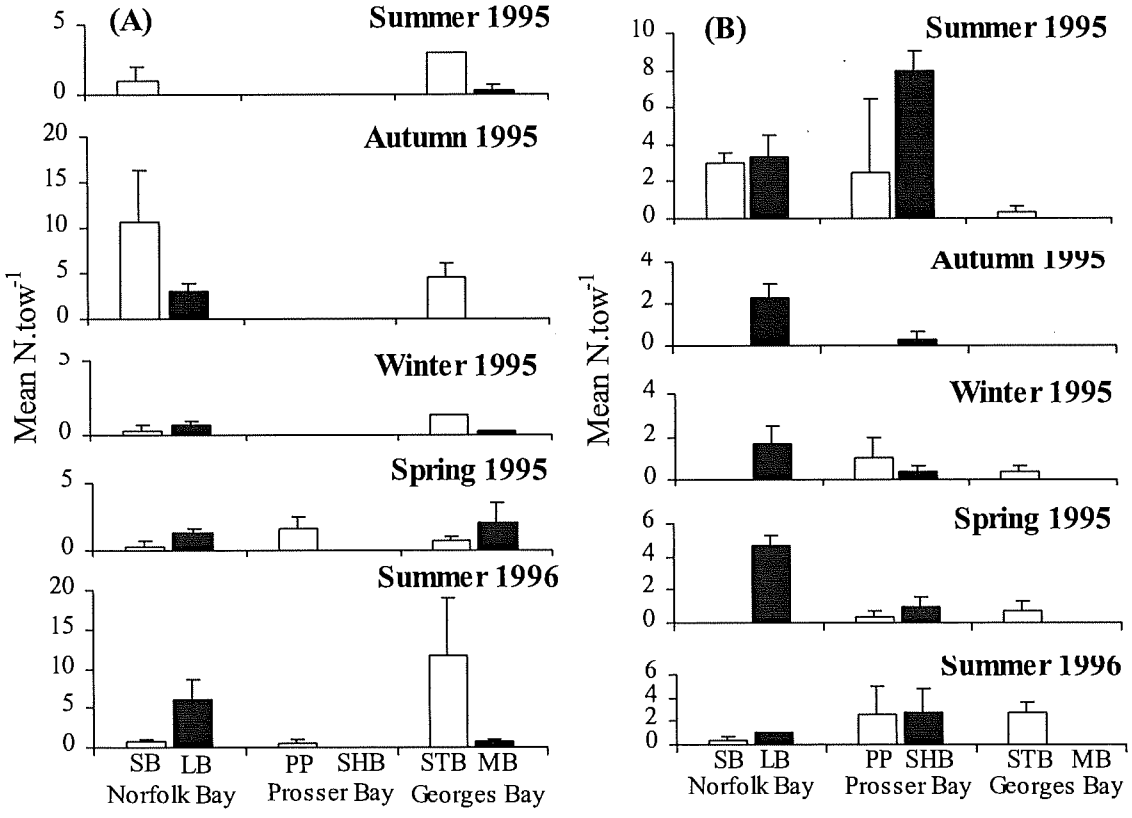



Fig. 5.1.5 Mean seasonal abundance (N.tow$^{-1}$) of (A) *Vincentia conspersa* and (B) *Cristiceps australis* in Norfolk Bay, Prosser Bay and Georges Bay. Error bars are standard error.

No significant differences were found between communities from sites within Norfolk Bay (p=0.206), or Prosser Bay (p=0.056), but site differences in Georges Bay were significant (p=0.008). The two species contributing most to this separation show opposing responses to site, with *N. balteatus* abundant at Steiglitz Beach, and *S. nigra* abundant at Moulting Bay (Table 5.1.6).

Table 5.1.6 Similarity percentage analysis of fish communities at sites within Georges Bay based on Bray-curtis simmilarities between 4th root transformed fish abundance data. Average dissimilarity between groups = 58.8%.

| Species | Average abundance | | Percent |
|---|---|---|---|
| | Steiglitz Beach | Moulting Bay | |
| *Neoodax balteatus* | 40.1 | 1.0 | 9.07 |
| *Stigmatopora nigra* | 0.8 | 21.6 | 7.48 |
| *Cristiceps australis* | 3.3 | 0 | 6.07 |
| *Stigmatopora argus* | 117.6 | 105.6 | 5.62 |
| *Rhombosolea tapirina* | 0 | 2.8 | 5.61 |

Despite considerable variability in the abundance of the dominant species, few consistent seasonal trends in community composition were detected. The exception is in Prosser Bay where samples taken in both summer 1995 and 1996 are distinct from those taken in all other seasons (Fig. 5.1.6). The difference (ANOSIM) is driven by low summer abundances of most common species including *Neoodax balteatus* and *Nesogobius pulchellus*, and high abundances of newly recruiting *Cristiceps australis* (see Fig. 5.1.5B).

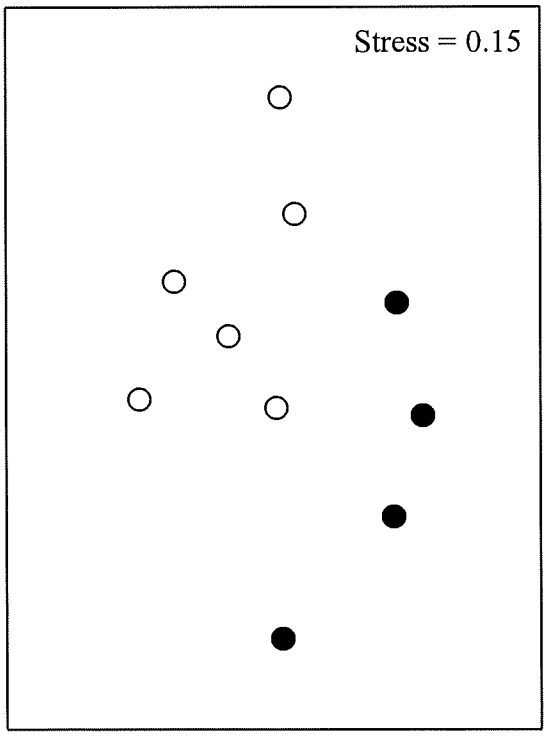



Fig. 5.1.6 Non-metric multi-dimensional scaling of fish communities sampled by beam trawl at *Heterozostera tasmanica* sites in Prosser Bay between summer 1995 and 1996. Closed circles represent summer samples while open circles are all other seasons.

### 5.1.4 Discussion

The present study has demonstrated that there were consistent differences in the abundance of fish in *Heterozostera* beds between all areas, abundances being highest in Norfolk Bay and lowest in Prosser Bay. However, as the number of species are similar between areas, the inter-area differences in abundance primarily reflected differences in abundance of the common species. For example, *Acanthaluteres spilomelanurus* was highly abundant in Norfolk Bay but was virtually absent from Georges Bay and Prosser Bay. Likewise, *Stigmatopora argus* was abundant in Norfolk Bay and Georges Bay but uncommon in Prosser Bay, while *S. nigra* was abundant in Georges Bay but absent from Norfolk Bay.

Given that the dominant species were most abundant in only one or two of the three areas, it is not surprising that the multi-dimensional scaling revealed a clear separation between fish communities between areas. This is consistent to that found for fish assemblages associated with *Posidonia* beds in three estuaries along the coast of NSW where differences were related to physical differences in hydrography and seagrass density between estuaries (Ferrell *et al.* 1993). Such physical differences were also present between the three areas in the present study. While there were few differences in salinity and temperature, the level of exposure ranged from the semi-exposed marine embayment of Prosser Bay to the enclosed waters of Georges Bay. Physical exposure through strong wave action and currents has been suggested to influence the structuring of seagrass fish assemblages through reducing abundances (Jenkins and Sutherland 1997). However, in the present study highest abundances were found at sites with the greatest fetch suggesting that the greater depth of *Heterozostera* beds (~3-6 m) results in a reduced influence of wave action.

The between area differences along the east coast of Tasmania contrasts with a similar study by Gray *et al.* (1996), who found that over a similar spatial scale in New South Wales (300km) there were no consistent inter-estuary differences in fish communities in *Zostera* habitats. There are several plausible explanations for this difference. Physical variability between the embayments sampled in this study may be greater than those of estuaries on the NSW coast. Secondly, many species sampled in the NSW study spawn in the open ocean and recruit to seagrass beds within the estuaries resulting in a much greater potential for larval mixing. Most species caught in the present study are resident within, and spawn within, the coastal embayments, and hence the opportunity for larval mixing and redistribution is reduced. Oceanic flow regimes in the areas are also quite different. In NSW, the East Australian Current runs parallel with the coast potentially providing similar recruits to each estuary.

There was also considerable variation in the density of *Heterozostera* beds between areas with highest density in Norfolk Bay and lowest in Prosser Bay. Jenkins and Sutherland (1997) found such differences in seagrass complexity to influence the abundance of individual species, particularly *Acanthaluteres.* spp which showed a preference for more

complex beds. This is consistent with that of *A. spilomelanurus* in Norfolk Bay where high abundances at Sommers Bay could be related to the high seagrass density. However, similar high seagrass density occurred at Steiglitz Beach where abundances of *A. spilomelanurus* were consistently low, this species having the highest percentage contribution to the separation of Norfolk Bay and Georges Bay in the multivariate analysis. Similar differences in abundances of *A. spilomelanurus* have been found between sites within an estuary (Jenkins and Sutherland 1997), and between estuaries (Ferrell *et al.* 1993), suggesting patchiness in larval supply at a range of spatial scales.

Such variations in larval supply may actually influence the relative abundance of most species that are closely associated with *Heterozostera* habitats, particularly *A. spilomelanurus* and *Stigmatopora* spp., which were uncommon in the low density beds in Prosser Bay. Continued loss of seagrass in the Prosser Bay area over the past four decades (Rees 1993), has led to the present situation in which seagrass densities are low in summer, with almost total dieback during winter. It appears that in Prosser Bay the seasonal presence of seagrass is in itself either not sufficient to attract high numbers of fish, post-settlement mortality is high or abundances of pre-recruits is low. As few seagrass beds exist for some tens of kilometres from the beds in Prosser Bay it is likely that larval supply may be limited. This suggests that the decline in density in isolated seagrass beds may result in a consistent decrease in abundance of seagrass associated species.

Sampling in several embayments demonstrated that there was a greater number of species that were unique to a particular embayment compared to than between sites within a single embayment. Around 30% of species captured more than once were unique to one embayment, compared with only 3% being unique to one site within Norfolk Bay. This was slightly higher in Georges Bay where *S. nigra* showed a preference for Steiglitz Beach and *Neoodax balteatus* preferred Moulting Bay, leading to a significant difference between sites reflecting the dominance of these species.

The finding that most abundant species were common in only one or two of the three areas has management implications in that conservation of *Heterozostera* beds in all areas is necessary to protect a wide range of seagrass associated fish species. In addition, management of such habitats is required at this large spatial scale if preservation of rarer species is of high importance to maintain biodiversity.

## 5.2 Spatial-temporal patterns of *Heterozostera* communities in Norfolk Bay

### 5.2.1 Introduction

Environmental differences between estuaries and embayments can be quite considerable due to physical structure and area. It is not surprising therefore, that fish community variability between areas can be high. However, environmental variability encountered within coastal bays and estuaries can also have considerable influence on spatial and temporal patterns of fish abundance and community composition. Examples of variables influencing fish community composition on small spatial scales include level of exposure (Blaber and Whitfield 1977, Last 1983), salinity (Sogard *et al.* 1989, Gray *et al.* 1990), turbidity (Blaber and Blaber 1980), hydrography (Sogard *et al.* 1987), position in the estuary (Bell *et al.* 1988), influence of adjacent habitats (Weinstein and Heck 1979, Ayvazian and Hyndes 1995), depth (Bell *et al.* 1992), habitat type (Gray *et al.* 1996) and variations in seagrass height and density (Stoner 1983, Bell and Westoby 1986a, b).

As different fish assemblages are often typical of different areas within an estuary, an understanding of the extent of variation is important information for management of such areas. The extent of environmental variation within an estuary is often unique to individual estuaries and is dependant on a range of factors including bathymetry, hydrography and catchment characteristics. In chapter 5.1, significant intra-area differences were found between *Heterozostera tasmanica* sites in only one out of three coastal areas. In addition, as patterns of fish assemblages may vary over time there is a clear need to conduct studies at a range of temporal scales.

The aim of this chapter is to examine in more detail the variations in abundance and community composition of fishes associated with *Heterozostera* beds within Norfolk Bay. In addition, interannual variations are examined in order to assess longer term temporal patterns in seagrass fish assemblages in south-eastern Tasmania.

### 5.2.2 Methods

Four sites in Norfolk Bay representative of the available *Heterozostera tasmanica* habitat were sampled bimonthly from February 1995 to December 1996 (Table 5.2.1). The demersal fish fauna was sampled at each site with a beam trawl, with catch rates calculated as the number of fish per tow. Full details of survey area and sampling gear is presented in Chapter 4.

Table 5.2.1 Site and habitat characteristics of *Heterozostera tasmanica* beds sampled in Norfolk Bay. Site codes are in parentheses.

| Site | Habitat | Seagrass Density | Depth (m) | Fetch |
|---|---|---|---|---|
| Sommers Bay (SB) | *H. tasmanica* | High | 3 - 6 | 9km Sth. |
| Prices Bay (PB) | *H. tasmanica* | Medium | 3 - 6 | 20km NW. |
| Lime Bay (LB) | *H. tasmanica* | Medium | 3 - 6 | 6.9km Nth. |
| Smooth Island (SI) | *H. tasmanica* | Low | 4 - 6 | 11.5km Sth. |

Variation in fish abundance and number of species per tow between years, months and sites was assessed using a three-way ANOVA. Site was considered to be a random factors, as sites were a subset of available seagrass beds within Norfolk Bay, and were chosen to be indicative of seagrass habitats in Norfolk Bay. Date of sampling was also considered random, as there was no *a-priori* reason for choosing sampling dates; they were chosen to give an even spread of samples throughout the year based on available resources. Year was considered a fixed factor. The resulting ANOVA model provides no appropriate test of main effects of year, as no denominator for calculation of the variance ratio which includes all appropriate error terms is available. Main effects for area were therefore estimated using a *quasi-F* calculation (Winer 1971), developed according to the protocol presented by Zar (1996). Where no significant interaction terms were detected, Ryans Q test was used to identify significant differences among means for main effects.

Inter-annual and inter-site differences in fish community structure were investigated by MDS ordination and a two-way crossed ANOSIM (site and year) applied to a matrix of Bray-Curtis similarities between samples. All abundance data were first transformed ($x^{0.25}$) to reduce the weighting of highly abundant species. A limitation of the two-way ANOSIM algorithm is it's inability to look beyond the level of main effects to detect interactions. As such, where interactions do occur, they may obscure the significance of main effects. Separate MDS plots for the two years of the study show a degree of variability in the distances between sites, and suggest that there may be an interaction between the site and year variables. To this end, a separate ANOSIM was run for each year. Similarity percentage (SIMPER) analysis was used to calculate species contributions to the average dissimilarity between groups.

## 5.2.3 Results

### *5.2.3.1 Environmental variability*

All sites showed consistent oceanic influence, and neither temperature or salinity varied between sites. Seasonal variations in temperature and salinity are shown in Fig. 5.2.1. A degree of seasonal seagrass die-back was observed in winter months of both years, however *Heterozostera* beds remained at all sites throughout the study.

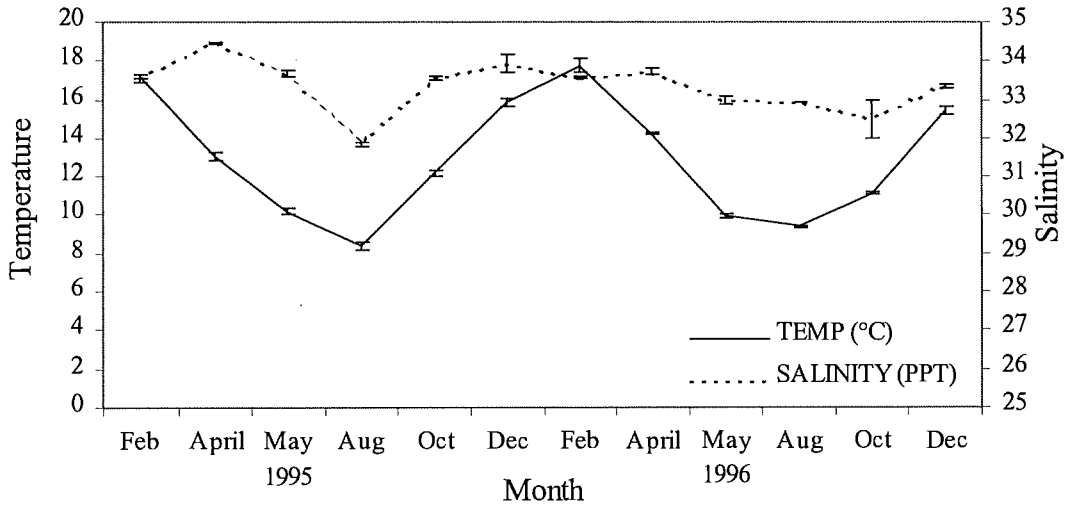



Fig. 5.2.1 Mean monthly temperature and salinity for *Heterozostera tasmanica* sampling sites in Norfolk Bay between February 1995 and December 1996. Error bars are standard error.

### *5.2.3.2 Catch composition*

A total of 26,224 fish (46 species, 21 families) were caught in 144 beam trawl tows. Thirteen species were only captured once; the remaining thirty-three species were used in further analysis (Table 5.2.2). A full list of scientific and common names is presented in Appendix 1. The catch was dominated by *Acanthaluteres spilomelanurus*, *Neoodax balteatus*, and *Stigmatopora argus*, making up at least 92% of the catch at each site. *Acanthaluteres spilomelanurus* was most abundant at Sommers Bay, Prices Bay and Lime Bay, and *N. balteatus* was most abundant at Smooth Island. *Nesogobius sp.6* (Last *et al.* 1983 *Nesogobius sp.3*) was the only species caught more than once that was unique to any site, being caught at Lime Bay only.

### *5.2.3.3 Fish abundance and species richness*

Abundance of demersal fishes in *Heterozostera* beds in Norfolk Bay varied significantly between all sites and months (Table 5.2.3, Fig. 5.2.2). Post-hoc tests indicate that abundance was highest at Sommers Bay and lowest at Smooth Island (Table 5.2.4). In terms of months, abundance was significantly higher in May than all other months in both years, with lowest abundances in December and February which showed no significant difference (Table 5.2.4). While temporal changes were not as prominent at Smooth Island as at the other sites, timing of peaks in abundance were consistent (Fig. 5.2.2).

Table 5.2.2 The total number of individuals and percentage of the total individuals for fish taxa collected by beam trawl at *Heterozostera tasmanica* sites at Sommers Bay, Prices Bay, Lime Bay and Smooth Island in Norfolk Bay between February 1995 and December 1996.

| Species | Sommers Bay | | Prices Bay | | Lime Bay | | Smooth Is. | |
|---|---|---|---|---|---|---|---|---|
| | n | % | n | % | n | % | n | % |
| **Moridae** | | | | | | | | |
| *Pseudophycis bachus* | 41 | 0.4 | 3 | 0.06 | 15 | 0.2 | 6 | 0.2 |
| **Ophidiidae** | | | | | | | | |
| *Genypterus tigerinus* | 1 | 0.01 | 1 | 0.02 | 0 | 0 | 0 | 0 |
| **Atherinidae** | | | | | | | | |
| *Atherinason hepsetoides* | 7 | 0.06 | 0 | 0 | 1 | 0.01 | 1 | 0.04 |
| **Sygnathidae** | | | | | | | | |
| *Hippocampus abdominalis* | 3 | 0.03 | 2 | 0.04 | 1 | 0.01 | 1 | 0.04 |
| *Mitotichthys semistriatus* | 9 | 0.08 | 3 | 0.06 | 18 | 0.2 | 4 | 0.2 |
| *Stigmatopora argus* | 1351 | 12.5 | 477 | 9.1 | 733 | 9.6 | 90 | 3.6 |
| *Vanacampus phillipi* | 21 | 0.2 | 48 | 0.9 | 35 | 0.5 | 15 | 0.6 |
| **Scorpaenidae** | | | | | | | | |
| *Helicolenus barathri* | 2 | 0.02 | 1 | 0.02 | 5 | 0.07 | 5 | 0.2 |
| *Gymnapistes marmoratus* | 54 | 0.5 | 32 | 0.6 | 23 | 0.3 | 56 | 2.2 |
| **Triglidae** | | | | | | | | |
| *Lepidotrigla papilio* | 0 | 0 | 1 | 0.02 | 2 | 0.03 | 2 | 0.08 |
| **Platycephalidae** | | | | | | | | |
| *Platycephalus bassensis* | 6 | 0.05 | 24 | 0.5 | 2 | 0.03 | 14 | 0.6 |
| **Apogonidae** | | | | | | | | |
| *Vincentia conspersa* | 66 | 0.6 | 57 | 1.1 | 71 | 0.9 | 21 | 0.8 |
| **Odacidae** | | | | | | | | |
| *Haletta semifasciata* | 2 | 0.02 | 3 | 0.06 | 1 | 0.01 | 0 | 0 |
| *Neoodax balteatus* | 1782 | 16.4 | 1863 | 35.6 | 1923 | 25.3 | 1262 | 49.8 |
| **Clinidae** | | | | | | | | |
| *Cristiceps australis* | 24 | 0.2 | 34 | 0.65 | 84 | 1.1 | 45 | 1.8 |
| *Heteroclinus adelaidae* | 0 | 0 | 0 | 0 | 4 | 0.05 | 4 | 0.2 |
| *Heteroclinus perspicillatus* | 0 | 0 | 0 | 0 | 7 | 0.09 | 4 | 0.2 |
| **Gobiidae** | | | | | | | | |
| *Nesogobius hinisbyi* | 1 | 0.01 | 17 | 0.3 | 1 | 0.01 | 7 | 0.3 |
| *Nesogobius pulchellus* | 11 | 0.1 | 3 | 0.06 | 5 | 0.07 | 2 | 0.1 |
| *Nesogobius* sp.1 | 8 | 0.07 | 12 | 0.2 | 7 | 0.09 | 12 | 0.5 |
| **Pleuronectidae** | | | | | | | | |
| *Ammotretis rostratus* | 2 | 0.02 | 2 | 0.04 | 0 | 0 | 1 | 0.04 |
| *Rhombosolea tapirina* | 2 | 0.02 | 8 | 0.13 | 2 | 0.03 | 2 | 0.08 |
| **Monocanthidae** | | | | | | | | |
| *Acanthaluteres vittiger* | 21 | 0.2 | 2 | 0.04 | 13 | 0.2 | 6 | 0.2 |
| *A. spilomelanurus* | 7399 | 68.3 | 2618 | 50.1 | 4633 | 60.1 | 966 | 38.2 |
| *Meuschenia australis* | 1 | 0.01 | 1 | 0.02 | 0 | 0 | 1 | 0.04 |
| *Brachaluteres jacksonianus* | 2 | 0.02 | 0 | 0 | 5 | 0.07 | 1 | 0.04 |
| *Meuschenia freycineti* | 20 | 0.2 | 17 | 0.3 | 5 | 0.07 | 4 | 0.2 |
| **Diodontidae** | | | | | | | | |
| *Diodon nicthemerus* | 2 | 0.02 | 3 | 0.06 | 1 | 0.01 | 0 | 0 |

Table 5.2.3 Three way ANOVA (year: fixed factor; month and site: random factors) of log transformed abundances of fish (N.tow$^{-1}$) at four *Heterozostera tasmanica* sites in Norfolk Bay between February 1995 and December 1996.

| Factor | Hypothesis | DF | MS | F | Prob. |
|---|---|---|---|---|---|
| Year, *A* | *quasi F* | 1 | 1.24 | 0.65 | 0.505 |
| Month, *B* | *B/BC* | 5 | 8.34 | 13.03 | <.001 |
| Site, *C* | *C/BC* | 3 | 16.53 | 25.81 | <.001 |
| Year*Month, *AB* | *AB/ABC* | 5 | 2.53 | 3.35 | 0.063 |
| Year*Site, *AC* | *AC/ABC* | 3 | 0.32 | 0.34 | 0.794 |
| Month*Site, *BC* | *BC/E* | 15 | 0.64 | 0.85 | 0.621 |
| Year*Month*Site, *ABC* | *ABC/E* | 15 | 0.94 | 1.25 | 0.252 |
| Error, *E* | | 96 | 0.76 | | |

Table 5.2.4 Ryans Q-test for abundances of fish (N.tow$^{-1}$) between site and month from ANOVA presented in Table 5.2.3. Bold underlining indicates no significant difference.

| **Site** | Smooth Is. | Prices Bay | Lime Bay | Sommers Bay |
|---|---|---|---|---|
| Mean abundance | 70 | 145 | 211 | 301 |

| **Month** | Dec | Feb | Oct | Apr | Aug | May |
|---|---|---|---|---|---|---|
| Mean abundance | 67 | 116 | 175 | 192 | 212 | 331 |

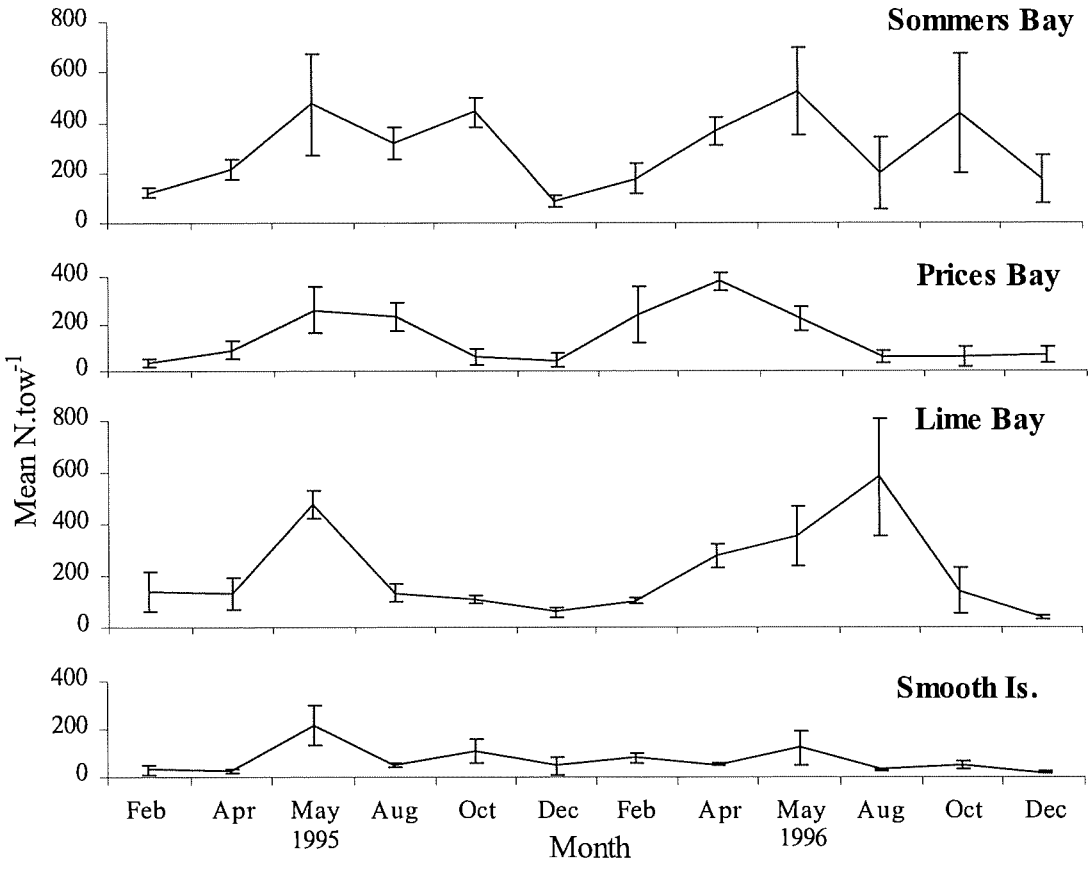



Fig. 5.2.2 Mean abundance of fish (N.tow$^{-1}$) at four *Heterozostera tasmanica* sites in Norfolk Bay between February 1995 and December 1996. Error bars are standard error.

The temporal variability in total abundance was driven by changes in numbers of all three highly abundant species. Due to the dominance of *A. spilomelanurus*, particularly at Sommers Bay, patterns in abundance of all species closely paralleled changes in the abundance of this species (Fig. 5.2.3). *Neoodax balteatus* (Fig. 5.2.4), and *S. argus* (Fig. 5.2.5) vary similarly, with peaks in abundance ranging from April to August in both 1995 and 1996.

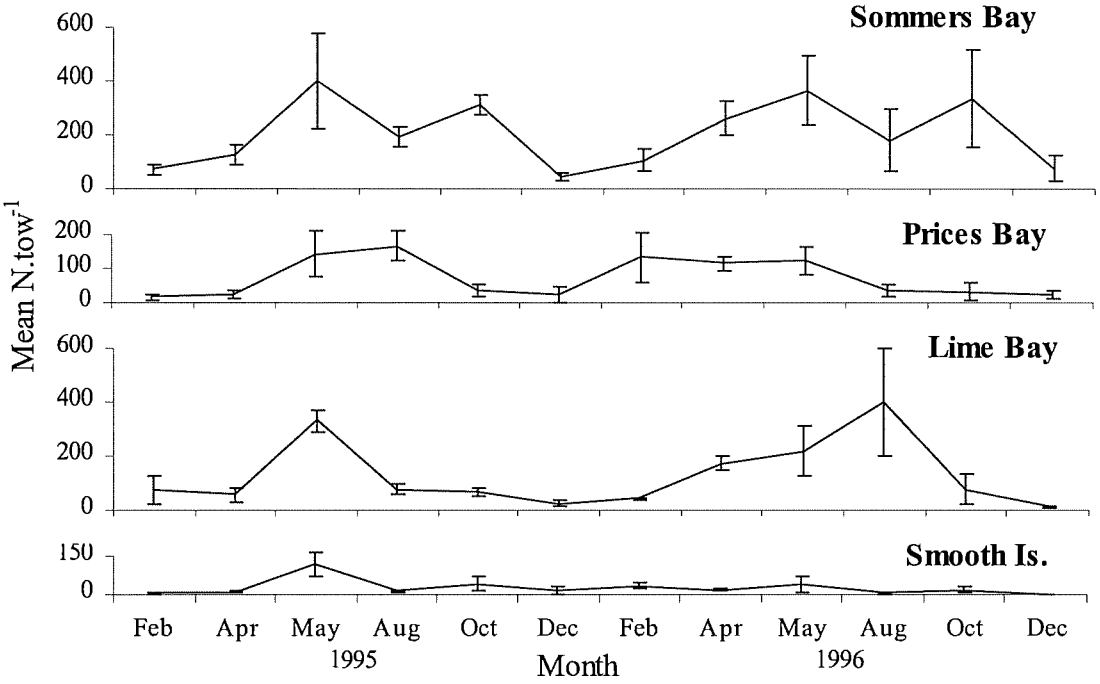



Fig. 5.2.3 Mean abundance (N.tow$^{-1}$) of *Acanthaluteres spilomelanurus* at four *Heterozostera tasmanica* sites in Norfolk Bay between February 1995 and December 1996. Error bars are standard error.

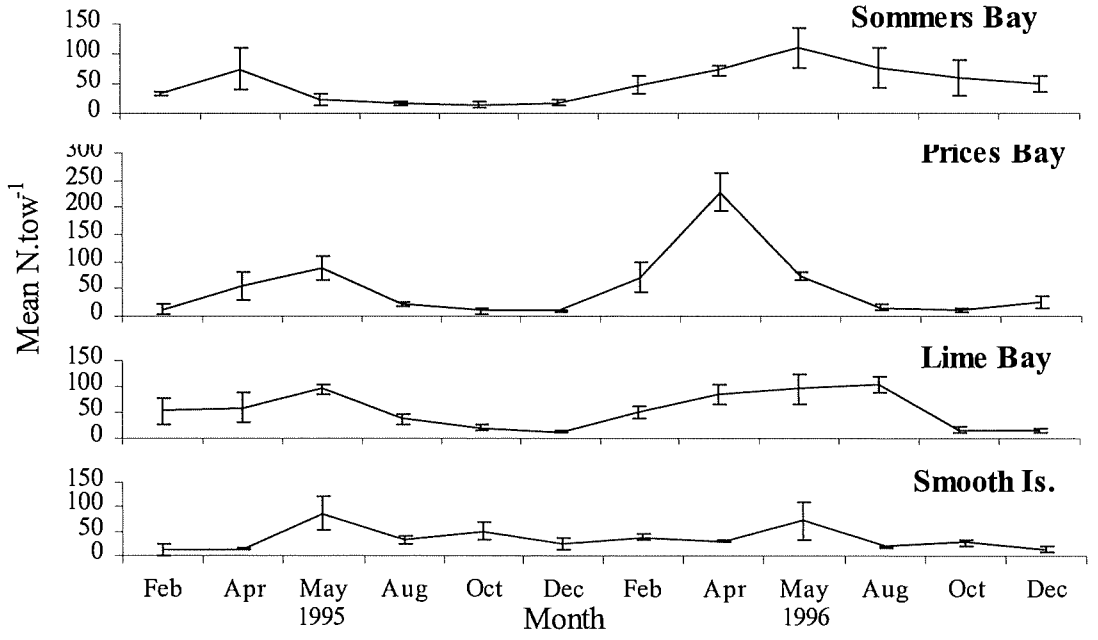



Fig. 5.2.4 Mean abundance (N.tow$^{-1}$) of *Neoodax balteatus* at four *Heterozostera tasmanica* sites in Norfolk Bay between February 1995 and December 1996. Error bars are standard error.

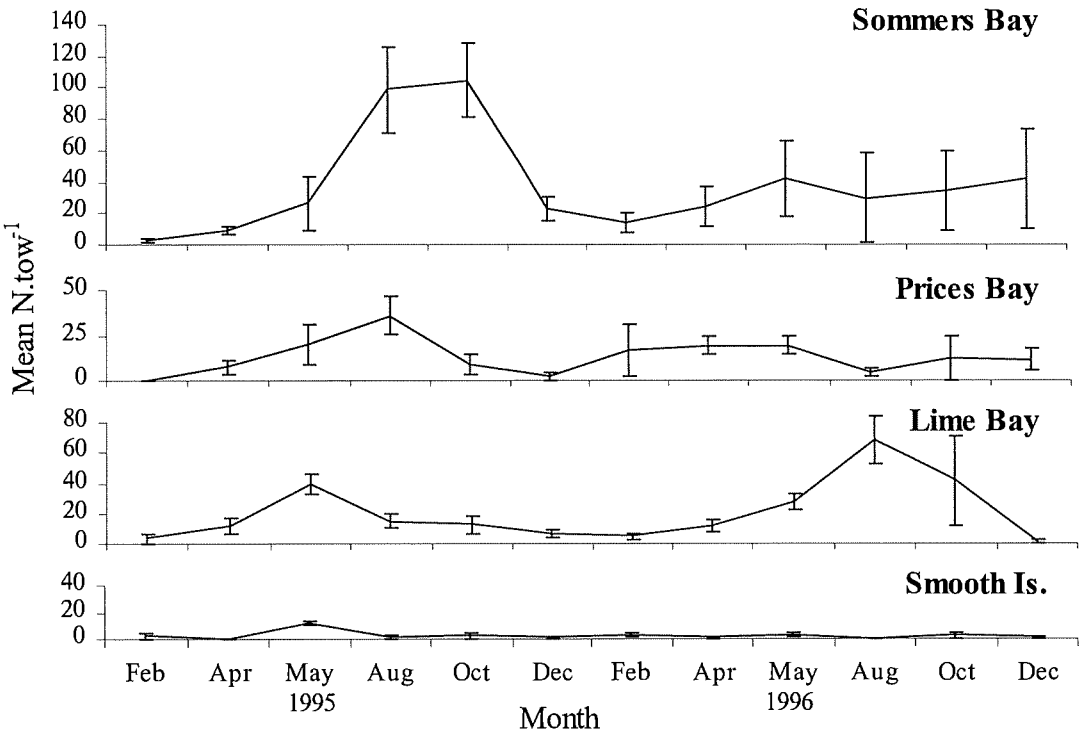



Fig. 5.2.5 Mean abundance (N.tow$^{-1}$) of *Stigmatopora argus* at four *Heterozostera tasmanica* sites in Norfolk Bay between February 1995 and December 1996. Error bars are standard error.

The number of species of fish in *Heterozostera* beds in Norfolk Bay varied significantly between sites (Table 5.2.5, Fig. 5.2.6). Post-hoc tests showed that more species were caught at Sommers Bay and Lime Bay than at Smooth Island, while Prices Bay was not significantly different to any of the other sites (Table 5.2.6). Temporal changes in number of species were less consistent than changes in abundance as no significant difference between sampling months were detected.

Table 5.2.5 Three way ANOVA (year: fixed factor; month and site: random factors) of number of species of fish (N.tow$^{-1}$) at four *Heterozostera tasmanica* sites in Norfolk bay between February 1995 and December 1996.

| Factor | DF | MS | F | Prob. |
|---|---|---|---|---|
| Year, *A* | 1 | 0.34 | 0.04 | 0.855 |
| Month, *B* | 5 | 11.71 | 2.74 | 0.059 |
| Site, *C* | 3 | 15.08 | 3.53 | 0.041 |
| Year*Month, *AB* | 5 | 10.94 | 2.76 | 0.058 |
| Year*Site, *AC* | 3 | 1.01 | 0.25 | 0.860 |
| Month*Site, *BC* | 15 | 4.27 | 1.06 | 0.407 |
| Year*Month*Site, *ABC* | 15 | 3.96 | 0.98 | 0.482 |
| Error, *E* | 96 | 4.04 | | |

Table 5.2.6 Ryans Q-test for number of species of fish (N.tow$^{-1}$) between sites from ANOVA presented in Table 5.2.5. Bold underlining indicates no significant difference.

| **Site** | Sommers Bay | Lime Bay | Prices Bay | Smooth Is. |
|---|---|---|---|---|
| Mean number of species | 7.3 | 7.2 | 6.4 | 5.9 |

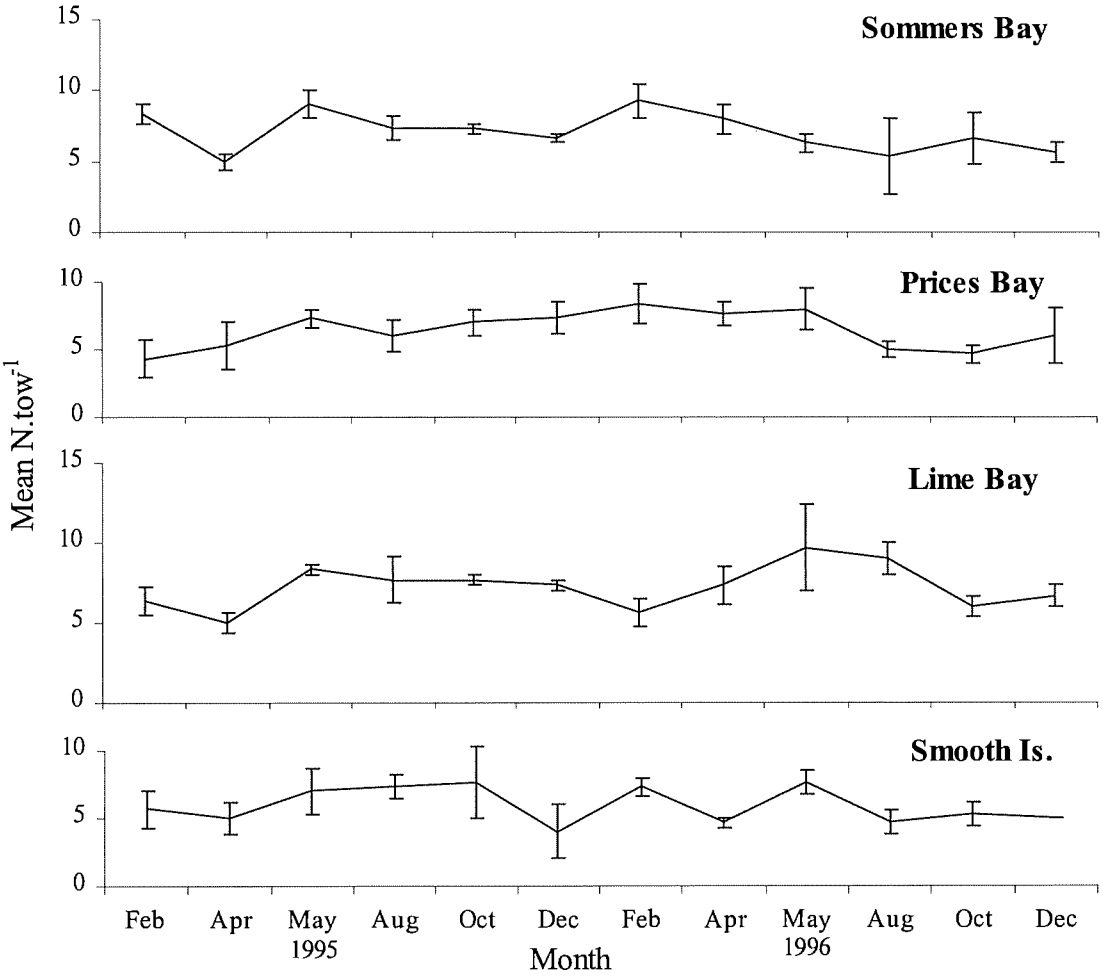



Fig. 5.2.6 Mean number of species of fish ($N.tow^{-1}$) at four *Heterozostera tasmanica* sites in Norfolk Bay between February 1995 and December 1996. Error bars are standard error.

### *5.2.3.4 Recruitment*

Size compositions for the three dominant species (*A. spilomelanurus*, *N. balteatus* and *S. argus)* indicate that the trends in abundance is not a reflection of temporal patterns in recruitment. Recruitment of *A. spilomelanurus* occurred in February of both years at around 1.5 cm with clear modal progression apparent throughout the year (Fig. 5.2.7). Similar temporal patterns of recruitment are evident for *N. balteatus* with new recruits at around 1.5-2.0 cm first present in February of both years (Fig. 5.2.8). There is clear progression of this 0+ cohort throughout the year.

Distinct modes are less evident for *S. argus,* with the smallest individuals present in August 1995 and May 1996 (Fig. 5.2.9). However, the low level of recruitment observed contributes very little to the considerable changes in abundance of this species.

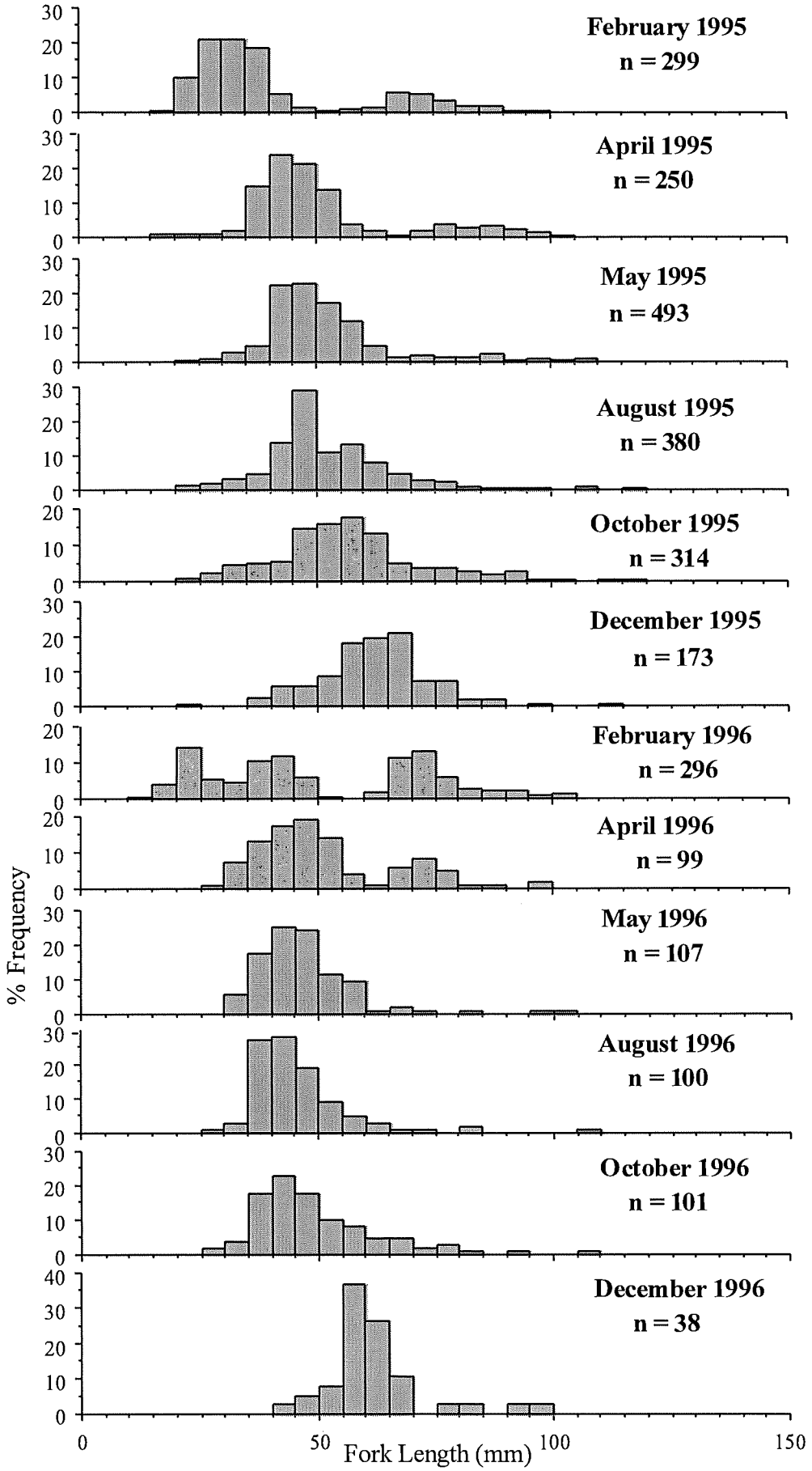



Fig. 5.2.7 Length-frequency distributions of *Acanthaluteres spilomelanurus* sampled from *Heterozostera* sites in Norfolk Bay between February 1995 and December 1996. n is sample size.

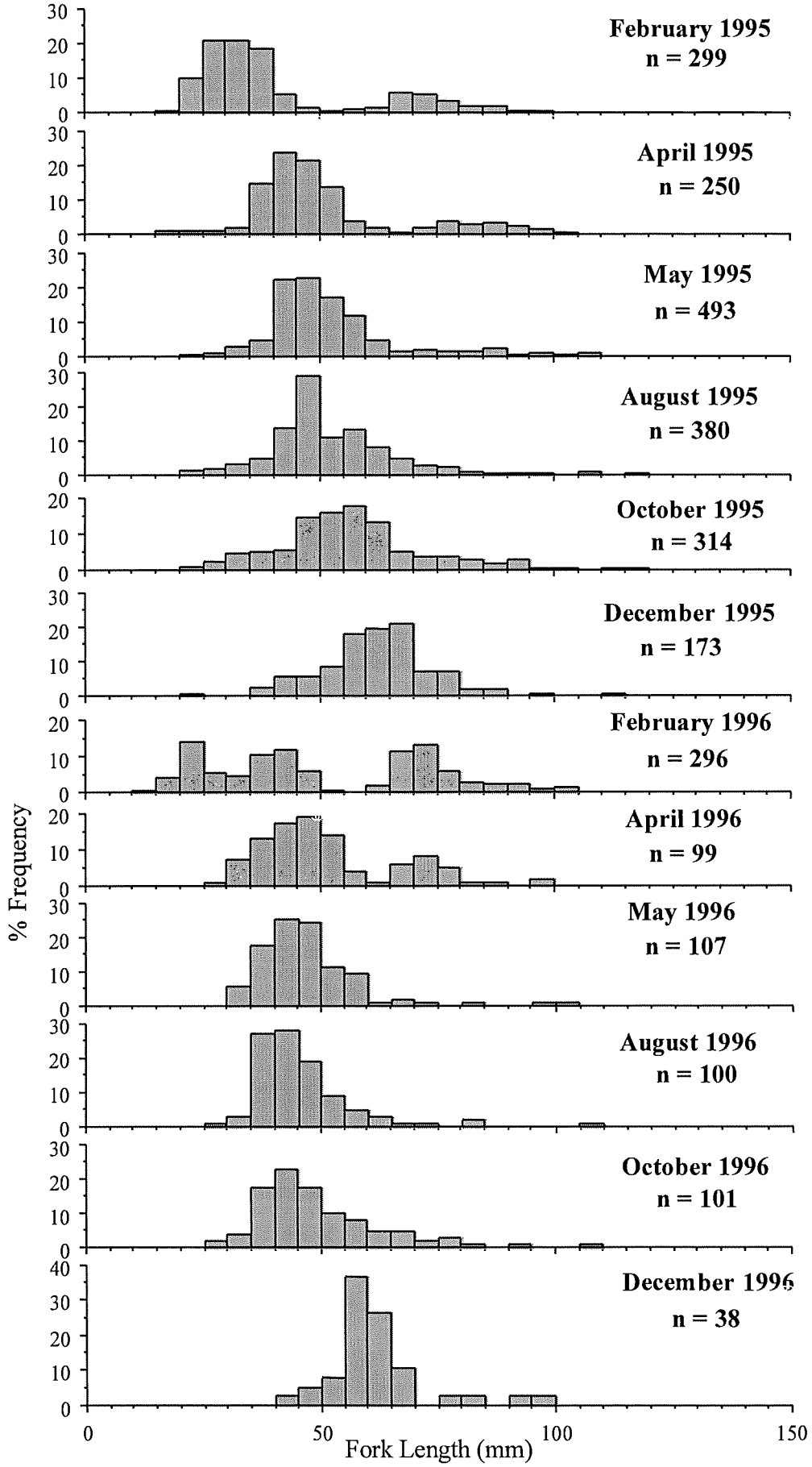



Fig. 5.2.8 Length-frequency distributions of *Neoodax balteatus* sampled from *Heterozostera tasmanica* sites in Norfolk Bay between February 1995 and December 1996. n is sample size.

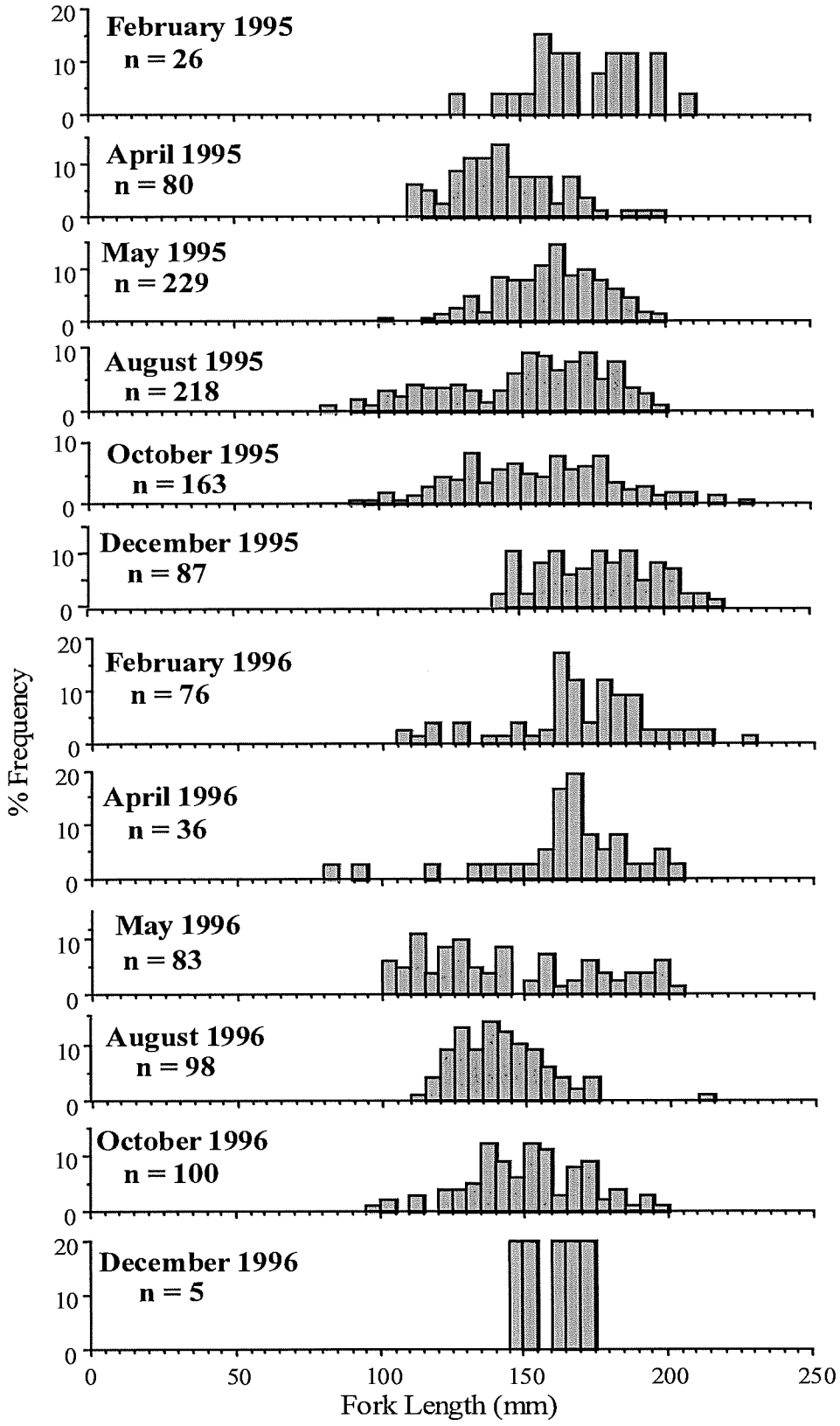



Fig. 5.2.9 Length-frequency distributions of *Stigmatopora argus* sampled from *Heterozostera tasmanica* sites in Norfolk Bay between February 1995 and December 1996. n is sample size.

*5.2.3.5 Community composition*

Two dimensional ordination plots for the two years of the study show varying degrees of separation between sites (Fig. 5.2.10A,B). For example, a clear separation between samples from Sommers Bay and Smooth Island were present in 1996 but considerable overlap is seen in 1995. Conversely, considerable overlap occurred between samples from Lime Bay and Sommers Bay in both years.

A two-way crossed ANOSIM detected both interannual and site differences (Table 5.2.7). The comparison between Sommers Bay and Lime Bay was non-significant ($P$=0.131), while the Sommers Bay/Smooth Island comparison bordered on significance ($P$=0.048); other site comparisons were significant. Differences in numbers of *A. spilomelanurus* contributed most to the separation of sites; this species ranked as the primary discriminator in three of four comparisons identified as significantly different (Table 5.2.8). *Stigmatopora argus* was the primary discriminator for the remaining comparison.

Separate ANOSIMs for 1995 and 1996 revealed a level of interannual variability not detected by the two-way analysis (Table 5.2.7). Greatest variability was seen between Lime Bay and Smooth Island. In 1995, there was no significant differences between fish communities from these sites ($P$=0.398), but differences were significant in 1996 ($P$=0.019). Differences between the four species (*A. spilomelanurus*, *S. argus, N. balteatus* and *G. marmoratus*) contributing most to the variation between these sites were greater in 1996 than in 1995 (Fig. 5.2.11).

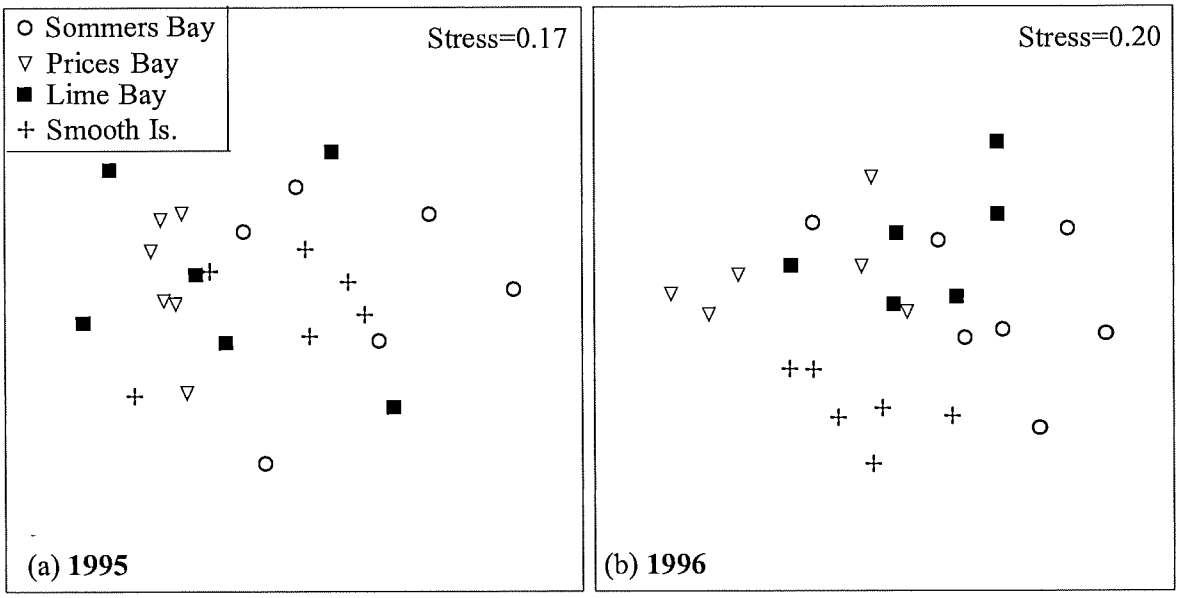



Fig. 5.2.10 Non-metric multi-dimensional scaling of fish communities sampled by beam trawl at *Heterozostera tasmanica* sites in Norfolk Bay between February 1995 and December 1996.

Table 5.2.7 ANOSIM probabilities for two-way (year × site) and one-way analysis (years).

| **Comparison** | **two-way** | **1995** | **1996** |
|---|---|---|---|
| **1995, 1996** | 0.009 | - | - |
| Sommers Bay, Prices Bay | <0.001 | 0.017 | 0.009 |
| Sommers Bay, Lime Bay | 0.131 | 0.147 | 0.275 |
| Sommers Bay, Smooth Is. | 0.001 | 0.024 | 0.004 |
| Prices Bay, Lime Bay | 0.006 | 0.015 | 0.074 |
| Prices Bay, Smooth Is. | 0.034 | 0.050 | 0.182 |
| Lime Bay, Smooth Island | 0.048 | 0.398 | 0.019 |

Table 5.2.8 Average dissimilarity between sites and ranked order of contribution (SIMPER) of species to site separations for comparisons identified as significant by ANOSIM.

| **Comparison** | SB/PB | PB/LB | PB/SI | LB/SI |
|---|---|---|---|---|
| **Average dissimilarity** | 36.1% | 36.4% | 36.8% | 37.7% |
| *Acanthaluteres spilomelanurus* | 1 | 1 | 2 | 1 |
| *Stigmatopora argus* | 2 | 3 | 1 | 2 |
| *Neoodax balteatus* | 4 | 2 | 3 | 6 |
| *Vincentia conspersa* | 9 | 4 | 4 | 4 |
| *Platycephalus bassensis* | 3 | 11 | 13 | 9 |
| *Meuschenia freycineti* | 8 | 7 | 5 | 14 |
| *Gymnapistes marmoratus* | 13 | 6 | 9 | 3 |
| *Vanocampus philipi* | 5 | 14 | 7 | 11 |
| *Cristiceps australis* | 10 | 8 | 12 | 13 |
| *Acanthaluteres vitteger* | 7 | 12 | 14 | 8 |

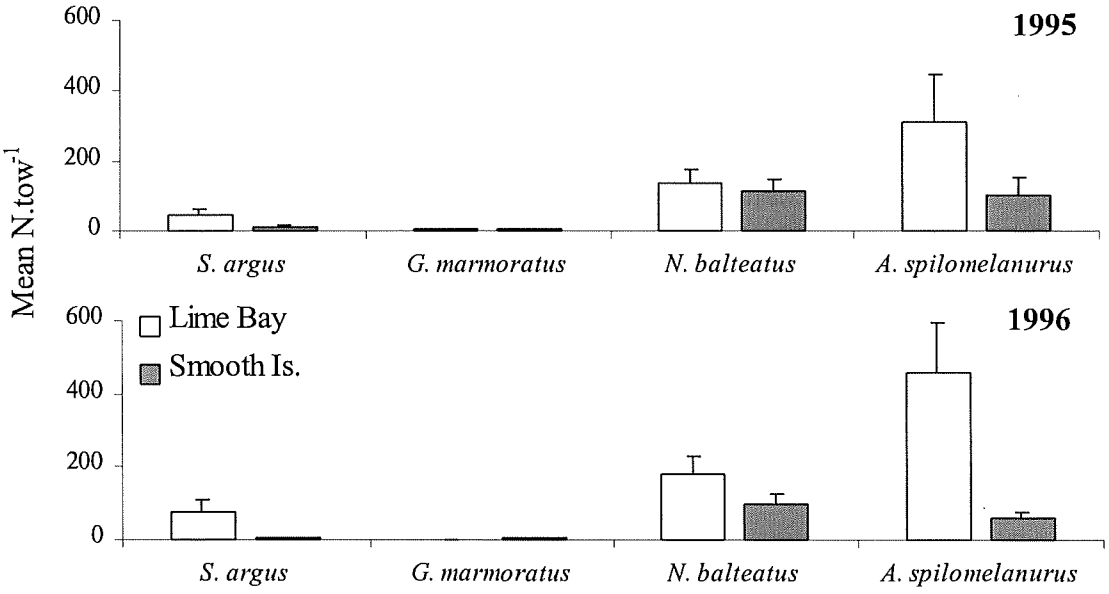



Fig. 5.2.11 Mean abundance (N.tow$^{-1}$) of *Stigmatopora argus, Gymnapistes marmoratus, Neoodax balteatus* and *Acanthaluteres spilomelanurus* at two *Heterozostera tasmanica* sites in Norfolk Bay in 1995 and 1996. Error bars are standard error.

No consistent patterns of seasonal variability at sites within years was observed. Sommers Bay was the only site to differ significantly between years (Table 5.2.9). The difference is driven primarily by the consistently higher abundances of juvenile *Pseudophycis bachus* in 1995 compared to 1996 (Fig. 5.2.12).

Table 5.2.9 Significance of interannual variability within sites from one-way ANOSIM.

| Site | Probability |
|---|---|
| Sommers Bay | 0.011 |
| Prices Bay | 0.074 |
| Lime Bay | 0.238 |
| Smooth Is. | 0.366 |

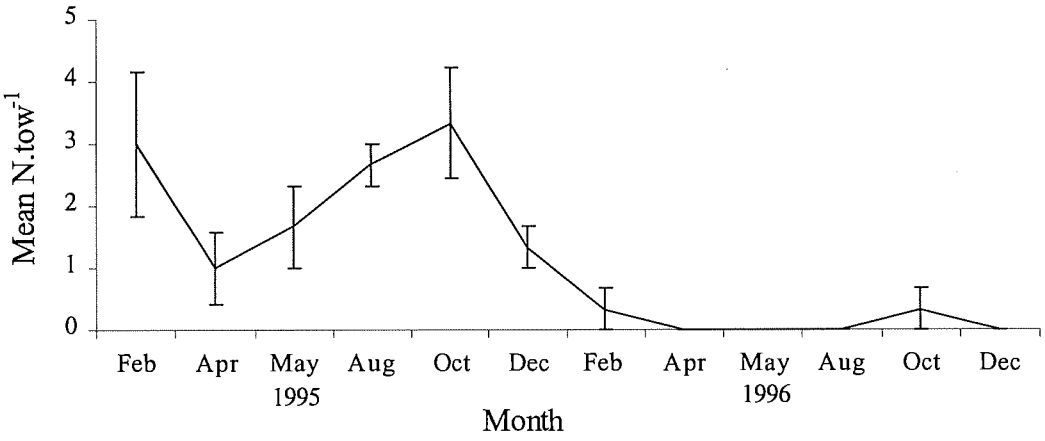



Fig. 5.2.12 Mean abundance (N.tow$^{-1}$) of *Pseudophycis bachus* caught at Sommers Bay between February 1995 and December 1996. Error bars are standard error.

## 5.2.4 Discussion

The present study revealed consistent differences in total fish abundance between all *Heterozostera* sites in Norfolk Bay. Differences in fish abundance and community composition in similar habitats within an estuary or embayment have been related to temperature and salinity (Loneragan *et al.* 1986) and turbidity (Blaber 1980). As all sites in Norfolk Bay have the same physical characteristics, differences in abundance and community composition are most likely to be related to differences in habitat structure between sites and/or patchiness in larval supply.

There were consistent differences in the structure of the *Heterozostera* beds between sites, with seagrass density consistently highest at Sommers Bay and lowest at Smooth Island. Similar patterns were present in fish abundance which were also highest at Sommers Bay and lowest at Smooth Island. Higher abundances at sites with higher seagrass density have been related to increased food availability and increases in available shelter allowing greater protection from predators (Heck & Orth 1980, Edgar & Shaw 1995b). However, the response to changes in seagrass standing stock appears to be species specific. Bell and

Westoby (1986a) found that thinning and shortening seagrass blades resulted in either increased or decreased abundance, or no change, depending on the species. Mills (1992) found similar results in artificial *Zostera/Heterozostera* beds of varying density in southern Tasmania. It is apparent that, rather than a consistent mechanism applying across all species, there are species specific behavioural responses to varying seagrass density. The between site trends in abundance in the present study were primarily driven by *A. spilomelanurus*, suggesting that the abundance of this species is strongly influenced by seagrass density. In addition, six of thirty-two species overall, show a distinct preference for the *Heterozostera* bed in Sommers Bay. Abundance of the second most abundant species (*Neoodax balteatus*) varied little between sites. While no individual species showed a distinct preference for the patchy *Heterozostera* bed at Smooth Island, *Gymnapistes marmoratus* was highly abundant at this site. This is consistent with Mills (1992) who found that this species shows a distinct preference for patchy seagrass.

Temporal variability in abundance was detected at all sites, and similar trends occurred across both years of the study. A common pattern of seasonal variability in previous studies has been for abundance to peak in summer/spring, and decrease to a minimum in winter (Adams 1976, Edgar and Shaw 1995b). Fish biomass has often been found to closely parallel seagrass biomass (Adams 1976, Heck & Orth 1980, Edgar & Shaw 1995b), which generally decreases as water temperature and/or light intensity decrease in winter (Walker and McComb 1988). Edgar and Shaw (1995b) found an increase in seagrass biomass and invertebrate production in spring was followed by recruitment of fish in summer. Benthic production decreased in autumn resulting in a decrease in fish abundance through mortality and/or emigration.

The late autumn/winter peaks in fish abundance recorded in the present study cannot be directly attributed to either temporal patterns of seagrass biomass or recruitment for the dominant species. Studies of seasonal changes in temperate Australian seagrass beds, including *Heterozostera tasmanica* (Bulthuis and Woelkerling 1983, Mills 1992), invariably show a unimodal seasonal variation in standing crop, with peak values in summer to early autumn, and minima during winter (Kirkman and Reid 1979, Larkum *et al.* 1984). This pattern was consistent in *Heterozostera* beds in Norfolk Bay in the present study. Sites for this study represented beds of highest seagrass density and cannot therefore be considered representative of all *Heterozostera* beds in Norfolk Bay. As winter dieback of seagrass begins, the deep fringes of beds and patchy deeper beds, where light first becomes limiting, die back first. Given the marginal existence of seagrass in these areas, it is likely that the degree of dieback will be greater than in denser, well established beds. Fish may leave these areas and select remaining areas of dense seagrass. As dieback progresses available habitat for seagrass-dependant species decreases and fish are concentrated into the small remaining areas of dense seagrass.

Length frequency analysis of abundant fish species clearly illustrate that recruitment is not responsible for the winter peak in abundance. Where recruitment peaks were detectable, they occurred in summer. A possible exception is *Stigmatopora argus*, with apparent recruitment peaks in May to August. However, the level of recruitment is not sufficient to explain the increase in abundance of this species. It is also clear that temporal variations in abundance are not strongly driven by seasonal recruitment of transient species utilising the beds as a temporary nursery area. This is in contrast to seagrass beds in NSW where juveniles of economically important species recruit to such habitats primarily during summer resulting in abundance peaking in that season (Middleton *et al.* 1984).

The mechanism leading to winter peaks in fish abundance must be operating on spatial scales of individual seagrass beds. It is unlikely that species such as the syngnathids, with limited mobility and exhibiting a very strong habitat association, are migrating over a long distance. The mechanism we propose involves the concentration of fish, by active habitat selection, into decreasing areas of favourable habitat.

This concentration effect must have implications for competition. Given that a large proportion of epifaunal crustacean production within seagrass beds is consumed by fish (Edgar and Shaw 1995a), food availability may become limiting as fish abundance increases. It is possible, however, that the mechanism resulting in high concentrations of fish is also acting on prey items such as mobile macrocrustaceans. If the above mechanism is correct, it means that ultimately the need for association with seagrass is stronger than the disadvantages conveyed by the increase in competition associated with increased fish numbers.

The observed seasonal differences do not appear to be consistent in all Tasmanian seagrass beds (see chapter 5.1), and may be unique to the Norfolk Bay area. Further studies are necessary to determine if the mechanism proposed above is correct. This would require broadening the sampling regime to include areas of deep, sparse seagrass, and a detailed study of temporal trends in seagrass biomass.

Local changes in environmental conditions within Norfolk Bay do appear to have an impact on fish community composition in seagrass beds. However, the differences detected were primarily due to changes in the abundance of species common to all sites rather than changes in the species composition. This occurred despite heavy transformation in the analysis to reduce the weighting of highly abundant species. Most species were caught at all sites, with one species found to be unique to a single site, three species present at two sites, and six species present at three sites. As all species not occurring at all four sites were rare, more intense sampling program may have picked some or all of these species up at further sites.

Sommers Bay was the only site to show significant within-site interannual variability in community composition, and this was due to poor recruitment of a single species

(*Pseudophycis bachus*) in 1996. Interannual differences in community composition may relate to physical changes at sites, such as interannual variability in seagrass biomass, or changes in hydrography leading to different patterns of larval recruitment, or fish movement. This results highlights the limitations of sampling only a small number of sites in order to assess the extent of natural fluctuations in assemblages and populations.

The results of the patterns of abundance and distribution of seagrass associated fish species in Norfolk Bay has important implications for the management of such habitats. It is clear that minimising the impact on marginal seagrass beds will provide considerably less protection to seagrass associated fish populations than targeting areas of highest seagrass density. The temporal trends in fish abundance also indicates it is important to minimise impacts on seagrass beds throughout the year as the seagrass that remain present throughout the colder months are particularly important for sustaining the resident populations.

## 5.3 Comparison of *Heterozostera* and unvegetated habitats

### 5.3.1 Introduction

Within soft-sediment habitats, seagrass beds are widely recognised as an important nursery area for many species of commercial and recreational importance by providing protection and increased food resources compared to bare substrates (see Bell and Pollard 1989). However, unvegetated habitats are becoming increasing recognised as an important habitat for juvenile fishes (Ferrell and Bell 1991, Jenkins *et al.* 1997, Ayvazian and Hyndes 1995, Edgar and Shaw 1995a, Hyndes *et al.* 1996), particularly for species that are protected by either camouflage or schooling behaviour. While levels of food production are higher in seagrass beds (Edgar 1990, Edgar *et al.* 1994), enhanced food production in shallow unvegetated habitats can occur due to the presence of detached macrophytes (Robertson and Lenanton 1984, May and Jenkins 1992) and regular phytoplankton blooms (McLachlan *et al.* 1981).

The greatest variability in soft-bottom fish communities are likely to be seen in a comparison of vegetated and unvegetated habitats. Previous studies have suggested seagrass beds contain a greater abundance and number of fish species than adjacent unvegetated areas (Connolly 1994, Gray *et al.* 1996). However, this difference has been found to be influenced by the distance between unvegetated and seagrass habitats (Ferrell and Bell 1991), relative depth between habitats (Jenkins *et al.* 1997) and size-class of fish being sampled (Edgar and Shaw 1995b).

The aim of this chapter is to examine the relative importance of subtidal *Heterozostera tasmanica* and unvegetated habitats for fish throughout southern and eastern Tasmania. This is done by comparing the abundance, number of species and community composition of both the demersal and larger mobile fish fauna between the two habitats in three areas along the east coast. The life-history stages and residency times of species utilising both habitats is examined and the relative significance of both habitats as a nursery area for economically important fish species described.

### 5.3.2 Methods

The demersal and larger mobile fish fauna was sampled at three areas around the coast of Tasmania (Norfolk Bay, Prosser Bay and Georges Bay) quarterly from summer 1995 to summer 1996. Full details of sampling gear and regime is presented in Chapter 4. In brief, at each area, two sites representing both unvegetated and *Heterozostera tasmanica* habitats were sampled with a beam trawl and multi-panel gillnets (Table 5.3.1). Beam trawl catch rates were calculated as the number of fish per tow while gillnet catch rates were calculated as the number of fish per hour.

Table 5.3.1 Site and habitat characteristics of unvegetated and *Heterozostera tasmanica* sites sampled in Norfolk Bay, Georges Bay and Prosser Bay.

| Area/Site | Habitat | Seagrass Density | Depth (m) |
|---|---|---|---|
| **Norfolk Bay** | | | |
| Lime Bay | *H. tasmanica* | Medium | 3 - 6 |
| Cascade Bay | Mud | | 8 - 12 |
| **Georges Bay** | | | |
| Steiglitz Beach | *H. tasmanica* | High | 2 - 5 |
| McDonalds Pt. | Mud | | 8 - 12 |
| **Prosser Bay** | | | |
| Paddys Point | *H. tasmanica* | Low | 3 - 5 |
| Raspins Beach | Sand | | 2 - 4 |

Variation in fish abundance and number of species per tow across habitats, areas and time of sampling was assessed by three-way ANOVA. Area was considered to be a random variable; areas were chosen from many possible sites around Tasmania, and we were not investigating hypotheses relating to particular sites. Similarly, although the term 'season' has been used as a convenient label for time of sampling, this variable was considered random; samples were taken quarterly because of available resources, not because we were specifically testing for differences between seasons. Habitat was considered a fixed factor.

The resulting ANOVA model (two random factors, one fixed factor) provides no appropriate test of main effects of the fixed factor (habitat), as no denominator is available for calculation of the variance ratio which includes all appropriate error terms. Main effects for area were therefore estimated using a *quasi-F* calculation (Winer 1971), developed according to the protocol presented by Zar (1996).

Differences in fish community structure between the two habitats were analysed by non-metric multi-dimensional scaling (MDS) and analysis of similarities (ANOSIM). For a justification of the choice of these techniques, and a description of their execution, see Chapter 4. Analyses were applied to $x^{0.25}$ transformed abundance data. All species represented by a single individual were excluded from analysis. Replicates taken on each sampling date were pooled to simplify interpretation of the MDS plots.

Significance of differences between fish communities in the two habitat types and across areas was tested by two-way analysis of similarities (ANOSIM) applied to a ranked matrix of Bray-Curtis similarities. In order to detect possible significant interactions between habitat and area, a one-way ANOSIM including each habitat sample as a unique site (ie two habitats × three areas = six sites) was performed. Species contributions to the average dissimilarity between groups were also computed. Seasonal differences were assessed by observing patterns of placement of points on the MDS plots.

### 5.3.3 Results

#### *5.3.3.1 Fish composition*

A total of 65 species were caught, with 21 species unique to *Heterozostera* habitats, 18 species unique to unvegetated habitats and 26 species common to both habitats. Species unique to either habitat tended to be rare; 14 species unique to unvegetated habitats, and 12 species unique to *Heterozostera* habitats were caught only once. Communities sampled by the two gear types were quite distinct, with only 10 species being caught by beam trawl and gillnets.

#### *5.3.3.2 Demersal fish composition*

Ninety beam trawl tows yielded 4,531 fish (44 species, 21 families), 4,116 from *Heterozostera* sites and 415 from unvegetated sites (Table 5.3.2). A full list of scientific and common names is presented in Appendix 1. Sixteen species were represented by a single specimen. *Heterozostera* sites were dominated by *Acanthaluteres spilomelanurus*, *Neoodax balteatus* and *Stigmatopora argus*, comprising 88% of the catch. *Platycephalus bassensis*, *Neoodax balteatus, Nesogobius* sp.1 and *Rhombosolea tapirina* were dominant at unvegetated sites, making up 60% of the catch. The most abundant of 14 species unique to *Heterozostera* and 8 species unique to unvegetated sites were *Acanthaluteres vittiger* (11 individuals) and *Tetreactenos glaber* (4 individuals) respectively.

Table 5.3.2 Total number of individuals and percentage of the total individuals for the fish taxa collected by beam trawl on *Heterozostera tasmanica* and unvegetated sites at Georges Bay, Norfolk Bay and Prosser Bay between January 1995 and 1996. ● Small demersal resident species caught regularly as juveniles and/or adults in beam trawl samples. ■ Larger resident species occurring in beam trawl samples as juveniles and gillnet samples as adults. ◆ Species occurring in beam trawl samples as juveniles, but not seen as adults in gillnet samples.

| Species | Georges Bay | | | | Norfolk Bay | | | | Prosser Bay | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | *H. tasmanica* | | *mud* | | *H. tasmanica* | | *mud* | | *H. tasmanica* | | *mud* | |
| | n | % | n | % | n | % | n | % | n | % | n | % |
| **Urolophidae** | | | | | | | | | | | | |
| *Urolophus cruciatus* | ■ 1 | 0.08 | 3 | 2.5 | 0 | 0 | 0 | 0 | 9 | 5.4 | 2 | 1.6 |
| *Urolophus paucimaculatus* | ● 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 1.8 | 1 | 0.8 |
| **Moridae** | | | | | | | | | | | | |
| *Pseudophycis bachus* | ■ 1 | 0.08 | 0 | 0 | 14 | 0.5 | 3 | 2.1 | 5 | 3.0 | 2 | 1.6 |
| **Syngnathidae** | | | | | | | | | | | | |
| *Hippocampus abdominalis* | ● 1 | 0.08 | 0 | 0 | 1 | 0.04 | 0 | 0 | 1 | 0.6 | 0 | 0 |
| *Mitotichthys semistriatus* | ● 0 | 0 | 0 | 0 | 6 | 0.2 | 0 | 0 | 2 | 1.2 | 1 | 0.8 |
| *Stigmatopora argus* | ● 700 | 57.7 | 0 | 0 | 231 | 8.4 | 0 | 0 | 24 | 14.4 | 5 | 3.9 |
| *Stigmatopora nigra* | ● 5 | 0.41 | 0 | 0 | 0 | 0 | 1 | 0.7 | 8 | 4.8 | 6 | 4.7 |
| *Vanacampus phillipi* | ● 45 | 3.7 | 0 | 0 | 14 | 0.5 | 3 | 2.1 | 5 | 3.0 | 0 | 0 |
| **Scorpaenidae** | | | | | | | | | | | | |
| *Helicolenus barathri* | ◆ 0 | 0 | 2 | 1.7 | 2 | 0.1 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Gymnapistes marmoratus* | ● 58 | 4.8 | 6 | 5.0 | 20 | 0.7 | 3 | 2.1 | 7 | 4.2 | 0 | 0 |
| **Triglidae** | | | | | | | | | | | | |
| *Lepidotrigla papilio* | ◆ 0 | 0 | 6 | 5.0 | 0 | 0 | 0 | 0 | 1 | 0.6 | 2 | 1.6 |
| **Platycephalidae** | | | | | | | | | | | | |
| *Platycephalus bassensis* | ■ 2 | 0.2 | 19 | 15.7 | 2 | 0.07 | 33 | 23.6 | 11 | 6.6 | 3 | 2.4 |
| **Apogonidae** | | | | | | | | | | | | |
| *Vincentia conspersa* | ● 64 | 5.3 | 2 | 1.7 | 33 | 1.2 | 0 | 0 | 6 | 3.6 | 0 | 0 |
| **Carangidae** | | | | | | | | | | | | |
| *Pseudocaranx dentex* | ● 0 | 0 | 2 | 1.7 | 0 | 0 | 0 | 0 | 2 | 1.2 | 2 | 1.6 |
| **Odacidae** | | | | | | | | | | | | |
| *Neoodax balteatus* | ● 244 | 20.1 | 1 | 0.8 | 650 | 24.0 | 42 | 30.0 | 22 | 13.2 | 24 | 18.9 |
| **Clinidae** | | | | | | | | | | | | |
| *Cristaceps australis* | ● 12 | 1.0 | 0 | 0 | 39 | 1.4 | 2 | 1.4 | 16 | 9.6 | 26 | 20.5 |
| *Heteroclinus perspicillatus* | ● 6 | 0.5 | 0 | 0 | 6 | 0.2 | 0 | 0 | 2 | 1.2 | 0 | 0 |
| **Gobiidae** | | | | | | | | | | | | |
| *Nesogobius hinsbyi* | ● 0 | 0 | 1 | 0.8 | 0 | 0 | 12 | 7.1 | 2 | 0.9 | 0 | 0 |
| *Nesogobius pulchellus* | ● 7 | 0.6 | 0 | 0 | 1 | 0.04 | 0 | 0 | 3 | 1.8 | 4 | 3.1 |
| *Pseudogobius olorum* | ● 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.9 | 0 | 0 |
| *Nesogobius* sp.1 | ● 4 | 0.3 | 24 | 19.8 | 5 | 0.2 | 38 | 27.1 | 23 | 13.8 | 9 | 7.1 |
| *Nesogobius* sp.3 | ● 1 | 0.08 | 1 | 0.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Pleuronectidae** | | | | | | | | | | | | |
| *Ammotretis rostratus* | ■ 0 | 0 | 5 | 4.1 | 0 | 0 | 0 | 0 | 0 | 0 | 10 | 7.9 |
| *Rhombosolea tapirina* | ■ 0 | 0 | 49 | 40.5 | 0 | 0 | 3 | 2.1 | 1 | 0.6 | 0 | 0 |
| *Taratretis derwentensis* | ● 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 1.6 |
| **Monocanthidae** | | | | | | | | | | | | |
| *Acanthaluteres vittiger* | ■ 2 | 0.2 | 0 | 0 | 8 | 0.3 | 0 | 0 | 1 | 0.6 | 0 | 0 |
| *A. spilomelanurus* | ● 43 | 3.5 | 0 | 0 | 1703 | 62 | 10 | 7.1 | 12 | 7.2 | 21 | 16.5 |
| *Brachaluteres jacksonianus* | ● 7 | 0.6 | 0 | 0 | 1 | 0.04 | 0 | 0 | 0 | 0 | 3 | 2.4 |
| *Meuschenia freycineti* | ■ 11 | 0.91 | 0 | 0 | 1 | 0.04 | 2 | 1.4 | 1 | 0.6 | 0 | 0 |
| **Tetraodontidae** | | | | | | | | | | | | |
| *Tetractenos glaber* | ■ 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 3.2 |

Consistently more fish were caught in *Heterozostera* than unvegetated sites in Norfolk Bay and Georges Bay, while the relationship does not hold for Prosser Bay (Table 5.3.3, Fig. 5.3.1). A significant interaction between habitat and area confirms the different response to habitat type in the different areas. There was no significant differences in numbers of fish between seasons. The habitat/season interaction is close to being significant (p=0.065), reflecting the greater seasonal variability in *Heterozostera* sites in Norfolk Bay and Georges Bay.

Table 5.3.3 Three way ANOVA of log transformed abundance of fish (N.tow$^{-1}$) caught by beam trawl. Habitat, fixed factor; area and season, random factors. DF, degrees of freedom; MS, mean square.

| Factor | *Hypothesis* | DF | MS | F | Prob. |
|---|---|---|---|---|---|
| Habitat, *A* | *quasi-F* | 1 | 3.80 | 5.1 | 0.109 |
| area, *B* | *B/BC* | 2 | 0.90 | 14.7 | 0.002 |
| season, *C* | *C/BC* | 4 | 0.03 | 0.53 | 0.720 |
| habitat*area, *AB* | *AB/ABC* | 2 | 0.56 | 8.46 | 0.014 |
| habitat*season, *AC* | *AC/ABC* | 4 | 0.25 | 3.43 | 0.065 |
| area*season, *BC* | *BC/E* | 8 | 0.06 | 0.92 | 0.510 |
| habitat*area*season, *ABC* | *ABC/E* | 8 | 0.07 | 1.11 | 0.371 |
| Error, *E* | | 60 | 0.07 | | |

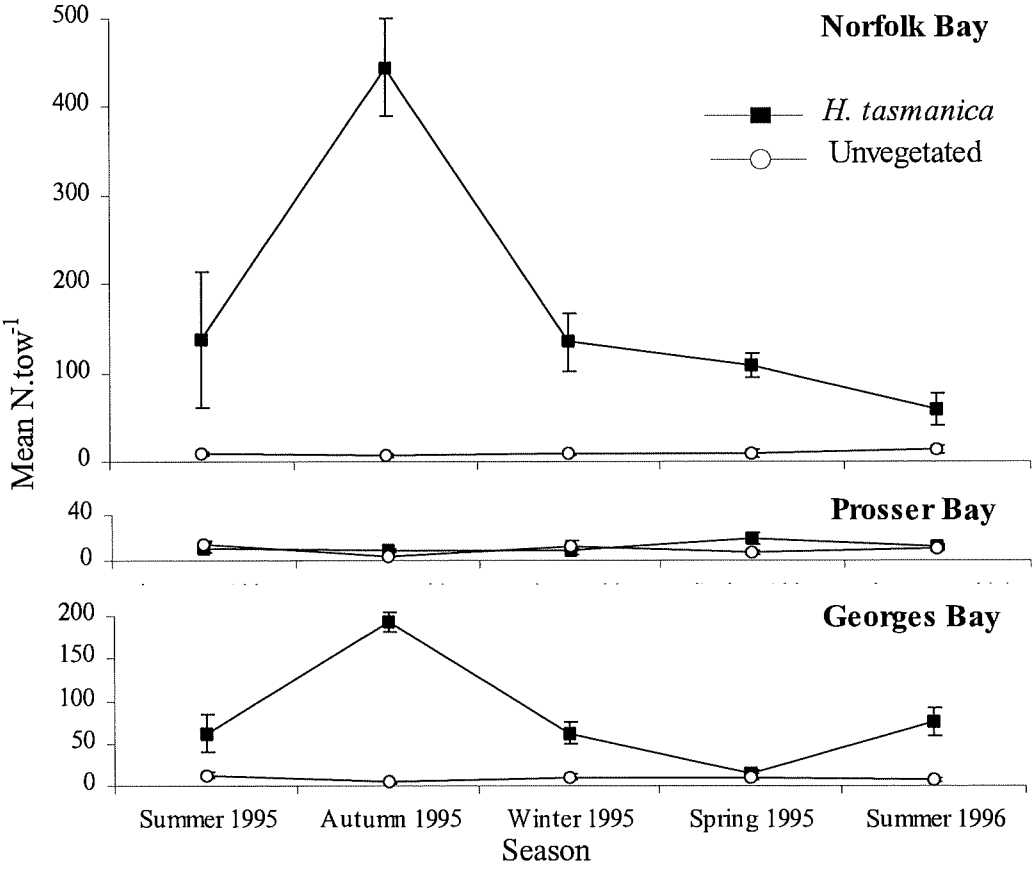



Fig. 5.3.1 Mean seasonal abundance (N.tow$^{-1}$) of fish caught by beam trawl at unvegetated and *Heterozostera tasmanica* sites in Norfolk Bay, Prosser Bay and Georges Bay between summer 1995 and 1996. Error bars are standard error.

Similar relationships can be seen in variability of number of fish species (Table 5.3.4, Fig. 5.3.2). However, actual differences in numbers of species are small, giving rise to few significant differences. The interaction between area and season, and the three-way interaction both border on significance, again due to different patterns at Prosser Bay. It is likely that the interaction between habitat and area is also non-random (p=0.079), largely due to the consistent difference between habitats in Norfolk Bay.

Table 5.3.4 Three way ANOVA of number of species of fish (N.tow$^{-1}$) caught by beam-trawl. Habitat, fixed factor; area and season, random factors. DF, degrees of freedom; MS, mean square.

| Factor | DF | MS | F | Prob. |
|---|---|---|---|---|
| habitat, *A* | 1 | 193.6 | 9.81 | 0.089 |
| area, *B* | 2 | 6.7 | 1.26 | 0.334 |
| season, *C* | 4 | 4.0 | 0.75 | 0.583 |
| habitat*area, *AB* | 2 | 18.2 | 3.54 | 0.079 |
| habitat*season, *AC* | 4 | 6.6 | 1.23 | 0.371 |
| area*season, *BC* | 8 | 5.3 | 2.10 | 0.049 |
| habitat*area*season, *ABC* | 8 | 5.14 | 2.04 | 0.057 |
| Error, *E* | 60 | 2.5 | | |

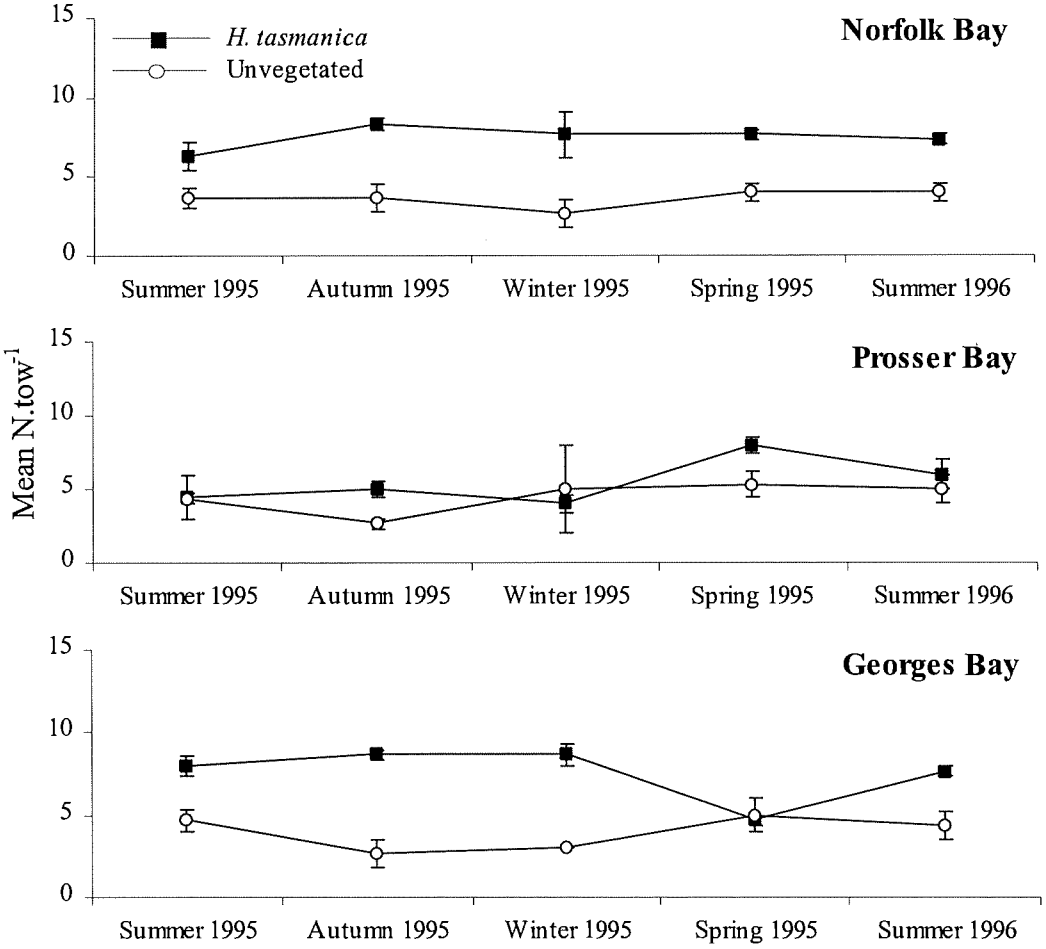



Fig. 5.3.2 Mean number of species (N.tow$^{-1}$) caught by beam trawl at unvegetated and *Heterozostera tasmanica* sites in Norfolk Bay, Prosser Bay and Georges Bay between summer 1995 and 1996. Error bars are standard error.

### *5.3.3.3 Larger mobile fish*

Sixty overnight gillnet sets yielded 696 fish (40 species, 33 families); 417 ( 27 species, 23 families) from *Heterozostera* sites and 277 (32 species, 26 families) from unvegetated sites (Table 5.3.5). Eleven species were represented by a single specimen. The catch was dominated by *Platycephalus bassensis*, *Pseudophycis bachus* and *Aldrichetta forsteri* in both *Heterozostera* (63%) and unvegetated (60%) sites. Of 13 species unique to unvegetated sites, *Mustelus antarcticus* (9 individuals) was most abundant. Eight species were unique to *Heterozostera* sites, and of these *Haletta semifasciata* (16 individuals) was the most abundant.

Table 5.3.5 Total number of individuals and percentage of the total individuals for the fish taxa collected from summer 1995 to summer 1996 using the gillnets on seagrass sites and unvegetated sites at Georges Bay, Norfolk Bay and Prosser Bay. ■ Larger resident species occurring in beam trawl samples as juveniles and gillnet samples as adults. ◆ Species caught as juveniles or adults in gillnets, but not caught as juveniles in beam trawl samples.

| Species | | Georges Bay | | | | Norfolk Bay | | | | Prosser Bay | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | *H. tasmanica* | | *mud* | | *H. tasmanica* | | *mud* | | *H. tasmanica* | | *mud* | |
| | | n | % | n | % | n | % | n | % | n | % | n | % |
| **Scyliorhinidae** | | | | | | | | | | | | | |
| *Cephaloscyllium laticeps* | ◆ | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 1.3 |
| **Triakidae** | | | | | | | | | | | | | |
| *Mustelus antarcticus* | ◆ | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 2.7 | 0 | 0 | 7 | 4.6 |
| *Galeorhinus galeus* | ◆ | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 5.5 | 0 | 0 | 0 | 0 |
| **Squalidae** | | | | | | | | | | | | | |
| *Squalus acanthias* | ◆ | 0 | 0 | 0 | 0 | 1 | 0.7 | 0 | 0 | 0 | 0 | 1 | 0.7 |
| **Pristiophoridae** | | | | | | | | | | | | | |
| *Pristiophorus nudipinnis* | ◆ | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 2.0 |
| **Rajidae** | | | | | | | | | | | | | |
| *Raja lemprier* | ◆ | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1.4 | 1 | 0.8 | 0 | 0 |
| *Raja whitleyi* | ◆ | 0 | 0 | 0 | 0 | 3 | 2.1 | 0 | 0 | 0 | 0 | 1 | 0.7 |
| **Callorhynchidae** | | | | | | | | | | | | | |
| *Callorhynchus milii* | ◆ | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1.4 | 1 | 0.8 | 0 | 0 |
| **Moridae** | | | | | | | | | | | | | |
| *Pseudophycis bachus* | ■ | 4 | 2.7 | 11 | 21 | 34 | 24.3 | 18 | 24.7 | 20 | 15.3 | 49 | 32.2 |
| **Ophydiidae** | | | | | | | | | | | | | |
| *Genypterus tigerinus* | ◆ | 2 | 1.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Scorpaenidae** | | | | | | | | | | | | | |
| *Neosebastes thetidis* | ■ | 1 | 0.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 1.3 |
| **Platycephalidae** | | | | | | | | | | | | | |
| *Platycephalus bassensis* | ■ | 14 | 9.6 | 4 | 7.7 | 81 | 57.9 | 40 | 54.8 | 24 | 18.3 | 24 | 15.8 |
| **Pomatomidae** | | | | | | | | | | | | | |
| *Pomatomus saltatrix* | ◆ | 2 | 1.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Carangidae** | | | | | | | | | | | | | |
| *Trachurus declivis* | ◆ | 11 | 7.5 | 4 | 7.7 | 0 | 0 | 0 | 0 | 12 | 9.2 | 15 | 9.9 |
| *Pseudocaranx dentex* | ■ | 1 | 0.7 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.8 | 1 | 0.7 |
| **Arripidae** | | | | | | | | | | | | | |
| *Arripis trutta* | ◆ | 2 | 1.4 | 2 | 3.8 | 1 | 0.7 | 0 | 0 | 1 | 0.8 | 11 | 7.2 |
| **Gerreidae** | | | | | | | | | | | | | |
| *Parequula melbournensis* | ◆ | 1 | 0.7 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.8 | 2 | 1.3 |
| **Girellidae** | | | | | | | | | | | | | |
| *Girella tricuspidata* | ◆ | 9 | 6.2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

Table 5.3.5 Continued

| Species | | Georges Bay | | | | Norfolk Bay | | | | Prosser Bay | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | *H. tasmanica* | | *mud* | | *H. tasmanica* | | *mud* | | *H. tasmanica* | | *mud* | |
| | | n | % | n | % | n | % | n | % | n | % | n | % |
| **Cheilodactylidae** | | | | | | | | | | | | | |
| *Dactylophora nigricans* | ◆ | 2 | 1.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Latrididae** | | | | | | | | | | | | | |
| *Latridopsis forsteri* | ◆ | 2 | 1.4 | 1 | 1.9 | 0 | 0 | 0 | 0 | 10 | 7.6 | 0 | 0 |
| **Mugilidae** | | | | | | | | | | | | | |
| *Aldrichetta forsteri* | ◆ | 43 | 29.5 | 7 | 13.5 | 3 | 2.1 | 0 | 0 | 39 | 29.8 | 14 | 9.2 |
| **Sphyraenidae** | | | | | | | | | | | | | |
| *Sphyraena novaehollandiae* | ◆ | 2 | 1.4 | 1 | 1.9 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.7 |
| **Odacidae** | | | | | | | | | | | | | |
| *Haletta semifasciata* | ◆ | 15 | 10.3 | 0 | 0 | 1 | 0.7 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Gempylidae** | | | | | | | | | | | | | |
| *Thyrsites atun* | ◆ | 0 | 0 | 0 | 0 | 1 | 0.7 | 0 | 0 | 2 | 1.5 | 2 | 1.3 |
| **Centrolophidae** | | | | | | | | | | | | | |
| *Seriolella brama* | ◆ | 2 | 1.4 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 3.1 | 1 | 0.9 |
| **Pleuronectidae** | | | | | | | | | | | | | |
| *Ammotretis rostratus* | ■ | 0 | 0 | 2 | 3.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Rhombosolea tapirina* | ■ | 10 | 6.8 | 17 | 32.7 | 1 | 0.7 | 1 | 1.4 | 0 | 0 | 1 | 0.7 |
| **Monocanthidae** | | | | | | | | | | | | | |
| *Acanthaluteres vittiger* | ■ | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8 | 6.1 | 7 | 4.6 |
| *Meuschenia freycineti* | ■ | 22 | 15.1 | 1 | 1.9 | 14 | 10 | 3 | 4.1 | 5 | 3.8 | 7 | 4.6 |

A greater number of fish were collected by gillnet from *Heterozostera* sites in all seasons in Norfolk Bay and Georges Bay, while there is no consistent difference in abundance between habitats in Prosser Bay (Table 5.3.6, Fig. 5.3.3). This differing response to habitat across areas is significant (habitat*area interaction; $p=0.017$). Seasonal changes in abundance were significant with fish abundance lowest in autumn and winter, which were significantly lower than all other seasons (Ryans Q test, $p<0.05$).

Table 5.3.6 Three way ANOVA of log transformed abundances ($N.hr^{-1}$) of fish caught by gillnet. Habitat, fixed factor; area and season, random factors. DF, degrees of freedom; MS, mean square.

| Factor | *Hypothesis* | DF | MS | F | Prob |
|---|---|---|---|---|---|
| habitat, *A* | *quasi-F* | 1 | 1.65 | 0.50 | 0.530 |
| area, *B* | *B/BC* | 2 | 0.75 | 1.21 | 0.347 |
| season, *C* | *C/BC* | 4 | 4.15 | 6.74 | 0.011 |
| habitat*area, *AB* | *AB/ABC* | 2 | 2.54 | 7.00 | 0.017 |
| habitat*season, *AC* | *AC/ABC* | 4 | 1.10 | 3.03 | 0.085 |
| area*season, *BC* | *BC/E* | 8 | 0.62 | 2.13 | 0.064 |
| habitat*area*season, *ABC* | *ABC/E* | 8 | 0.36 | 1.25 | 0.304 |
| Error, *E* | | 30 | 0.29 | | |

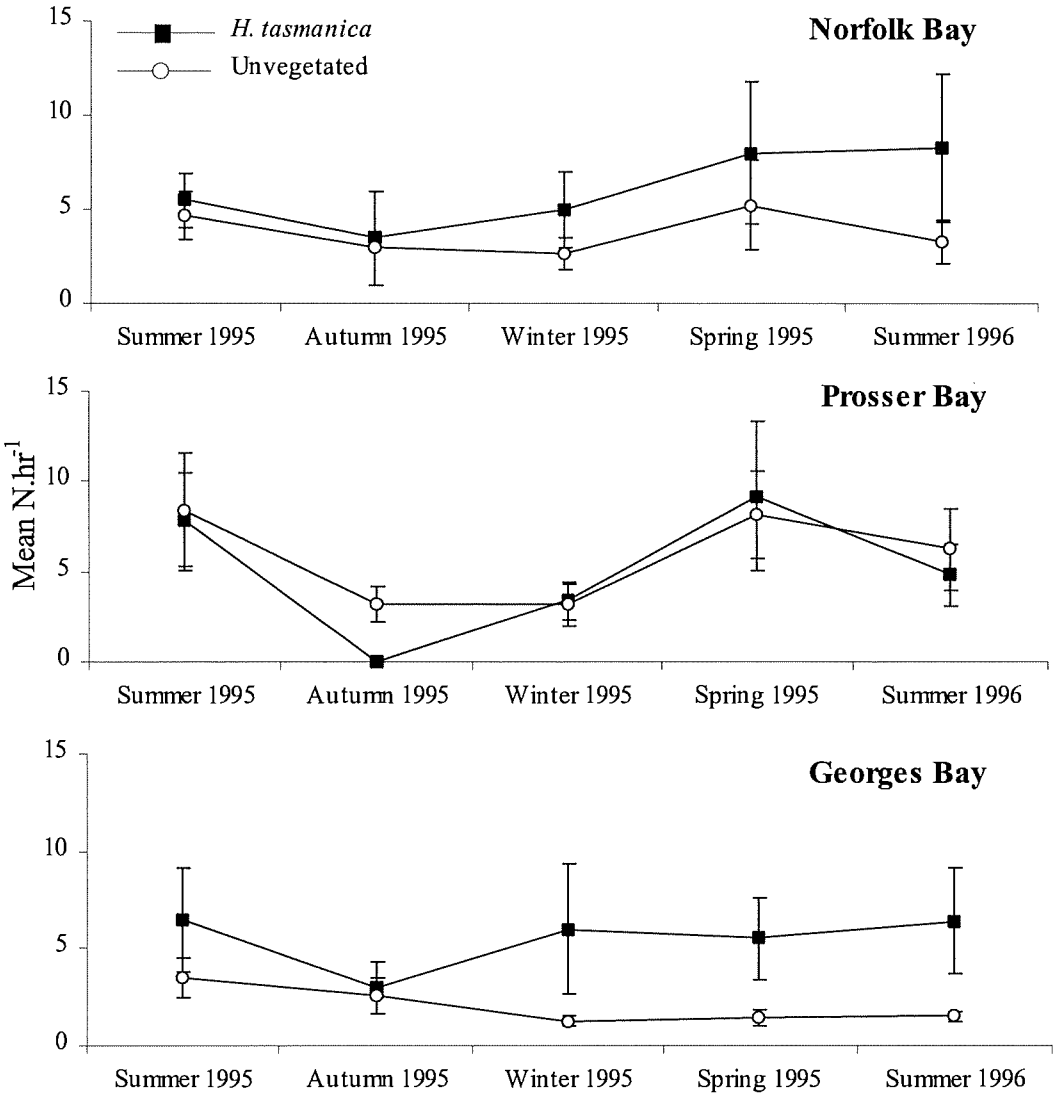



Fig. 5.3.3 Mean seasonal abundance ($N.hr^{-1}$) of fish caught by gillnet at unvegetated and *Heterozostera tasmanica* sites in Norfolk Bay, Prosser Bay and Georges Bay between summer 1995 and 1996. Error bars are standard error.

Differences in number of species collected per gillnet set were significant across areas and seasons (Table 5.3.7, Fig. 5.3.4). Number of species caught at Prosser Bay and Georges Bay was significantly higher than at Norfolk Bay, while samples from summer and spring contained significantly more species than those from winter or autumn (Table 5.3.8).

Table 5.3.7 Three way ANOVA for number of species ($N.hr^{-1}$) caught by gillnet. Habitat, fixed factor; area and season, random factors. DF, degrees of freedom; MS, mean square.

| Factor | DF | MS | F | Prob. |
|---|---|---|---|---|
| habitat, *A* | 1 | 14.0 | 0.99 | 0.502 |
| area, *B* | 2 | 15.0 | 6.92 | 0.018 |
| season, *C* | 4 | 18.1 | 8.37 | 0.006 |
| habitat*area, *AB* | 2 | 10.1 | 2.70 | 0.192 |
| habitat*season, *AC* | 4 | 3.8 | 0.59 | 0.737 |
| area*season, *BC* | 8 | 2.2 | 1.01 | 0.451 |
| habitat*area*season, *ABC* | 8 | 3.7 | 1.74 | 0.131 |
| Error, *E* | 30 | 2.2 | | |

Table 5.3.8 Ryans Q-test for number of species (N.hr$^{-1}$) caught by gillnet between area and season from ANOVA in Table 5.3.7. Bold underlining indicates no significant difference.

| **Area** | Norfolk Bay | Prosser Bay | Georges Bay |
|---|---|---|---|
| Mean | 2.9 | 4.4 | 4.4 |

| **Season** | Autumn '95 | Winter '95 | Summer '96 | Summer '95 | Spring '95 |
|---|---|---|---|---|---|
| Mean | 2.5 | 2.6 | 4.3 | 4.9 | 5.0 |

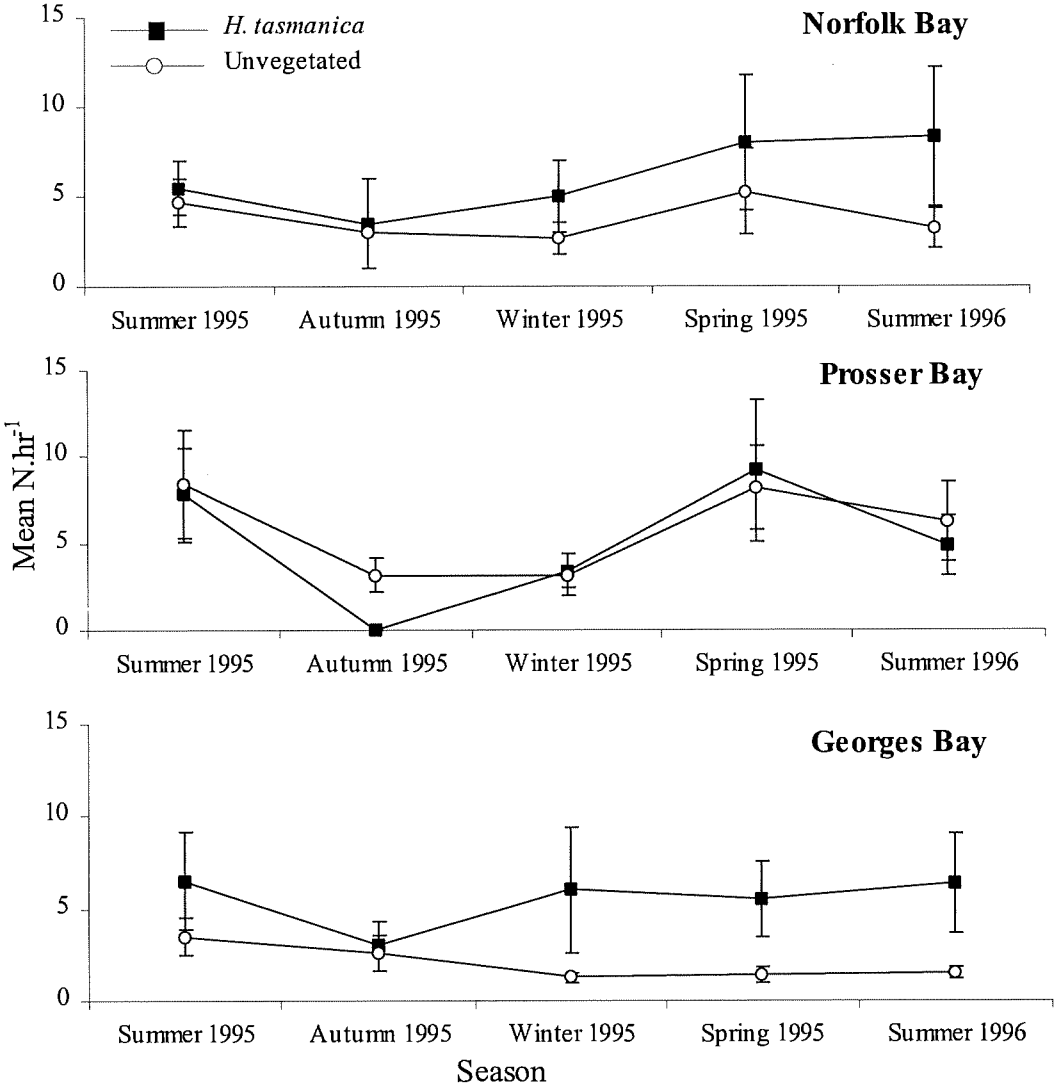



Fig. 5.3.4 Mean number of species (N.hr$^{-1}$) per season caught by gillnet at unvegetated and *Heterozostera tasmanica* sites in Norfolk Bay, Prosser Bay and Georges Bay between summer 1995 and 1996. Error bars are standard error.

### *5.3.3.4 Demersal community composition*

Univariate statistics suggest a strong separation of demersal fish communities at *Heterozostera* and unvegetated sites within Norfolk Bay and Georges Bay, but not Prosser

Bay. This is confirmed by the two-dimensional ordination plot where habitat differences are highly significant (two-way ANOSIM; $p<0.001$) (Fig. 5.3.5A), as are differences between areas ($p<0.001$). One-way ANOSIM between the six sites identifies an 'interaction' between habitat and area. While differences across habitats are highly significant for Georges Bay ($p<0.001$) and Norfolk Bay ($p=0.008$), there is no significant difference between fish communities from the two habitats at Prosser Bay ($p=0.135$).

Habitat differences are not as clear in the ordination of the gillnet samples (Fig. 5.3.5B). Differences between communities from the two habitat types across areas are non-significant when analysed by two-way ANOSIM ($p=0.07$), while differences between areas were significant ($p<0.001$). The one-way test for interaction between habitat type and area detects no significant differences in communities from the two habitat types at Norfolk Bay ($p=0.127$) and Prosser Bay ($p=0.849$), while the difference at Georges Bay is significant ($p=0.039$), but only just so. This difference is driven by a greater number of *Meuschenia freycineti* and *Aldrichetta forsteri* in the *Heterozostera* site.

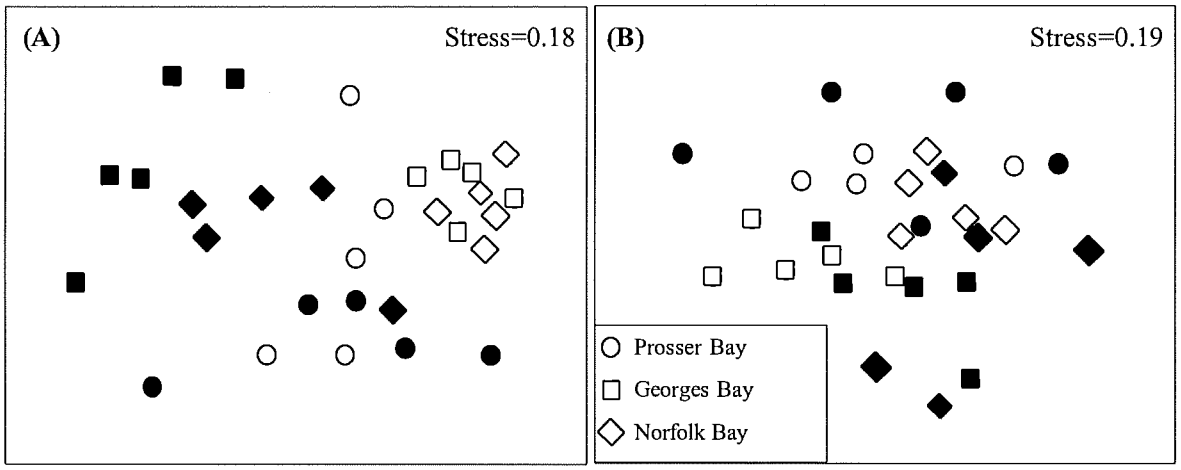



Fig. 5.3.5 Non-metric multi-dimensional scaling of fish communities sampled by (A) beam trawl and (B) gillnet at unvegetated (closed symbols) and *Heterozostera tasmanica* sites (open symbols) in Norfolk Bay, Prosser Bay and Georges Bay between summer 1995 and 1996.

Species contributions to the separation of demersal communities from the two habitat types are shown in Table 5.3.9. Despite heavy transformation to reduce the influence of abundant species, the three most abundant species from *Heterozostera* sites contribute most to the separation. *Stigmatopora argus* was consistently more abundant at *Heterozostera* sites in Norfolk Bay and Georges Bay, and was not caught at unvegetated sites in these areas (Fig. 5.3.6A). At Prosser Bay, although sample sizes were low, higher numbers of this species were found in *Heterozostera* than in unvegetated samples during summer. *Acanthaluteres spilomelanurus* occurs in substantially higher numbers at Norfolk Bay than other areas, but otherwise patterns of distribution are similar to *S. argus* (Fig. 5.3.6B). Similar patterns can be seen for *Neoodax balteatus* (Fig. 5.3.7A), however, *Nesogobius* sp1. was consistently higher in unvegetated sites in Norfolk Bay and Prosser Bay, particularly in winter (Fig. 5.3.7B).

Table 5.3.9 Ten species contributing most to the separation of communities from *Heterozostera* and unvegetated sites. Average abundance per haul at each site, and cumulative % contribution to the separation are given.

| Species | Average abundance | | Cumulative % |
|---|---|---|---|
| | *Heterozostera* | Unvegetated | |
| *Stigmatopora argus* | 63.5 | 0.5 | 10.5 |
| *Acanthaluteres spilomelanurus* | 117.2 | 1.9 | 19.8 |
| *Neoodax balteatus* | 61.2 | 4.6 | 27.8 |
| *Vanacampus phillipi* | 4.27 | 0.2 | 33.5 |
| *Cristiceps australis* | 4.6 | 2.2 | 39.1 |
| *Vincentia conspersa* | 6.9 | 0.2 | 44.6 |
| *Nesogobius sp. 1* | 2.1 | 4.4 | 49.4 |
| *Gymnapistes marmoratus* | 5.7 | 0.6 | 54.1 |
| *Platycephalus bassensis* | 1.1 | 3.7 | 58.4 |
| *Rhombosolea taparina* | 0.1 | 3.3 | 62.4 |

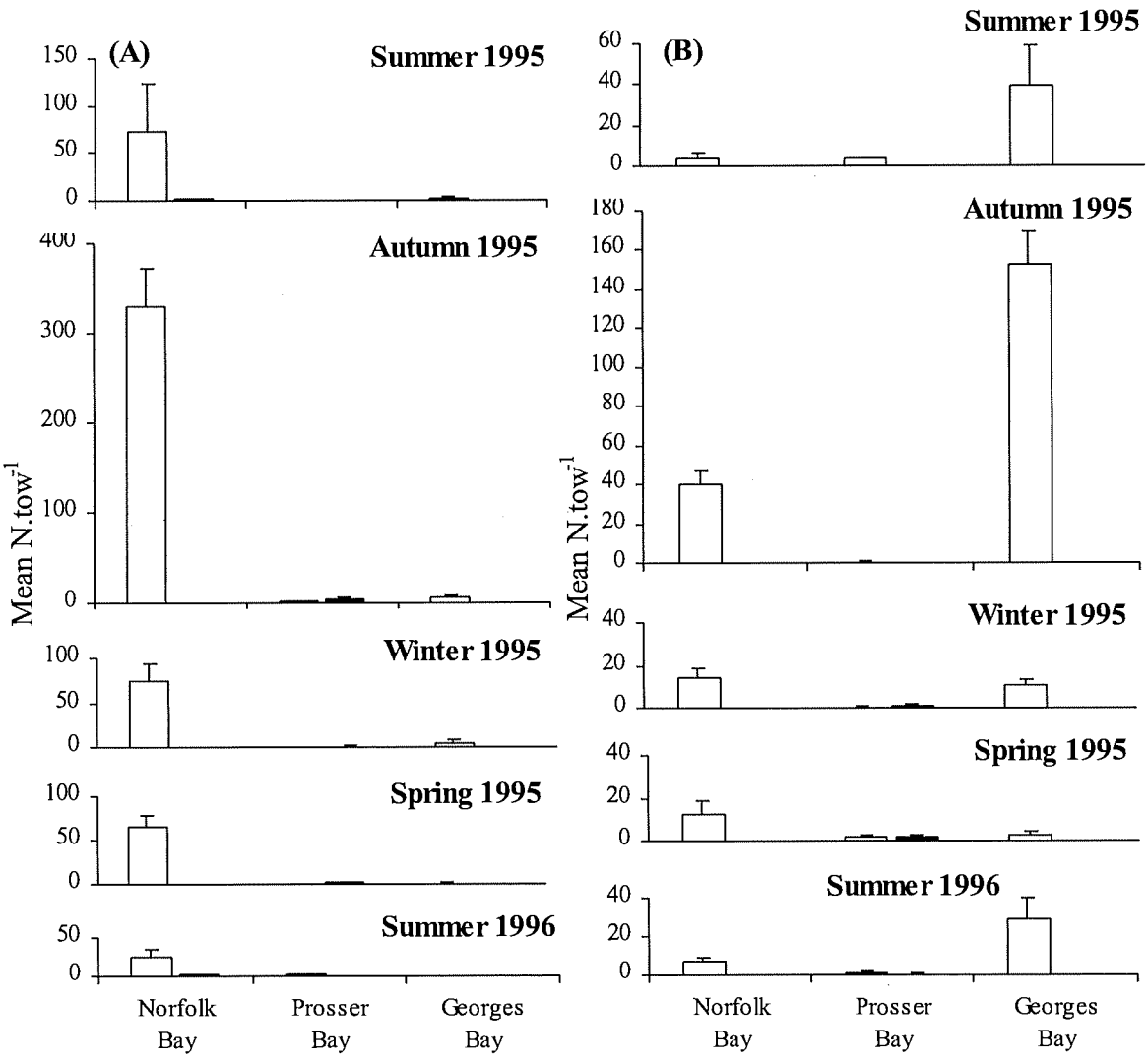



Fig. 5.3.6 Mean seasonal abundance (N.tow$^{-1}$) of (a) *Acanthaluteres spilomelanurus* and (b) *Stigmatopora argus* caught by beam trawl at *Heterozostera tasmanica* (light bars) and unvegetated sites (dark bars) in Norfolk Bay, Prosser Bay and Georges Bay between summer 1995 and 1996. Error bars are standard error.

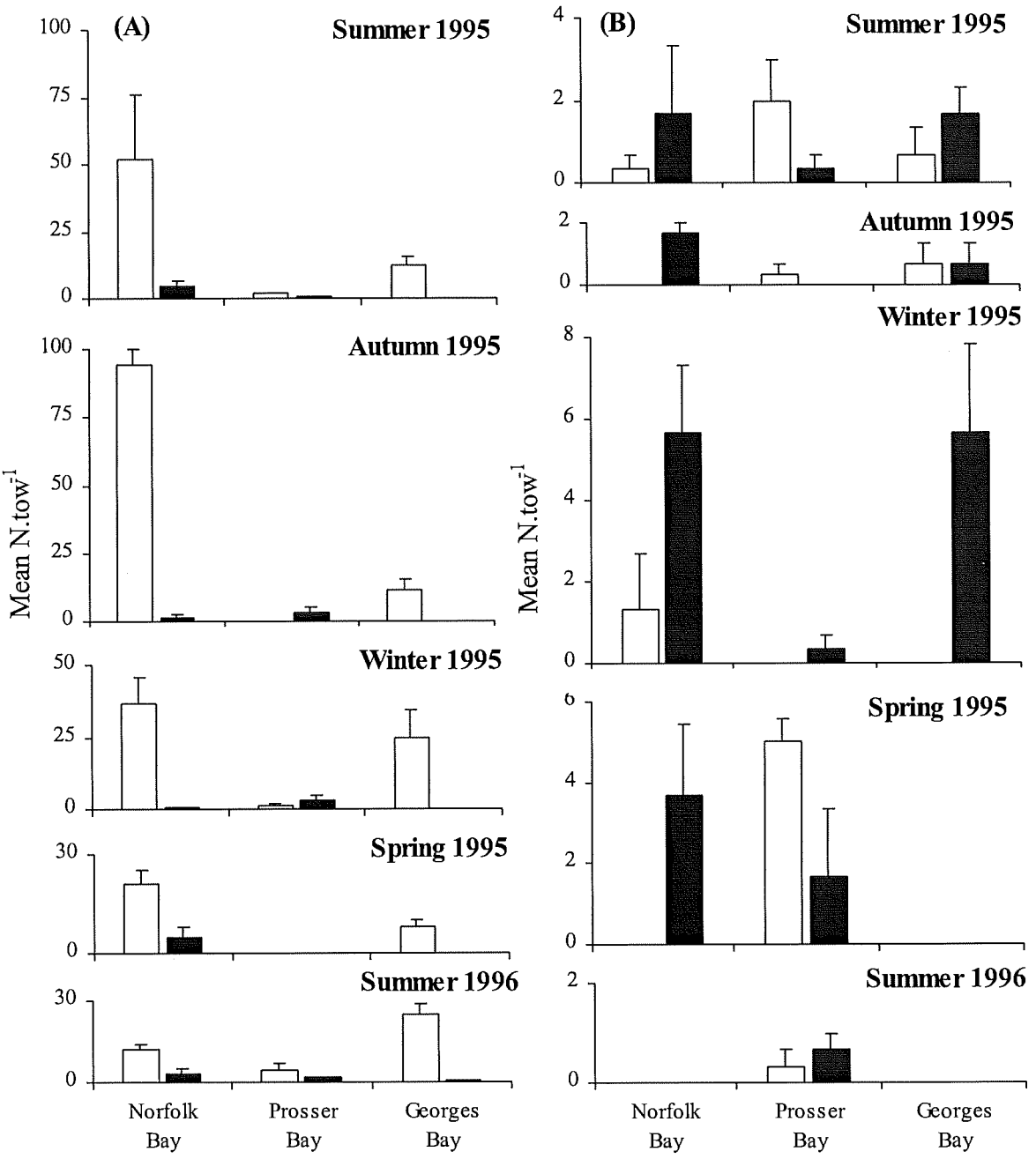



Fig. 5.3.7 Mean seasonal abundance (N.tow$^{-1}$) of (a) *Neoodax balteatus* and (b) *Nesogobius* sp1. caught by beam trawl at *Heterozostera tasmanica* (light bars) and unvegetated sites (dark bars) in Norfolk Bay, Prosser Bay and Georges Bay between summer 1995 and 1996. Error bars are standard error.

The abundance of *Rhombosolea tapirina* was consistently higher in unvegetated sites in Norfolk Bay and Georges Bay, with few fish present in Prosser Bay Fig. (5.3.8A). The abundance of *Platycephalus bassensis* was consistently higher in unvegetated sites in Norfolk Bay and Georges Bay, with the reverse true in Prosser Bay Fig. (5.3.8B). The distribution patterns of these species are less stable than those seen for species common in *Heterozostera* sites.

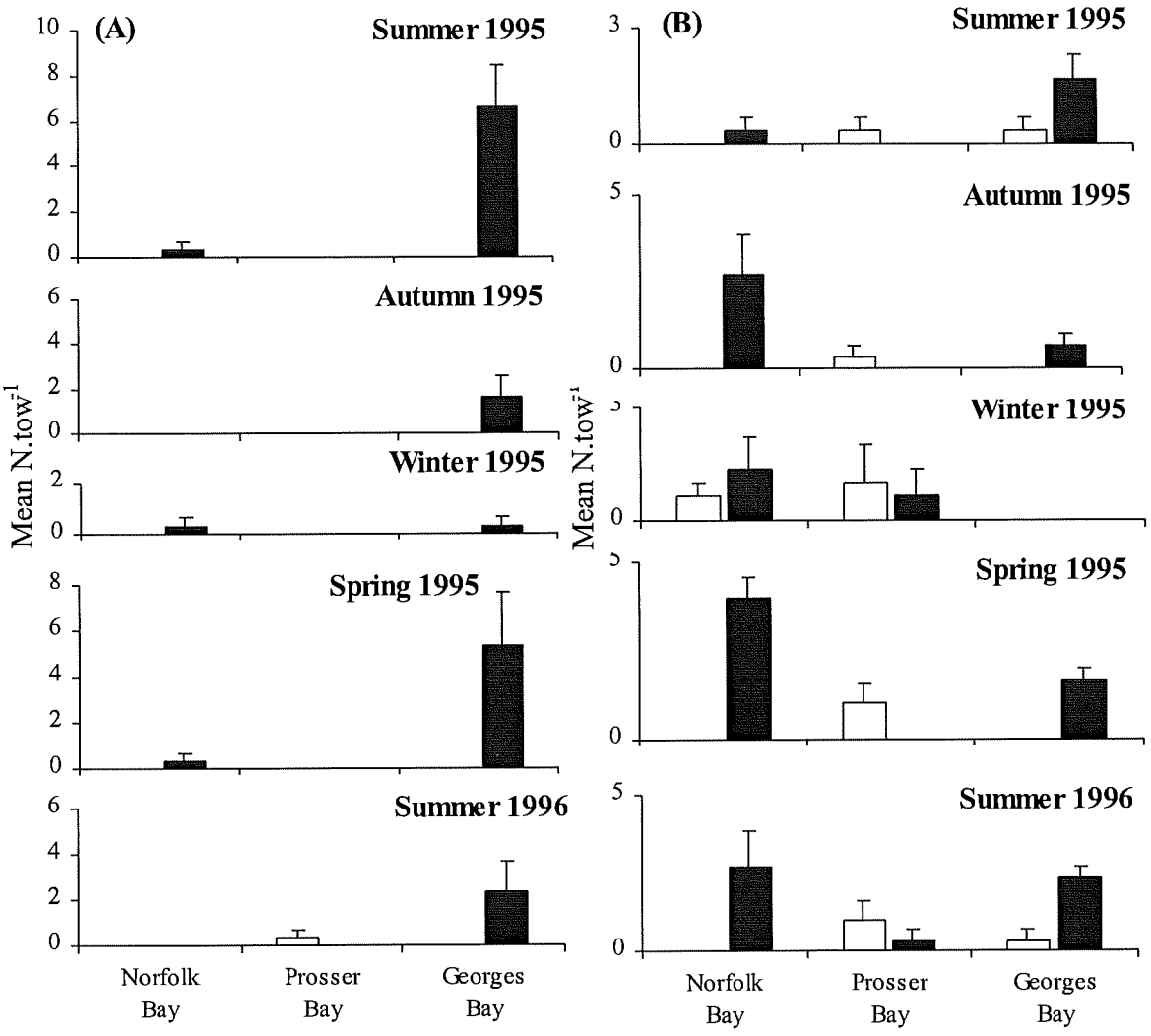



Fig. 5.3.8 Mean seasonal abundance (N.tow$^{-1}$) of (a) *Rhombosolea tapirina* and (b) *Platycephalus bassensis* caught by beam trawl at *Heterozostera tasmanica* (light bars) and unvegetated sites (dark bars) in Norfolk Bay, Prosser Bay and Georges Bay between summer 1995 and 1996. Error bars are standard error.

### *5.3.3.5 Seasonal variability*

No consistent patterns of seasonal variability in community composition were seen across areas or habitat types. However, the degree of seasonal variability (ie. the spread of points on the MDS plots) varies between habitat types. A measure of average similarity between samples taken from different seasons within sites, as generated by SIMPER analysis is presented in Table 5.3.10. With the exception of gillnet samples at Georges Bay, communities are considerably more stable across seasons in *Heterozostera* sites.

Table 5.3.10 Percentage similarity of samples taken by beam trawl and gillnet within sites at the three areas.

| | Norfolk Bay | | Prosser Bay | | Georges Bay | |
|---|---|---|---|---|---|---|
| | *Hz.* | Unveg. | *Hz.* | Unveg. | *Hz.* | Unveg. |
| **Beam-trawl** | 70.6 | 56.8 | 43.1 | 39.6 | 63.7 | 46.1 |
| **Gillnet** | 57.0 | 32.1 | 40.6 | 20.6 | 50.4 | 50.1 |

*5.3.2.6 Life-history stages and residency*

Species were grouped based on life-history stages and residency times; four groups were identified.

- Small demersal resident species, caught regularly as juveniles and adults in beam trawl samples: Of the 28 species caught more than once by beam trawl, 17 species fall into this group. Included are the three most abundant species from *Heterozostera* sites. This group comprises 98% of the beam-trawl catch from *Heterozostera* sites and 64% from unvegetated sites.

- Larger resident species occurring in beam-trawl samples as juveniles, and gillnet samples as adults: This group contains 7 species including *Platycephalus bassensis* and *Pseudophycis bachus*; both abundant in gillnet samples. Species from this group represent 62% of the total gillnet catch, 32% of fish caught by beam-trawl in unvegetated habitats and 1.5% of fish beam-trawled in *Heterozostera* habitats.

- Species occurring as juveniles in beam-trawl samples, but not seen as adults in gillnet samples: This group contains only 2 species (*Helicolenus barathri*, and *Lepidotrigla papilio*) with a total of just 13 individuals caught in both *Heterozostera* and unvegetated habitats.

- Species caught as juveniles or adults in gillnets, but not caught as juveniles in beam-trawl samples: This group includes 21 species making up 38 % of fish abundance from gillnet samples.

5.3.4 Discussion

*5.3.4.1 Fish abundance and diversity*

While beam trawl abundance in this study show the same pattern, with 91% of fish coming from *Heterozostera* sites, this pattern was not seen consistently across all areas as there was no difference in abundance between habitats in Prosser Bay. Previous studies have suggested seagrass beds contain a greater abundance and number of fish species than adjacent unvegetated areas (Heck *et al.* 1989, Connolly 1994, Grey *et al* 1996). However, this difference has been found to be influenced by the distance between unvegetated and seagrass habitats (Ferrell and Bell 1991), relative depth between habitats (Jenkins *et al.* 1997) and size-class of fish being sampled (Edgar and Shaw 1995b).

In the present study, geographic/environmental differences could account for the differences in *Heterozostera* beds and hence the difference in fish communities in Prosser Bay. However, Rees (1993) showed that historically there have been substantial seagrass beds within the bay, similar in area to those seen in Norfolk Bay and Georges Bay. Continued loss of seagrass in the Prosser Bay area over the past four decades has led to the

current situation in which seagrass densities are sparse in summer, with almost total dieback occurring during winter. Higher abundances of fish at sites with higher seagrass density have been related to increased food availability and increases in available shelter allowing greater protection from predators (Heck & Orth 1980, Edgar & Shaw 1995b). It appears that in Prosser Bay the seasonal presence of seagrass is in itself either not sufficient to attract high numbers of fish or post-settlement mortality is high. Results presented in Chapter 5.2 suggested that areas of *Heterozostera* that remain present throughout the colder months are important for sustaining the resident populations.

The presence of some fish species usually associated with *Heterozostera* habitats (eg. *Neoodax balteatus*, *Acanthaluteres spilomelanurus*) in moderate numbers at unvegetated sites in Prosser Bay is also worthy of note. It appears that in the absence of dense, stable seagrass beds, drift algae that was regularly present at the unvegetated sites becomes an important habitat. This is consistent with the findings of Sogard and Able (1991) who found that *Ulva* was an important fish habitat in areas devoid of seagrass, but was unimportant where seagrass was present.

The habitat response of larger non-resident species caught by gillnet was less marked than that of the small demersal species, and observed differences were generally not consistent across sites or seasons. The result does not, however, mean that seagrass habitats are unimportant to large mobile fish species. Given the greater abundance of prey species (benthic crustaceans and small fish) within seagrass beds, it is probable that larger fish would forage in these areas. This may indeed be the case, however, fish may be caught by gillnet over seagrass as they move between seagrass beds. Dietary analysis in previous studies suggest that many fish species present in high numbers in seagrass beds may be important prey for species such as *Platycephalus bassensis* and *Pseudophycis bachus* (Edgar and Shaw 1995a). A more detailed analysis of fish movement over short time scales, and fish diets would be required to fully asses the importance of seagrass beds as feeding areas for larger fish.

#### *5.3.4.2 Fish communities*

The morphology of small demersal fish associated with the two habitat types were quite distinct. Demersal fish communities in *Heterozostera* beds were dominated by small resident species growing to not greater than 10 cm. Many of these species were cryptic and associated closely with, and in some cases mimicked, the seagrass blades. *Acanthaluteres spilomelanurus* and *Neoodax balteatus* become cryptic within seagrass beds by hanging vertically/diagonally in the water column with the seagrass canopy, and both species exhibit camouflage colouration. The syngnathids are a more extreme case, morphologically mimicking the seagrass blades.

Dorso-ventral flattening, a close association with the substratum, and sand coloration are common amongst families abundant in beam trawl catches from unvegetated habitats

(Gobidae, Pleuronectidae, Platycephalidae). Juveniles of larger species made up a far greater proportion of the catch in unvegetated habitats (31%) than in *Heterozostera* beds (1.5%).

Many studies have found seagrass beds to be important nursery areas for larger fish species, with larvae recruiting to seagrass beds and juveniles or adults then moving into deeper water habitats (see Bell and Pollard 1984). In contrast, in the present study all abundant species were resident, with the seagrass beds being important for all life-history stages. Only two species (*Helicolenus barathri* and *Lepidotrigla papilio*) were found in seagrass beds only as juveniles. It is worthy of note that of a total of 65 species captured during the study, only 10 species were caught in both gillnets and beam trawl samples. This highlights the point that the *Heterozostera* beds do not play a particularly significant role as a nursery area for large fish species throughout southern and eastern Tasmania.

Seagrass beds are characterised by a stable community of small fish when compared with unvegetated sites. This is shown by the scatter of points in multidimensional space in Fig. 5.3.5. There is a particularly tight association between communities from seagrass sites at Norfolk Bay and Georges Bay, while scatter is far greater among samples from unvegetated sites. A more variable community composition is also seen in the unstable *Heterozostera* beds in Prosser Bay. Similar differences in community stability have been shown in studies using analyses elsewhere in Australia (Connolly 1994, Grey *et al.* 1996).

Area was more important than habitat type in the structuring of communities of larger mobile fish caught by gillnets as community composition did not vary between vegetated and unvegetated sites. Greater mobility and foraging range of these larger species, combined with the long duration of gillnet sets may mean that the techniques employed in the study were not capable of detecting any differences in habitat usage.

## 5.4 Comparison of *Posidonia* and *Heterozostera* communities

### 5.4.1 Introduction

Gradients in environmental parameters such as salinity and turbidity are common in estuaries. Where seagrass occurs predominantly within estuaries, these variables can be expected to affect the composition of fish communities. However, the vast majority of seagrass beds in Tasmania occur in bays or sheltered coastal areas where there is little variation in environmental gradients and physical differences in habitat type are more likely to effect fish communities. One such major physical difference is seen where more than one species of seagrass is present within an area. While the fish faunas within monospecific seagrass beds has been well described (Burchmore *et al.* 1984, Humphries *et al.* 1992, Ferrell *et al.* 1993), few studies have compared the fish communities in adjacent habitats containing different seagrass species.

In a comparison between the structure of fish communities in beds of *Posidonia australis* and *Zostera capricorni*, Middleton *et al.* (1984) found that while neither abundance or species richness differed between habitats there was a significant difference in the community composition. Beds of *P. australis* generally contained larger resident species and larger juveniles of seasonally transient species utilising the habitat as a nursery area. These patterns were related to the variations in the structural complexity of the canopy of the two seagrass species.

Five species of seagrass (*Heterozostera tasmanica*, *P. australis*, *Halophila australis*, *Zostera muelleri* and *Amphibolis antarctica*) occur in coastal waters of Tasmania. The two most dominant species by far are *P. australis* and *H. tasmanica*. While *H. tasmanica* is widely distributed around the entire coast, *P. australis* is largely restricted to the north coast and Bass Strait islands. In the lower reaches of the Tamar River both species occur in monospecific stands (Jordan 1995), which provides an opportunity to compare fish faunas between seagrass species and remove geographic differences as a variable. Therefore, the aim of this chapter is to compare the seasonal abundance, species richness, biomass and community composition of the fish fauna of *H. tasmanica* and *P. australis* beds and examine the importance of such habitats as a nursery area for economically important species.

### 5.4.2 Methods

Two sites representative of *Heterozostera tasmanica* and *Posidonia australis* habitats were sampled seasonally between summer 1995 and summer 1996 (Table 5.4.1). On each occasion the demersal fish fauna was sampled at each site with three non-overlapping 3-min beam trawl tows at a tow speed of 2 knots. All sampling was conducted within 2 hours of high tide during the day. Beam trawl catch rates were calculated as the number of fish per tow ($N.tow^{-1}$). Full details of sampling regime and gear is presented in Chapter 4.

Table 5.4.1 Site and habitat characteristics of *Heterozostera tasmanica* and *Posidonia australis* beds sampled in the Tamar River.

| Site | Habitat | Seagrass Density | Depth (m) | Fetch |
|---|---|---|---|---|
| Sandy Beach (SB) | *H. tasmanica* | medium | 2 - 6 | 3km East |
| Kelso Bay (KB) | *H. tasmanica* | low | 2 - 5 | 1.5km NE. |
| Lagoon Bay (LB) | *P. australis* | high | 2 - 5 | 1.8km NE. |
| NW. Bank (NWB) | *P. australis* | medium | 2 - 4 | 1.3km West |

Variation in fish abundance, biomass and number of species caught per tow across habitats (*H. tasmanica or P. australis*), sites and seasons was assessed using a three-way nested ANOVA. Habitat and season were considered to be fixed factors, while site was considered a random factor, and nested within habitat. Data were tested for conformity to the assumptions of ANOVA using the $F_{max}$ test for heteroscedascity (Sokal and Rohlf 1981) and by examining residual plots. Tests revealed that $ln(x+1)$ transformation was appropriate for fish abundance and wet weight data; no transformation was necessary for analysis of number of species per tow. Post-hoc multiple comparison tests for main effects were not performed, as significant first order interactions were detected in all analysis.

Differences in fish community structure between the two habitats were analysed by non-metric multi-dimensional scaling (MDS) and analysis of similarities (ANOSIM). A matrix of ranked similarities was generated using the Bray-Curtis similarity index (Bray and Curtis 1957) applied to $x^{0.25}$ transformed abundance data. Two dimensional MDS plots were generated from this matrix.

Significance of differences between fish communities in the two habitat types was tested by one-way analysis of similarities (ANOSIM) applied to the ranked similarity matrix. While a nested two factor analysis would be appropriate (factors habitat, and site nested in habitat), the ANOSIM two-way nested algorithm lacks power where degrees of freedom of the main effects are low. One way ANOSIM was therefore used to test differences between habitats, and between sites within habitats.

## 5.4.3 Results

### *5.4.3.1 Hydrography*

All sites showed a strong oceanic influence. While surface salinity varied marginally, bottom salinities were relatively constant varying from 31-34 ppt at all sites, and across all seasons. Bottom temperature ranged seasonally from 8.8° to 17.9° C. Differences in temperature were significantly different between seasons in both habitats (Table 5.4.2), with summer and spring temperatures being greater than those in autumn and winter (Ryans Q test; p=0.05). There was no significant difference between habitats and no interaction indicating that similar seasonal changes in temperature occurred across both habitat types.

Table 5.4.2 Two-way ANOVA (habitat and season: fixed factors) of temperature and salinity for *Posidonia australis* and *Heterozostera tasmanica* sites in the Tamar river.

| Factor | | Temperature | | | Salinity | | |
|---|---|---|---|---|---|---|---|
| | DF | MS | F | Prob. | MS | F | Prob. |
| Season | 3 | 54.4 | 253.3 | <0.001 | 1.4 | 0.8 | 0.527 |
| Habitat | 1 | 0.45 | 1.9 | 0.264 | 15.0 | 7.1 | 0.075 |
| Habitat*Season | 3 | 0.24 | 1.1 | 0.396 | 2.1 | 1.2 | 0.371 |
| Error | 8 | 0.21 | | | 1.7 | | |

*5.4.3.2 Fish composition*

A total of 2,680 fish (1,230 from *Heterozostera,*1,450 from *Posidonia*) from 41 species in 18 families were collected by beam trawl in the Tamar River (Table 5.4.3). A full list of scientific and common names is presented in Appendix 1. The four dominant species, *Acanthaluteres spilomelanurus, A. vitteger, Neoodax balteatus and Vincentia conspersa*, comprised 91% of the total sample. Three of these abundant species were small, permanent residents, caught seasonally as juveniles and consistently as adults, while *A. vitteger* was the only abundant species caught predominantly as juveniles. Seven species (*Pseudophycis bachus, Neosebastes thetedis, Helecolinis barathri, Pseudocaranx dentex, Upenichthys lineatus, Notolabrus tetricus, Dotalabrus aurantiacus* and *Haletta semifasciata*) occurred in samples only as juveniles, but these species were rare, representing only 2.5% of total fish abundance.

Species represented by a single individual (8 species), and all species of atherinids were excluded from statistical analysis. Atherinids were captured only on one sampling date, and the schooling behaviour of these fish makes information on distribution unreliable. Taxonomic differences between juvenile *Acanthaluteres vitteger* and *A. spilomelanurus* were not fully appreciated early in the study, so these species have been combined for analysis. Available data suggest similar abundances and distribution of these two species in the Tamar River. Abundances of the remaining 29 species broken down by habitat and site (seasons pooled) are listed in Table 5.4.3.

*5.4.3.3. Fish abundance, species richness and biomass*

Neither abundance nor number of species varied significantly between *Heterozostera* and *Posidonia* sites (Table 5.4.4, Fig. 5.4.1A,B). For both variables, a significant interaction between season and site was detected. The degree of seasonal variability in abundance is similar in the two habitats, but differs significantly between sites within habitat (season*site interaction; p=0.002). Of particular note is the difference in mean abundance at the two *Posidonia* sites in summer and autumn 1995 (Fig. 5.4.1A). The high total abundance at Lagoon Bay in these seasons was largely due to high numbers of *Acanthaluteres* spp. Abundance at *Heterozostera* sites peaked in summer with uniform low abundance in other seasons.

Table 5.4.3 The total number of individuals and percentage of the total individuals (excluding atherinids) for the fish taxa collected by beam trawl from two *Heterozostera tasmanica* and two *Posidonia australis* sites in the Tamar River between summer 1995 and 1996.

| | ***Heterozostera tasmanica*** | | | | ***Posidonia australis*** | | | |
|---|---|---|---|---|---|---|---|---|
| | **Sandy Beach** | | **Kelso Bay** | | **Lagoon Bay** | | **NW Bank** | |
| Species | n | % | n | % | n | % | n | % |
| **Gobiesocidae** | | | | | | | | |
| *Alabes dorsalis* | 0 | 0 | 2 | 0.4 | 0 | 0 | 0 | 0 |
| **Moridae** | | | | | | | | |
| *Pseudophycis bachus* | 1 | 0.2 | 2 | 0.4 | 0 | 0 | 0 | 0 |
| **Sygnathidae** | | | | | | | | |
| *Hippocampus abdominalis* | 3 | 0.5 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Stigmatopora argus* | 1 | 0.2 | 5 | 0.9 | 14 | 1.4 | 1 | 0.2 |
| *Stigmatopora nigra* | 5 | 0.8 | 2 | 0.4 | 0 | 0 | 0 | 0 |
| *Vanacampus phillipi* | 9 | 1.4 | 5 | 0.9 | 0 | 0 | 0 | 0 |
| **Scorpaenidae** | | | | | | | | |
| *Neosebastes thetidis* | 0 | 0 | 1 | 0.2 | 1 | 0.1 | 0 | 0 |
| *Helicolenus barathri* | 2 | 0.3 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Gymnapistes marmoratus* | 4 | 0.6 | 3 | 0.5 | 1 | 0.1 | 1 | 0.2 |
| **Platycephalidae** | | | | | | | | |
| *Platycephalus bassensis* | 17 | 2.6 | 5 | 0.9 | 0 | 0 | 0 | 0 |
| **Apogonidae** | | | | | | | | |
| *Vincentia conspersa* | 65 | 9.9 | 81 | 14.3 | 2 | 0.2 | 3 | 0.7 |
| **Carangidae** | | | | | | | | |
| *Pseudocaranx dentex* | 2 | 0.3 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Mullidae** | | | | | | | | |
| *Upeneichthys lineatus* | 1 | 0.2 | 1 | 0.2 | 10 | 1.0 | 7 | 1.7 |
| **Labridae** | | | | | | | | |
| *Notolabrus tetricus* | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0.72 |
| *Dotalabrus aurantiacus* | 0 | 0 | 0 | 0 | 7 | 0.7 | 5 | 1.2 |
| **Odacidae** | | | | | | | | |
| *Haletta semifasciata* | 3 | 0.5 | 1 | 0.2 | 0 | 0 | 7 | 1.7 |
| *Neoodax balteatus* | 34 | 5.2 | 114 | 20.1 | 54 | 5.6 | 63 | 15.0 |
| *Siphonognathus radiatus* | 0 | 0 | 0 | 0 | 5 | 0.5 | 1 | 0.2 |
| **Clinidae** | | | | | | | | |
| *Cristaceps australis* | 3 | 0.5 | 14 | 2.5 | 2 | 0.2 | 2 | 0.5 |
| *Heteroclinus tristis* | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.5 |
| *Heteroclinus perspicillatus* | 1 | 0.2 | 4 | 0.7 | 0 | 0 | 0 | 0 |
| **Gobiidae** | | | | | | | | |
| *Nesogobius pulchellus* | 8 | 1.2 | 11 | 1.9 | 0 | 0 | 0 | 0 |
| *Nesogobius* sp.1 | 1 | 0.2 | 3 | 0.5 | 0 | 0 | 0 | 0 |
| *Nesogobius* sp.3 | 6 | 0.9 | 3 | 0.5 | 0 | 0 | 0 | 0 |
| **Monocanthidae** | | | | | | | | |
| *Acanthaluteres vittiger* | 11 | 1.7 | 26 | 4.6 | 240 | 24.7 | 111 | 26.5 |
| *Scobinichthys granulatus* | 0 | 0 | 0 | 0 | 2 | 0.2 | 1 | 0.2 |
| *Eubalichthys gunnii* | 0 | 0 | 1 | 0.2 | 1 | 0.1 | 0 | 0 |
| *Meuschenia freycineti* | 2 | 0.3 | 4 | 0.7 | 5 | 0.5 | 10 | 2.4 |
| *A. spilomelanurus* | 150 | 22.8 | 275 | 48.6 | 626 | 64.5 | 92 | 22.0 |
| **Tetraodontidae** | | | | | | | | |
| *Tetractenos glaber* | 0 | 0 | 3 | 0.5 | 0 | 0 | 0 | 0 |

Mean number of species varied consistently in *Posidonia* sites, peaking in autumn and at a minimum in spring (Fig. 5.4.1B). Variability at *Heterozostera* sites is less consistent, resulting in an interaction between season and site bordering on significance (p=0.051).

Table 5.4.4 Three way nested ANOVA (habitat and season: fixed factors; site random factor nested in habitat) of log transformed abundances and number of species of fish (N.tow$^{-1}$) in *Heterozostera tasmanica* and *Posidonia australis* sites in the Tamar River.

| | | Log abundance | | | | Number of species | | |
|---|---|---|---|---|---|---|---|---|
| Factor | Hypothesis | DF | MS | F | Prob | MS | F | Prob |
| Habitat, *A* | *Quasi F* | 1 | 0.9 | 0.4 | 0.632 | 28.0 | 1.4 | 0.444 |
| Season, *B* | *B/BC(A)* | 4 | 9.8 | 5.3 | 0.022 | 15.6 | 1.6 | 0.262 |
| Site(Habitat), *C(A)* | *C(A)/BC(A)* | 2 | 2.0 | 1.1 | 0.387 | 1.9 | 0.2 | 0.822 |
| Habitat*Season, *AB* | *AB/BC(A)* | 4 | 3.9 | 2.1 | 0.169 | 27.4 | 2.8 | 0.099 |
| Season*Site(Habitat),*BC(A)* | *BC(A)/E* | 8 | 1.8 | 2.6 | 0.023 | 9.7 | 2.2 | 0.051 |
| Error, *E* | | 40 | 0.7 | | | 4.5 | | |

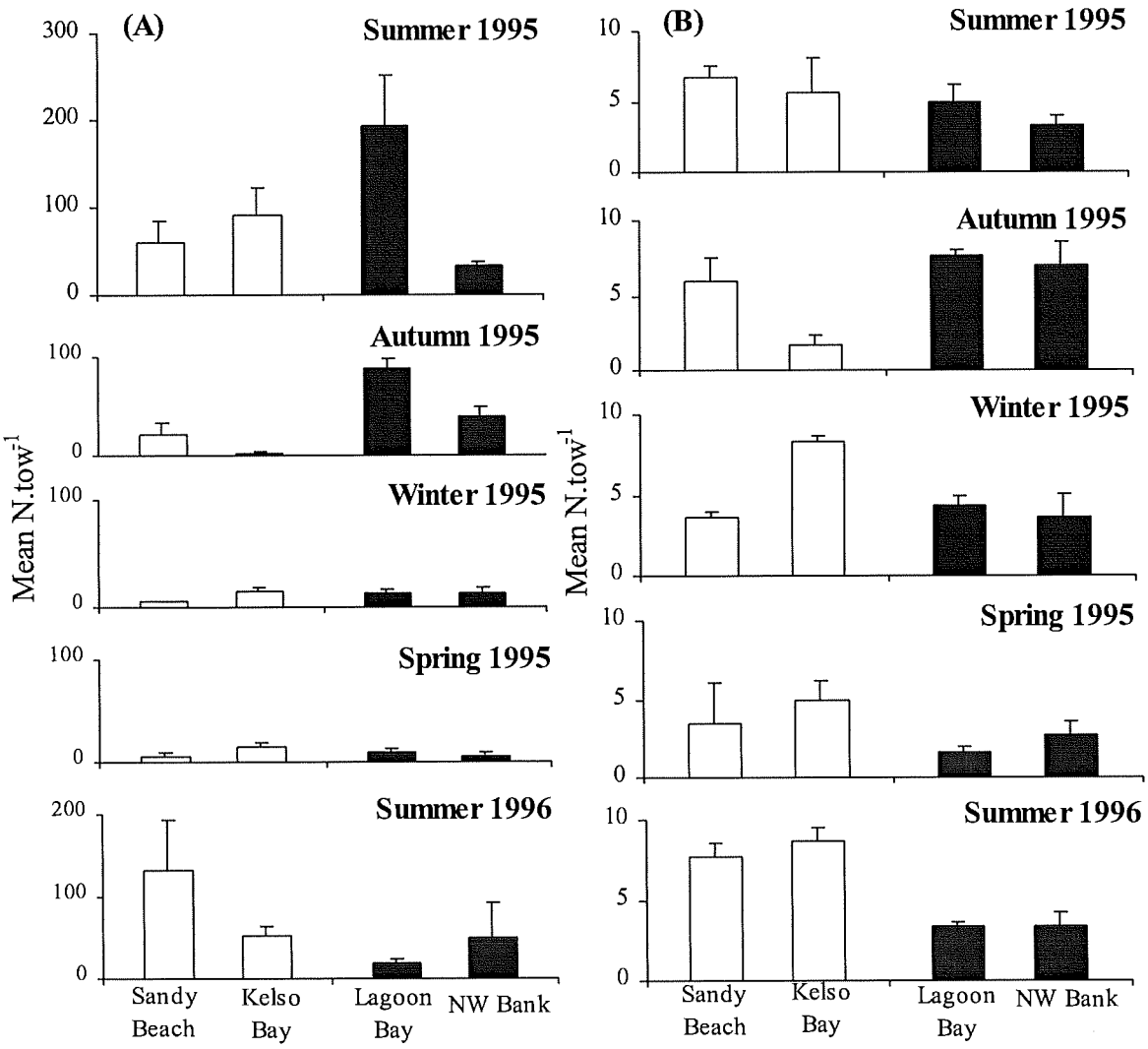



Fig. 5.4.1 (A) Mean abundance of fishes (N.tow$^{-1}$) and (B) mean number of species of fish (N.tow$^{-1}$) collected by beam trawl from two *Heterozostera tasmanica* (light bars) and two *Posidonia australis* sites (dark bars) in the Tamar River between summer 1995 and 1996. Error bars are standard error.

A greater biomass of fish was caught from *Posidonia* sites than *Heterozostera* sites in autumn 1995 and summer 1996 (Table 5.4.5, Fig. 5.4.2). Biomass of fish did not vary significantly between sites within habitats (p=0.760), however a significant interaction between habitat and season was detected (p=0.004). These differences are due to high numbers of small individuals, but also an increase in the number of larger fish, mainly *Acanthaluteres vitteger* and *Meuschenia freycinetti.*

Table 5.4.5 Three way nested ANOVA (habitat and season: fixed factors; site random factor nested in habitat) of biomass of fish (kg.tow$^{-1}$) in *Heterozostera tasmanica* and *Posidonia australis* sites in the Tamar River.

| | | Log wet weight | | | |
|---|---|---|---|---|---|
| **Factor** | Hypothesis | DF | MS | F | Prob. |
| Habitat, *A* | *Quasi F* | 1 | 0.09 | 15.31 | 0.056 |
| Season, *B* | *B/BC(A)* | 4 | 0.10 | 6.87 | 0.011 |
| Site(Habitat), *C(A)* | *C(A)/BC(A)* | 2 | 0.01 | 0.28 | 0.760 |
| Habitat*Season, *AB* | *AB/BC(A)* | 4 | 0.14 | 9.57 | 0.004 |
| Season*Site(Habitat), *BC(A)* | *BC(A)/E* | 8 | 0.01 | 0.72 | 0.676 |
| Error, *E* | | 40 | 0.02 | | |

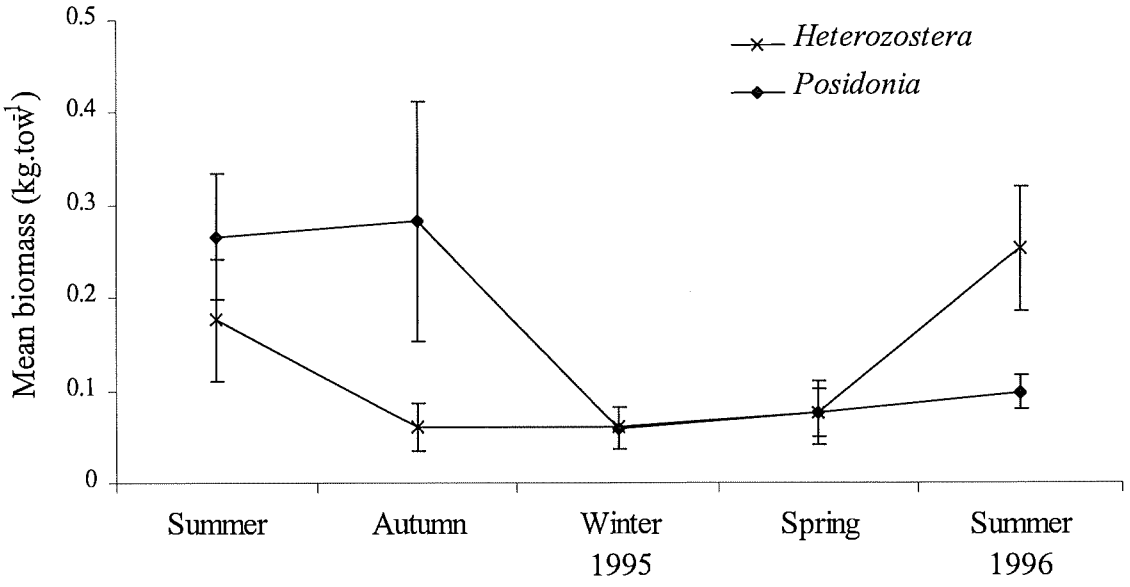



Fig. 5.4.2 Mean biomass of fishes (kg.tow$^{-1}$) collected by beam trawl from two *Heterozostera tasmanica* and two *Posidonia australis* sites in the Tamar River between summer 1995 and 1996. Error bars are standard error.

### *5.4.3.4. Community composition*

Given the close proximity of *Heterozostera* and *Posidonia* sites there was a high proportion of species caught more than once that were unique to each habitat type. Thirteen of 29 species were caught only in *Heterozostera*, 5 species were unique to *Posidonia* and 12 species were common to both habitats (Table 5.4.3). The majority of

species showed preference for one of the habitats, with only 7 out of 30 species having less than 75% of their total catch in a single habitat type.

Sixteen species were considered common (>75% of catch) only in *Heterozostera* sites. All species of Gobidae were caught exclusively in *Heterozostera* and were distributed uniformly between Sandy Beach and Kelso Bay. Three species of Syngnathids occurred exclusively in *Heterozostera* beds, while a fourth (*S. argus*) showed no distinct habitat preference. Juveniles of the commercially and recreationally important sand flathead (*Platycephalus bassensis*) were also captured at both *Heterozostera* sites, but never at *Posidonia* sites. The apogonid, *Vincenta conspersa* was the most abundant species considered common only at *Heterozostera* sites.

Six species were considered common only in *Posidonia* beds. Two species of Labrid were caught exclusively in *Posidonia,* probably reflecting the close proximity of reef habitat to these sites. Juveniles of *Upeneichthys vlamingii* and *Acanthaluteres vitteger* showed a preference for *Posidonia* beds. The abundant *Acanthaluteres* sp. showed a degree of preference for *Posidonia*, with 70% of individuals coming from this habitat. Seven species were more evenly distributed between habitats, particularly the third most abundant species, *Neoodax balteatus*, which showed no habitat preference. Where species were present in *Posidonia* and *Heterozostera* in numbers sufficient to give some indication of size compositions (*Stigmatopora argus*, *Neoodax balteatus*, *Acanthaluteres sp.*, *Meuschenia freycineti* and *Cristiceps australis*), length frequency distribution were compared across habitats. Differences were only slight and tended to be inconsistent, the one exception being *Stigmatopora* argus where individuals caught in *Posidonia* were consistently larger than those caught in *Heterozostera.*

Non-metric multi-dimensional scaling clearly separates fish communities from *Posidonia* and *Heterozostera* sites (Fig. 5.4.3). Analysis of similarities shows the difference between fish communities to be highly significant ($p<0.001$), but differences between sites within habitat to be non-significant (*Posidonia*; $p=0.571$, *Heterozostera*; $p=0.754$). The 10 species contributing most to the separation are listed in Table 5.4.6. Despite heavy transformation, the most significant discriminator is the highly abundant *Acanthaluteres* spp. grouping. In autumn, winter and spring, there were more *Acanthaluteres* spp. in *Posidonia* than in *Heterozostera* sites, while this is not the case in either summer sample (Fig. 5.4.5). In summer 1995, *Acanthaluteres* spp. showed a site preference for the Lagoon Bay *Posidonia* bed, but in the same season, catches of this genus from NW Bank *Posidonia* were lower than catches from either *Heterozostera* site. Due to the possibility that *Acanthaluteres* spp. may be showing a site preference rather than a habitat preference, the multivariate analyses were redone with this species removed from the matrix. The resulting ordination plot (Fig. 5.4.4) shows few differences from Fig. 5.4.3. Significance levels change only slightly for differences between sites within habitats, and there is no change for differences between habitats.

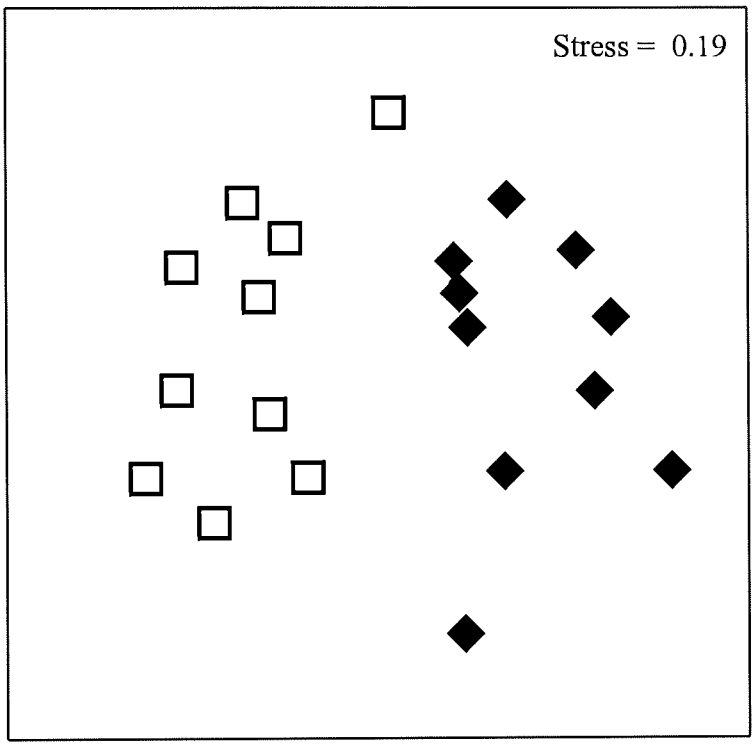



Fig. 5.4.3 Non-metric multi-dimensional scaling of fish communities from two *Posidonia australis* (squares) and two *Heterozostera tasmanica* sites (diamonds) in the Tamar River.

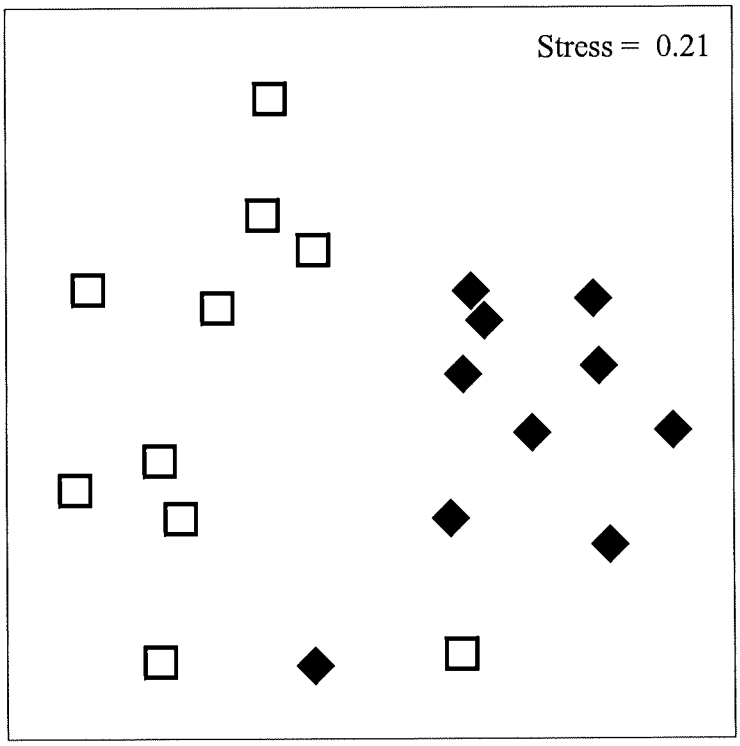



Fig. 5.4.4 Non-metric multi-dimensional scaling of fish communities from two *Posidonia australis* (squares) and two *Heterozostera tasmanica* sites (diamonds) in the Tamar River, excluding *Acanthaluteres* spp.

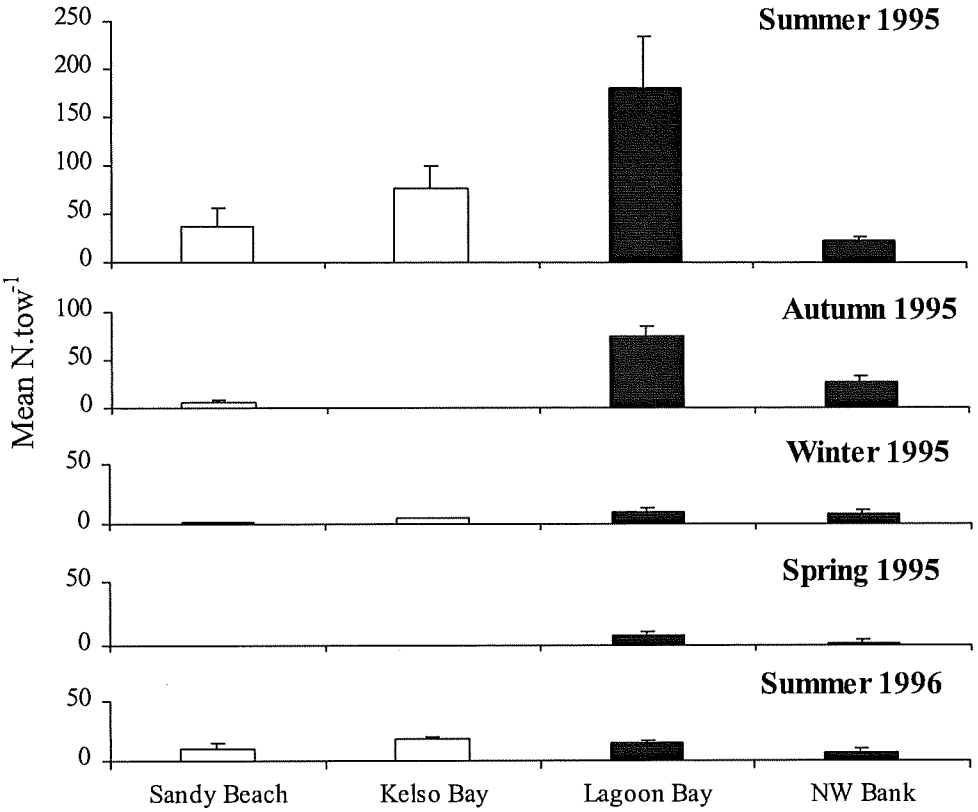



Fig. 5.4.5 Mean abundance (N.tow$^{-1}$) of *Acanthaluteres* spp. collected by beam trawl from two *Heterozostera tasmanica* (light bars) and two *Posidonia australis* sites (dark bars) in the Tamar River between summer 1995 and 1996. Error bars are standard error.

Patterns of distribution were far more consistent for the next four species contributing to the separation of the communities. *Vincentia conspersa* (Fig. 5.4.6A), *Nesogobius pulchellus* (Fig. 5.4.6B), and *Platycephalus bassensis* (Fig. 5.4.7A) were caught regularly in high numbers at either *Heterozostera* site, but rarely in *Posidonia* sites. *Dotalabrus auranticus* was caught consistently in low numbers at either *Posidonia* site, but never in *Heterozostera* sites (Fig. 5.4.7B).

Table 5.4.6 SIMPER analysis of *Heterozostera tasmanica* and *Posidonia australis* fish communities in the Tamar River based on Bray-curtis simmilarities between 4$^{th}$ root transformed fish abundance (N.tow$^{-1}$). Average dissimilarity between groups (ANOSIM) = 68.1%.

| Species | Average abundance | | Percent |
|---|---|---|---|
| | *P. australis* | *H. tasmanica* | |
| *Acanthaluteres spp.* | 125.2 | 47.3 | 12.3 |
| *V. conspersa* | 0.6 | 14.0 | 9.4 |
| *N. pulchellus* | 0 | 2.3 | 7.0 |
| *P. bassensis* | 0 | 2.1 | 6.1 |
| *D. aurantiacus* | 1.4 | 0.0 | 5.2 |
| *M. freycineti* | 1.9 | 0.8 | 4.8 |
| *S. argus* | 1.8 | 0.8 | 4.8 |
| *C. australis* | 0.5 | 1.3 | 4.4 |
| *Vanacampus phillipi* | 0.0 | 0.75 | 4.0 |
| *Upeneichthys lineatus* | 2.1 | 0.3 | 4.0 |

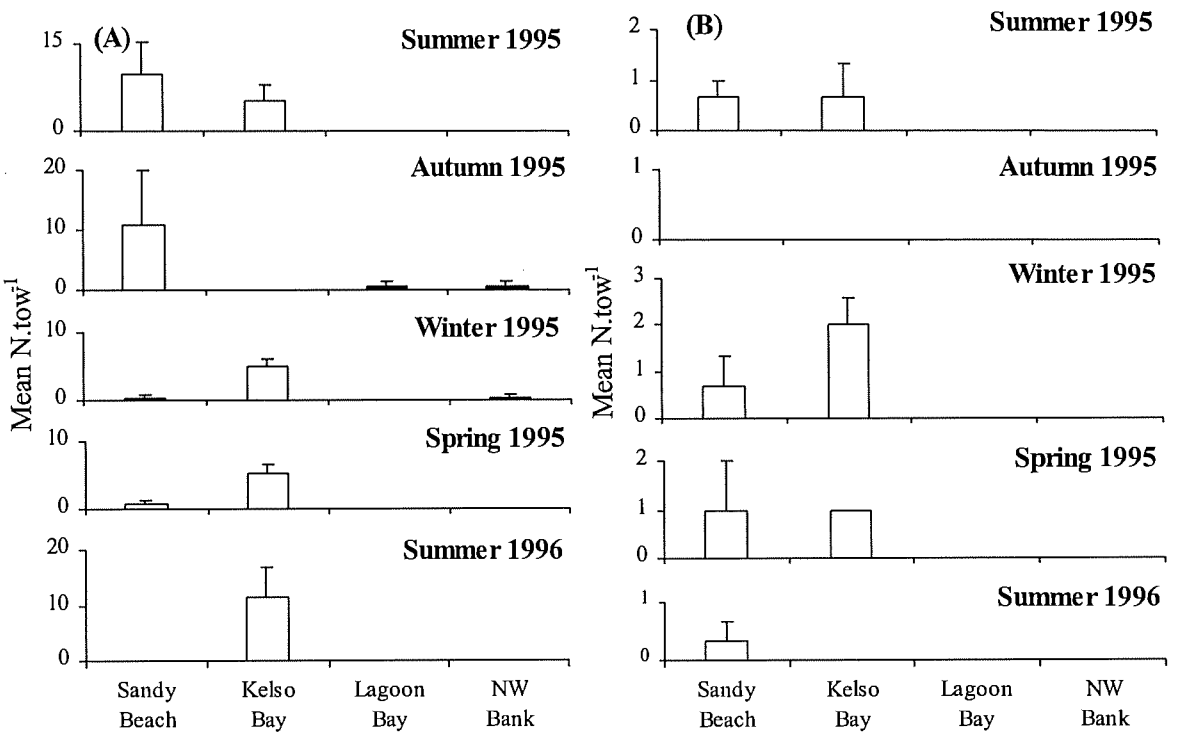



Fig. 5.4.6 Mean seasonal abundance (N.tow$^{-1}$) of (A) *Vincentia conspersa* and (B) *Nesogobius pulchellus* caught by beam trawl from two *Heterozostera tasmanica* (light bars) and two *Posidonia australis* sites (dark bars) in the Tamar River between summer 1995 and 1996. Error bars are standard error.

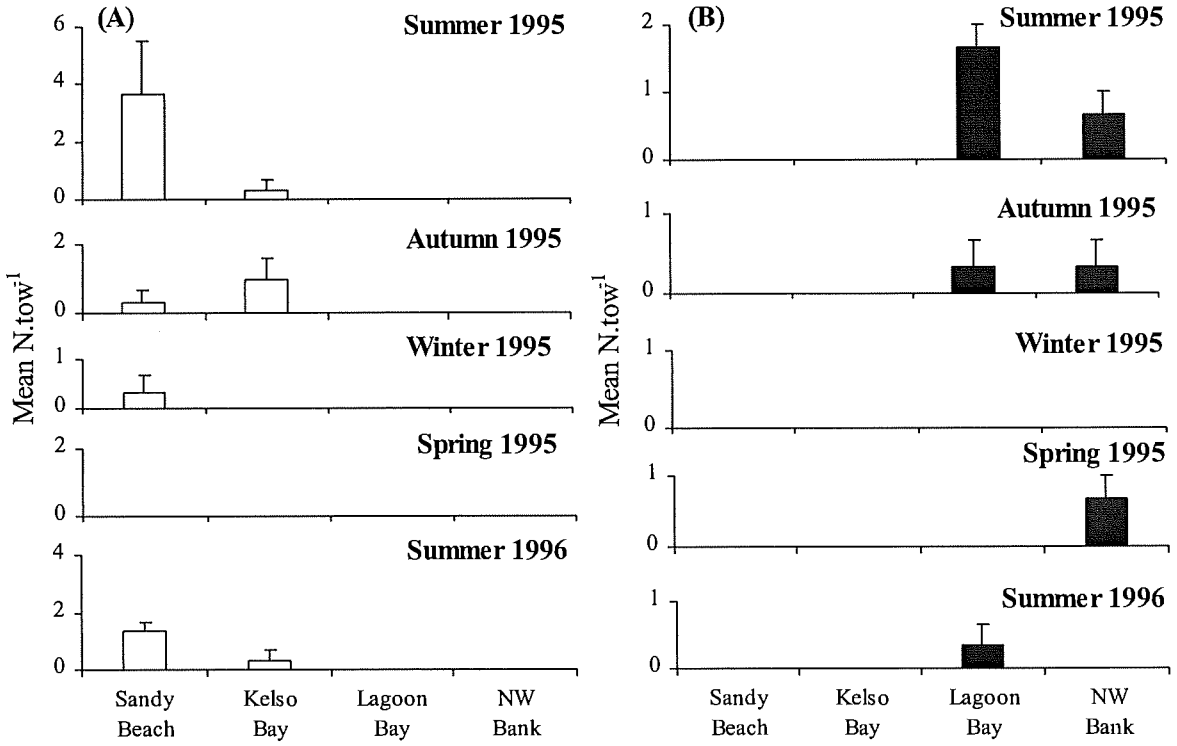



Fig. 5.4.7 Mean seasonal abundance (N.tow$^{-1}$) of (A) *Platycephalus bassensis* and (B) *Dotalabrus auranticus* caught by beam trawl from two *Heterozostera tasmanica* (light bars) and two *Posidonia australis* sites (dark bars) in the Tamar River between summer 1995 and 1996. Error bars are standard error.

### 5.4.4 Discussion

The present study demonstrated that while *Posidonia* and *Heterozostera* beds in the Tamar River supported similar abundances and numbers of species, the overall fish communities were significantly different. Over 60% of species captured were restricted to one or other habitat, and over 75% of species showed a clear preference for one habitat. These differences in fish community may relate to physical differences in habitat structure. The nature of the canopy cover is physically different with *Heterozostera* beds consisting of plants with a short, narrow leaf which is essentially straight while *Posidonia* blades are relatively long and broad, and tend to lay over towards the tops of the blades creating a closed-in canopy. Sediment properties within the beds also differ considerably with *Posidonia* occurring in course sand, while *Heterozostera* is found in areas with a finer silt substrate.

The degree of variability between biotic communities in different species of seagrass varies depending on the taxa being studied. A common pattern amongst invertebrate studies is that many species are common to adjacent beds of different seagrass species (Wells and Rose 1985). However, Bell and Westoby (1986a) found distinct decapod communities within *Zostera* and *Posidonia* beds. Edgar (1990) found a strong response of invertebrate fauna to a more diverse range of habitat structures with separate faunas associated with detached macrophytes, seagrass fronds, plant debris and seagrass rhizomes.

Fish community composition commonly varies considerably between different seagrass species (Martin and Cooper 1981, Stoner 1983). Fish communities from *Zostera capricorni* and *Posidonia australis* beds (both of similar leaf morphology to the species in the current study) have been compared in two previous studies from New South Wales. Young (1981) found significant differences in fish species composition and species abundances between the two habitats. Similarly, Middleton *et al.* (1984) found that while abundance and species richness of fish did not vary significantly between habitats, fish communities from the two habitats were distinct. While more fish were caught in *Zostera* beds, biomass was greater in *Posidonia*.

The distribution of seagrass species is often related to environmental gradients, such as depth (Shepherd and Robertson 1989), turbidity, salinity and water temperature (Shepherd and Womersley 1981). As such, separating community differences driven by differences in seagrass species compared to those caused by other environmental gradients is difficult. However, the aim of the study was to investigate the relative importance to fish communities of *Posidonia* and *Heterozostera* seagrass beds *in situ*, rather than the direct effects of the different seagrass blade morphology. Hence no attempt has been made to separate out variables such as flow regimes, sediment properties.

Fish biomass differed between the two habitats on two occasions, and on both occasions was greater in *Posidonia* sites. Analysis of size compositions indicates this was due to

greater numbers of fish in *Posidonia* sites (although not significantly so) and an small increase in numbers of larger individuals (mainly *A. vittiger* and *M. freycineti*). Middleton *et al.* (1984) found consistently greater fish biomass in *Posidonia* beds than *Zostera* beds, and showed a serial usage pattern amongst several species whereby juveniles first recruit into *Zostera* beds then move into *Posidonia* beds with increasing size. Such ontogenetic progressions were rare in the present study, with only *Stigmatopora argus* showing a notable difference in size structure between the two habitats.

Few species were found to be utilising either *Posidonia* or *Heterozostera* beds as a nursery area, none of which are economically important species. *Pseudophycis bachus*, *N. thetidis*, *N. tetricus*, *D. aurantiacus* and *H. semifasciata* were caught only as juveniles in beam trawl samples, but are also common in seagrass beds as adults (D. Mills unpubl. data). *Upenichthys lineatus* was caught in both *Posidonia* and *Heterozostera* beds only as juveniles, but adults of this species are captured regularly only over sand.

## 5.5 Spatial patterns in shallow beach assemblages

### 5.5.1 Introduction

While seagrass beds are widely recognised as an important nursery area for juvenile fishes in shallow coastal habitats (see Bell and Pollard 1989), unvegetated habitats are becoming increasing recognised as an important habitat for juvenile fishes (Ferrell and Bell 1991, Jenkins *et al.* 1997). Such coastal unvegetated habitats regarded as nursery areas range in exposure from surf zones (Bennett 1989) to sheltered sand beaches (Ayvazian and Hyndes 1995, Hyndes *et al.* 1996). While levels of food production are higher in seagrass beds (Edgar 1990, Edgar *et al.* 1994), enhanced food production in shallow unvegetated habitats can occur due to the presence of detached macrophytes (Robertson and Lenanton 1984, May and Jenkins 1992) and regular phytoplankton blooms (McLachlan *et al.* 1981).

Throughout southern and eastern Tasmania, unvegetated habitats are the dominant habitat type in shallow water with most seagrass beds restricted to waters deeper than ~1 m at low tide, generally outside a broad sandflat. While sampling sites were chosen to be representative of nearshore beach habitats, the majority of which were unvegetated, several sites consisted of patchy beds of *Heterozostera tasmanica* with small amounts of *Zostera capricorni* also present. The aim of this chapter is to examine the relative importance of such shallow beach habitats for fish throughout southern and eastern Tasmania. This is done by comparing the abundance, number of species and community composition of the fish fauna at a large range of sites in around the south-east coast.

### 5.5.2 Methods

Full details of survey sites and methods for sampling of nearshore beach habitats is presented in Chapter 4.2.4. In brief, the fish fauna were sampled monthly from December 1996 to February 1997 at 27 sites throughout south-eastern Tasmania with a 25 m beach seine with a 3 m drop and mesh size of 20 mm (see Fig. 4.6, Table 4.3). A single seine haul was made at each site on each sampling occasion.

Heirachical cluster analysis of Bray-Curtis similarities between fish abundance data from all sites showed four major groupings at 45% similarity. Identified groupings were imposed on a two-dimensional MDS plot generated from the same similarity matrix. Fetch values for each site, measured as the greatest distance reached by a 30 degree arc from the sampling site, were represented for each point by varying the size of the point (Clarke and Warwick 1994). The correlation between fish community composition and fetch can thus be assessed by observing how well sites of different fetch fit the groupings generated by the cluster analysis.

### 5.5.3 Results and Discussion

A total of 46 species in 21 families were caught in 81 seine hauls at the 27 sites throughout south-east Tasmania (Table 5.5.1). A full list of scientific and common names is presented in Appendix 1. Overall, the nearshore beach fish fauna was dominated by atherinids (predominantly *Atherinosoma microstoma* and *Leptatherina presbyteroides)*, Eastern Australian salmon (*Arripis trutta)*, gobies (predominantly *Nesogobius* sp.1), flounders (*Ammotretis rostratus* and *Rhombosolea tapirina*) and yellow-eye mullet (*Aldrichetta forsteri*). The relative dominance of species differed slightly between unvegetated and *Heterozostera* sites, with *Neoodax balteatus* and *Acanthaluteres spilomelanurus* making up around 47% of the catch at the seagrass site in Carnarvon Bay. The dominance of individual families also differed between areas, which may be related to the varying levels of exposure and estuarine gradients.

Species of commercial and recreational importance (*A. rostratus, A. lituratus, R. tapirina, A. trutta, Sillago flindersi, Platycephalus bassensis* and *A. forsteri*) dominated the overall catch at most sites, particularly at unvegetated habitats, where they made up from 6% of the abundance at Alonnah Beach to 97% at Roaring Beach. The majority of the individuals of these species were 0+ and 1+ fish indicating that shallow beach habitats in south-east Tasmania are an important habitat for juveniles of these species. Detailed analysis of the size composition and biological parameters of the key economically important species is presented in Chapters 5.9 to 5.11.

In addition, the community structure of fishes in this study is similar to that of shallow sand habitats of Port Phillip Bay which were also dominated by atherinids, gobies, mullet and flounders (Jenkins *et al.* 1997). The major difference is the presence of large numbers of *Arripis trutta* in the present study while very few were caught in Port Phillip Bay.

Groupings of fish communities generated by cluster analysis show a good correlation with fetch (Fig. 5.5.1). Mean fetch values for groups 1 to 4 are 4.7, 2.8, 1.9, and 0.85 respectively. Five notable outliers, where fetch values are unexpectedly high or low, appear in the MDS plot. In four cases, it can be argued fetch under-represents exposure/wave activity due to geographic effects. The two low exposure points that fall within group 1 are both from the Tasman Peninsula (see Fig. 4.6). Safety Cove (SC: fetch = 0.9) is within Port Arthur, which opens directly to the Southern Ocean and receives significant swell activity from the south-east to south-west quadrants. The mouth of Port Arthur is shaped in such a way to funnel swells into the bay. Cliffs along the western shore of Port Arthur result in swells being deflected to the sampling site. Similarly, while White Beach (WB: fetch = 2.2) is subject to similar funnelling effects and deflection resulting in high wave activity. The low fetch outliers in group 2 can be explained by the same mechanism. Parsons Bay (PB: fetch = 0.4) is adjacent to White Beach and experiences similar wave activity. The entrance channel to Cloudy Bay Lagoon (CL: fetch = 0.2) on southern Bruny Island is very similar in aspect and conditions to the Port Arthur area.

The site showing high fetch within group 3 is Gordon Beach (GB) within the D'Entrecasteaux Channel. The high fetch at this site is in a north-easterly direction, and is due to the shape of Bruny Island. Much of the distance of the fetch is within the sheltered waters of Apollo Bay on Bruny Island.

Species contributions to the separation of the four groups identified in the MDS plot are shown in Table 5.5.2. Group 1 are characterised by high abundances of *Arripis trutta*, while group 2 are characterised by high abundances *Leptatherina presbyteroides* and intermediate abundance of *Rhombosolea tapirina*. Group 3 are characterised by high abundances of *Nesogobius* sp.1 and *R. tapirina*, while group 4 were driven by high abundances of *Atherinosoma microstoma* and *Pseudaphritis urvillii*.

Table 5.5.1 Total number of individuals and percentage of the total individuals for the fish taxa collected by beach seine at nearshore beach habitats in five areas throughout south-east Tasmania from December 1996 to February 1997.

| | **Bruny Island** | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Location | Alonnah Bch. | | Cloudy Lagoon | | Cloudy Lag. Entrance | | Isthmus Bay | |
| Species | n | % | n | % | n | % | n | % |
| **Atherinidae** | | | | | | | | |
| *Atherinosoma microstoma* | 0 | 0 | 0 | 0 | 0 | 0 | 198 | 31.8 |
| *Kestratherina esox* | 0 | 0 | 3 | 0.53 | 0 | 0 | 26 | 4.18 |
| *Leptatherina presbyteroides* | 1990 | 90.9 | 25 | 4.43 | 19 | 3.70 | 16 | 2.57 |
| **Syngnathidae** | | | | | | | | |
| *Mitotichthys mollisoni* | 1 | 0.05 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Platycephalidae** | | | | | | | | |
| *Platycephalus bassensis* | 10 | 0.46 | 1 | 0.18 | 0 | 0 | 0 | 0 |
| **Arripidae** | | | | | | | | |
| *Arripis trutta* | 8 | 0.36 | 3 | 0.53 | 376 | 73.1 | 0 | 0 |
| **Mugilidae** | | | | | | | | |
| *Aldrichetta forsteri* | 0 | 0 | 1 | 0.18 | 4 | 0.78 | 6 | 0.96 |
| **Odacidae** | | | | | | | | |
| *Neoodax balteatus* | 35 | 1.60 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Clinidae** | | | | | | | | |
| *Heteroclinus perspicillatus* | 0 | 0 | 7 | 1.24 | 0 | 0 | 0 | 0 |
| **Gobiidae** | | | | | | | | |
| *Nesogobius* sp.1 | 27 | 1.23 | 438 | 77.7 | 33 | 6.42 | 257 | 41.3 |
| *Pseudogobius olorum* | 0 | 0 | 1 | 0.18 | 0 | 0 | 0 | 0 |
| **Pleuronectidae** | | | | | | | | |
| *Ammotretis rostratus* | 94 | 4.29 | 24 | 4.25 | 59 | 11.5 | 9 | 1.44 |
| *Rhombosolea tapirina* | 25 | 1.14 | 61 | 10.8 | 23 | 4.47 | 110 | 17.7 |

Table 5.5.1 (Cont)

| | D'Entrecasteaux Channel | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | Gordon Bch. | | Conningham Bch. | | NW Bay | | Dover Bch. | | Dover Hotel Bch. | | Roaring Bch. | |
| **Atherinidae** | n | % | n | % | n | % | n | % | n | % | n | % |
| *Atherinosoma microstoma* | 0 | 0 | 80 | 16.8 | 95 | 13.6 | 2 | 0.43 | 0 | 0 | 0 | 0 |
| *Kathetostoma laeve* | 0 | 0 | 1 | 0.21 | 0 | 0 | 1 | 0.22 | 0 | 0 | 0 | 0 |
| *Kestratherina esox* | 10 | 1.88 | 4 | 0.84 | 40 | 5.75 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Leptatherina presbyteroides* | 105 | 19.8 | 75 | 15.7 | 59 | 8.48 | 166 | 36.2 | 680 | 72.8 | 0 | 0 |
| **Syngnathidae** | | | | | | | | | | | | |
| *Hippocampus abdominalis* | 2 | 0.38 | 0 | 0 | 2 | 0.29 | 0 | 0 | 1 | 0.11 | 0 | 0 |
| *Mitotichthys mollisoni* | 0 | 0 | 0 | 0 | 0 | 0 | 8 | 1.74 | 0 | 0 | 0 | 0 |
| *Mitotichthys semistriatus* | 0 | 0 | 0 | 0 | 2 | 0.29 | 2 | 0.44 | 0 | 0 | 0 | 0 |
| *Stigmatopora argus* | 0 | 0 | 2 | 0.42 | 7 | 1.01 | 3 | 0.65 | 0 | 0 | 0 | 0 |
| *Stigmatopora nigra* | 1 | 0.19 | 0 | 0 | 9 | 1.29 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Vanacampus phillipi* | 0 | 0 | 0 | 0 | 9 | 1.29 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Scorpaenidae** | | | | | | | | | | | | |
| *Gymnapistes marmoratus* | 0 | 0 | 0 | 0 | 80 | 11.5 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Platycephalidae** | | | | | | | | | | | | |
| *Platycephalus bassensis* | 1 | 0.19 | 0 | 0 | 3 | 0.43 | 0 | 0 | 5 | 0.53 | 0 | 0 |
| **Sillaginidae** | | | | | | | | | | | | |
| *Sillago flindersi* | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0.65 | 0 | 0 | 0 | 0 |
| **Arripidae** | | | | | | | | | | | | |
| *Arripis trutta* | 13 | 2.45 | 3 | 0.63 | 0 | 0 | 75 | 16.3 | 0 | 0 | 96 | 80.7 |
| **Sparidae** | | | | | | | | | | | | |
| *Acanthopagrus butcheri* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.22 | 0 | 0 | 0 | 0 |
| **Mugilidae** | | | | | | | | | | | | |
| *Aldrichetta forsteri* | 1 | 0.19 | 0 | 0 | 3 | 0.43 | 26 | 5.66 | 0 | 0 | 5 | 4.20 |
| **Odacidae** | | | | | | | | | | | | |
| *Neoodax balteatus* | 84 | 15.8 | 0 | 0 | 17 | 2.44 | 33 | 7.19 | 5 | 0.53 | 0 | 0 |
| **Leptoscopidae** | | | | | | | | | | | | |
| *Lesueurina platycephala* | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0.87 | 0 | 0 | 4 | 3.36 |
| **Bovichtidae** | | | | | | | | | | | | |
| *Pseudaphritis urvillii* | 0 | 0 | 0 | 0 | 11 | 1.58 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Clinidae** | | | | | | | | | | | | |
| *Cristaceps australis* | 3 | 0.56 | 1 | 0.21 | 7 | 1.01 | 3 | 0.65 | 0 | 0 | 0 | 0 |
| *Heteroclinus adelaidae* | 1 | 0.18 | 0 | 0 | 1 | 0.14 | 0 | 0 | 0 | 0 | 0 | 0 |
| *H. perspicillatus* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.22 | 0 | 0 | 0 | 0 |
| **Gobiidae** | | | | | | | | | | | | |
| *Favonigobius tamarensis* | 0 | 0 | 0 | 0 | 2 | 0.29 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nesogobius hinsbyi* | 26 | 4.90 | 115 | 24.1 | 1 | 0.14 | 0 | 0 | 20 | 2.14 | 0 | 0 |
| *Nesogobius pulchellus* | 4 | 0.75 | 0 | 0 | 4 | 0.57 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nesogobius* sp.1 | 92 | 17.3 | 14 | 2.93 | 214 | 30.7 | 24 | 5.23 | 160 | 17.1 | 0 | 0 |
| *Nesogobius* sp.6 | 0 | 0 | 0 | 0 | 2 | 0.29 | 3 | 0.65 | 0 | 0 | 0 | 0 |
| *Tasmanogobius lasti* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 12 | 1.28 | 0 | 0 |
| **Pleuronectidae** | | | | | | | | | | | | |
| *Ammotretis lituratus* | 1 | 0.19 | 0 | 0 | 0 | 0 | 3 | 0.65 | 0 | 0 | 10 | 8.40 |
| *Ammotretis rostratus* | 30 | 5.65 | 34 | 7.13 | 0 | 0 | 35 | 7.62 | 10 | 1.07 | 4 | 3.36 |
| *Rhombosolea tapirina* | 133 | 25.0 | 141 | 29.6 | 112 | 16.1 | 31 | 6.75 | 38 | 4.07 | 0 | 0 |
| **Monocanthidae** | | | | | | | | | | | | |
| *A. spilomelanurus* | 24 | 4.52 | 2 | 0.42 | 6 | 0.86 | 5 | 1.09 | 1 | 0.11 | 0 | 0 |
| *Meuschenia freycineti* | 0 | 0 | 0 | 0 | 2 | 0.29 | 2 | 0.44 | 0 | 0 | 0 | 0 |
| **Tetraodontidae** | | | | | | | | | | | | |
| *Tetractenos glaber* | 0 | 0 | 5 | 1.05 | 8 | 1.15 | 1 | 0.22 | 2 | 0.21 | 0 | 0 |

Table 5.5.1 (Cont)

| | Derwent Estuary | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Location | Lowecroft Bay | | Cornelian Bay | | Howrah Bch. | | Kingston Bch. | | Nutgrove Bch. | |
| Species | n | % | n | % | n | % | n | % | n | % |
| **Anguillidae** | | | | | | | | | | |
| *Anguilla reinhardtii* | 0 | 0 | 1 | 0.17 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Galaxiidae** | | | | | | | | | | |
| *Galaxias* sp. | 10 | 1.44 | 8 | 1.33 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Galaxias maculatus* | 2 | 0.29 | 1 | 0.17 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Galaxias truttaceus* | 3 | 0.43 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Atherinidae** | | | | | | | | | | |
| *Atherinosoma microstoma* | 478 | 69.1 | 211 | 35.1 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Leptatherina presbyteroides* | 0 | 0 | 10 | 1.66 | 8 | 14.0 | 200 | 49.5 | 109 | 39.2 |
| **Syngnathidae** | | | | | | | | | | |
| *Hippocampus abdominalis* | 0 | 0 | 0 | 0 | 3 | 5.26 | 0 | 0 | 0 | 0 |
| *Stigmatopora argus* | 1 | 0.14 | 1 | 0.17 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Scorpaenidae** | | | | | | | | | | |
| *Gymnapistes marmoratus* | 7 | 1.01 | 26 | 4.33 | 0 | 0 | 0 | 0 | 2 | 0.72 |
| **Platycephalidae** | | | | | | | | | | |
| *Platycephalus bassensis* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.25 | 1 | 0.36 |
| **Triglidae** | | | | | | | | | | |
| *Lepidotrigla papilio* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Arripidae** | | | | | | | | | | |
| *Arripis trutta* | 0 | 0 | 2 | 0.33 | 7 | 12.3 | 22 | 5.44 | 12 | 4.32 |
| **Sparidae** | | | | | | | | | | |
| *Acanthopagrus butcheri* | 0 | 0 | 2 | 0.33 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Mugilidae** | | | | | | | | | | |
| *Aldrichetta forsteri* | 73 | 10.5 | 236 | 39.3 | 10 | 17.5 | 104 | 25.7 | 25 | 8.99 |
| **Leptoscopidae** | | | | | | | | | | |
| *Crapalatus munroi* | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.49 | 0 | 0 |
| *Lesueurina platycephala* | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0.99 | 0 | 0 |
| **Bovichtidae** | | | | | | | | | | |
| *Pseudaphritis urvillii* | 72 | 10.4 | 16 | 2.66 | 0 | 0 | 0 | 0 | 2 | 0.72 |
| **Blenniidae** | | | | | | | | | | |
| *Parablennius tasmanianus* | 0 | 0 | 0 | 0 | 1 | 1.75 | 0 | 0 | 0 | 0 |
| **Clinidae** | | | | | | | | | | |
| *Heteroclinus perspicillatus* | 0 | 0 | 1 | 0.17 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Gobiidae** | | | | | | | | | | |
| *Favonigobius tamarensis* | 5 | 0.72 | 1 | 0.17 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nesogobius hinsbyi* | 1 | 0.14 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nesogobius* sp.1 | 2 | 0.29 | 39 | 6.49 | 1 | 1.75 | 0 | 0 | 10 | 3.60 |
| *Pseudogobius olorum* | 21 | 3.03 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tasmanogobius lasti* | 3 | 0.43 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Pleuronectidae** | | | | | | | | | | |
| *Ammotretis lituratus* | 0 | 0 | 0 | 0 | 0 | 0 | 24 | 5.94 | 0 | 0 |
| *Ammotretis rostratus* | 0 | 0 | 0 | 0 | 18 | 31.6 | 27 | 6.68 | 19 | 6.83 |
| *Rhombosolea tapirina* | 13 | 1.88 | 35 | 5.82 | 9 | 15.8 | 20 | 4.95 | 93 | 33.4 |
| *Taratretis derwentensis* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 1.80 |
| **Monocanthidae** | | | | | | | | | | |
| *A. spilomelanurus* | 0 | 0 | 6 | 1.00 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Meuschenia freycineti* | 1 | 0.14 | 3 | 0.50 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Tetraodontidae** | | | | | | | | | | |
| *Tetractenos glaber* | 0 | 0 | 2 | 0.33 | 0 | 0 | 0 | 0 | 0 | 0 |

Table 5.5.1 (Cont)

| | Derwent Estuary (cont) | | | |
|---|---|---|---|---|
| Location | Rokeby Bch. | | Sth. Arm Bch. | |
| Species | n | % | n | % |
| **Anguillidae** | | | | |
| *Anguilla reinhardtii* | 0 | 0 | 0 | 0 |
| **Galaxiidae** | | | | |
| *Galaxias* sp. | 0 | 0 | 0 | 0 |
| *Galaxias maculatus* | 22 | 8.53 | 0 | 0 |
| *Galaxias truttaceus* | 0 | 0 | 0 | 0 |
| **Atherinidae** | | | | |
| *Atherinosoma microstoma* | 1 | 0.39 | 0 | 0 |
| *Leptatherina presbyteroides* | 1 | 0.39 | 83 | 79.8 |
| **Syngnathidae** | | | | |
| *Hippocampus abdominalis* | 0 | 0 | 0 | 0 |
| *Stigmatopora argus* | 1 | 0.39 | 0 | 0 |
| **Scorpaenidae** | | | | |
| *Gymnapistes marmoratus* | 1 | 0.39 | 0 | 0 |
| **Platycephalidae** | | | | |
| *Platycephalus bassensis* | 2 | 0.77 | 0 | 0 |
| **Triglidae** | | | | |
| *Lepidotrigla papilio* | 1 | 0.39 | 0 | 0 |
| **Arripidae** | | | | |
| *Arripis trutta* | 0 | 0 | 4 | 3.85 |
| **Sparidae** | | | | |
| *Acanthopagrus butcheri* | 0 | 0 | 0 | 0 |
| **Mugilidae** | | | | |
| *Aldrichetta forsteri* | 0 | 0 | 0 | 0 |
| **Leptoscopidae** | | | | |
| *Crapalatus munroi* | 0 | 0 | 0 | 0 |
| *Lesueurina platycephala* | 0 | 0 | 0 | 0 |
| **Bovichtidae** | | | | |
| *Pseudaphritis urvillii* | 1 | 0.39 | 0 | 0 |
| **Blenniidae** | | | | |
| *Parablennius tasmanianus* | 2 | 0.77 | 0 | 0 |
| **Clinidae** | | | | |
| *Heteroclinus perspicillatus* | 0 | 0 | 0 | 0 |
| **Gobiidae** | | | | |
| *Favonigobius tamarensis* | 0 | 0 | 0 | 0 |
| *Nesogobius hinsbyi* | 67 | 26.0 | 0 | 0 |
| *Nesogobius* sp.1 | 0 | 0 | 0 | 0 |
| *Pseudogobius olorum* | 0 | 0 | 0 | 0 |
| *Tasmanogobius lasti* | 5 | 1.93 | 0 | 0 |
| **Pleuronectidae** | | | | |
| *Ammotretis lituratus* | 0 | 0 | 0 | 0 |
| *Ammotretis rostratus* | 5 | 1.94 | 9 | 8.65 |
| *Rhombosolea tapirina* | 45 | 17.4 | 8 | 7.69 |
| *Taratretis derwentensis* | 0 | 0 | 0 | 0 |
| **Monocanthidae** | | | | |
| *A. spilomelanurus* | 1 | 0.39 | 0 | 0 |
| *Meuschenia freycineti* | 103 | 39.9 | 0 | 0 |
| **Tetraodontidae** | | | | |
| *Tetractenos glaber* | 0 | 0 | 0 | 0 |

Table 5.5.1 (Cont)

| | Frederick Henry Bay | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Location | Seven Mile Beach | | Lauderdale Bch. | | Tiger Head | | Marion Narrows | | Dunalley Bch. | |
| Species | n | % | n | % | n | % | n | % | n | % |
| **Rajidae** | | | | | | | | | | |
| *Raja lemprier* | 1 | 1.59 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Gobiesocidae** | | | | | | | | | | |
| *Alabes dorsalis* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.09 | 0 | 0 |
| **Atherinidae** | | | | | | | | | | |
| *Atherinosoma microstoma* | 2 | 3.17 | 1 | 0.66 | 14 | 1.40 | 16 | 1.44 | 0 | 0 |
| *Kestratherina esox* | 0 | 0 | 0 | 0 | 0 | 0 | 19 | 1.71 | 5 | 1.02 |
| *Leptatherina presbyteroides* | 5 | 7.94 | 4 | 2.65 | 706 | 70.8 | 59 | 5.32 | 272 | 55.4 |
| **Syngnathidae** | | | | | | | | | | |
| *Hippocampus breviceps* | 0 | 0 | 0 | 0 | 1 | 0.10 | 0 | 0 | 0 | 0 |
| *Mitotichthys semistriatus* | 0 | 0 | 0 | 0 | 2 | 0.20 | 0 | 0 | 0 | 0 |
| *Stigmatopora argus* | 0 | 0 | 0 | 0 | 1 | 0.10 | 0 | 0 | 0 | 0 |
| **Scorpaenidae** | | | | | | | | | | |
| *Gymnapistes marmoratus* | 0 | 0 | 0 | 0 | 4 | 0.40 | 0 | 0 | 0 | 0 |
| **Triglidae** | | | | | | | | | | |
| *Lepidotrigla mulhalli* | 0 | 0 | 2 | 1.32 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Arripidae** | | | | | | | | | | |
| *Arripis trutta* | 17 | 27.0 | 52 | 34.4 | 0 | 0 | 1 | 0.09 | 1 | 0.20 |
| **Mugilidae** | | | | | | | | | | |
| *Aldrichetta forsteri* | 5 | 7.94 | 56 | 37.1 | 0 | 0 | 11 | 0.99 | 124 | 25.2 |
| **Leptoscopidae** | | | | | | | | | | |
| *Crapalatus munroi* | 0 | 0 | 1 | 0.66 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Lesueurina platycephala* | 14 | 22.2 | 12 | 7.95 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Odacidae** | | | | | | | | | | |
| *Neoodax balteatus* | 0 | 0 | 0 | 0 | 14 | 1.40 | 21 | 1.89 | 0 | 0 |
| **Blenniidae** | | | | | | | | | | |
| *Parablennius tasmanianus* | 0 | 0 | 0 | 0 | 1 | 0.10 | 0 | 0 | 0 | 0 |
| **Clinidae** | | | | | | | | | | |
| *Cristaceps australis* | 0 | 0 | 0 | 0 | 1 | 0.10 | 0 | 0 | 0 | 0 |
| *Heteroclinus perspicillatus* | 0 | 0 | 0 | 0 | 9 | 0.90 | 2 | 0.18 | 0 | 0 |
| **Gobiidae** | | | | | | | | | | |
| *Favonigobius tamarensis* | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0.27 | 0 | 0 |
| *Nesogobius* sp.1 | 0 | 0 | 0 | 0 | 52 | 5.22 | 888 | 80.1 | 28 | 5.70 |
| **Pleuronectidae** | | | | | | | | | | |
| *Ammotretis lituratus* | 4 | 6.35 | 0 | 0 | 0 | 0 | 4 | 0.36 | 0 | 0 |
| *Amnotretis rostratus* | 0 | 0 | 2 | 1.32 | 0 | 0 | 6 | 0.54 | 0 | 0 |
| *Rhombosolea tapirina* | 2 | 3.17 | 14 | 9.27 | 20 | 2.01 | 77 | 6.94 | 61 | 12.4 |
| *Taratretis derwentensis* | 10 | 15.9 | 7 | 4.64 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Monocanthidae** | | | | | | | | | | |
| *Acanthaluteres vittiger* | 0 | 0 | 0 | 0 | 9 | 0.90 | 0 | 0 | 0 | 0 |
| *A. spilomelanurus* | 0 | 0 | 0 | 0 | 157 | 15.7 | 1 | 0.09 | 0 | 0 |
| *Brachaluteres jacksonianus* | 0 | 0 | 0 | 0 | 1 | 0.10 | 0 | 0 | 0 | 0 |
| *Meuschenia freycineti* | 0 | 0 | 0 | 0 | 5 | 0.50 | 0 | 0 | 0 | 0 |
| **Tetraodontidae** | | | | | | | | | | |
| *Contusus brevicaudus* | 1 | 1.59 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Contusus richei* | 2 | 3.17 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

Table 5.5.1 (Cont)

| | Tasman Peninsula | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Location | Carnarvon Bay | | Parsons Bay | | Safety Cove | | Stewarts Bay | | White Bch. | |
| Species | n | % | n | % | n | % | n | % | n | % |
| **Atherinidae** | | | | | | | | | | |
| *Kestratherina esox* | 0 | 0 | 1 | 0.27 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Leptatherina presbyteroides* | 0 | 0 | 46 | 12.3 | 0 | 0 | 8 | 2.93 | 67 | 28.9 |
| **Syngnathidae** | | | | | | | | | | |
| *Hippocampus abdominalis* | 2 | 0.44 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Stigmatopora argus* | 0 | 0 | 0 | 0 | 4 | 1.04 | 0 | 0 | 0 | 0 |
| *Mitotichthys mollisoni* | 1 | 0.22 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Mitotichthys semistriatus* | 1 | 0.22 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Scorpaenidae** | | | | | | | | | | |
| *Gymnapistes marmoratus* | 23 | 5.09 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Triglidae** | | | | | | | | | | |
| *Lepidotrigla mulhalli* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.37 | 0 | 0 |
| **Carangidae** | | | | | | | | | | |
| *Pseudocaranx dentex* | 0 | 0 | 0 | 0 | 0 | 0 | 10 | 3.66 | 0 | 0 |
| **Arripidae** | | | | | | | | | | |
| *Arripis trutta* | 79 | 17.5 | 6 | 1.61 | 194 | 50.5 | 182 | 66.7 | 78 | 33.6 |
| **Mugilidae** | | | | | | | | | | |
| *Aldrichetta forsteri* | 62 | 13.7 | 32 | 8.58 | 160 | 41.7 | 8 | 2.93 | 21 | 9.05 |
| **Platycephalidae** | | | | | | | | | | |
| *Platycephalus bassensis* | 1 | 0.22 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Odacidae** | | | | | | | | | | |
| *Neoodax balteatus* | 210 | 46.5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Leptoscopidae** | | | | | | | | | | |
| *Crapalatus munroi* | 0 | 0 | 0 | 0 | 1 | 0.26 | 0 | 0 | 0 | 0 |
| *Lesueurina platycephala* | 0 | 0 | 0 | 0 | 1 | 0.26 | 0 | 0 | 1 | 0.43 |
| **Clinidae** | | | | | | | | | | |
| *Cristaceps australis* | 1 | 0.22 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Heteroclinus puellarum* | 1 | 0.22 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Gobiidae** | | | | | | | | | | |
| *Nesogobius* sp.1 | 48 | 10.6 | 186 | 49.9 | 0 | 0 | 1 | 0.37 | 0 | 0 |
| **Pleuronectidae** | | | | | | | | | | |
| *Ammotretis lituratus* | 0 | 0 | 0 | 0 | 1 | 0.26 | 0 | 0 | 11 | 4.74 |
| *Ammotretis rostratus* | 15 | 3.32 | 88 | 23.6 | 6 | 1.56 | 28 | 10.3 | 52 | 22.4 |
| *Rhombosolea tapirina* | 7 | 1.55 | 13 | 3.49 | 17 | 4.43 | 35 | 12.8 | 2 | 0.86 |
| **Monocanthidae** | | | | | | | | | | |
| *A. spilomelanurus* | 1 | 0.22 | 1 | 0.27 | 0 | 0 | 0 | 0 | 0 | 0 |

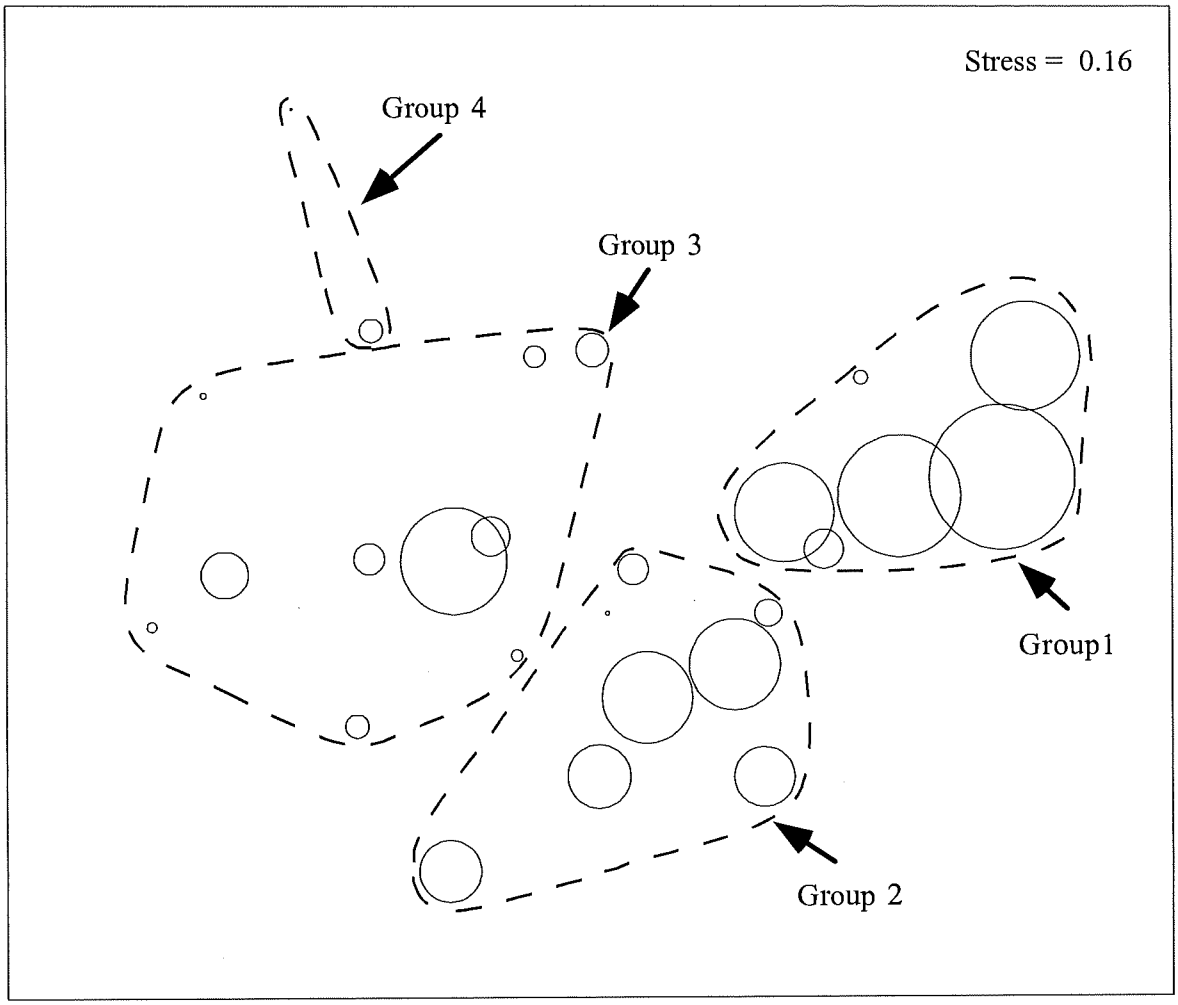



Fig. 5.5.1 Non-metric multi-dimensional scaling of fish communities sampled by beach seine at shallow beach sites throughout south-east Tasmania.

Table 5.5.2 Average abundance of ten species contributing most to the separation of groups from shallow beach sites throughout south-east Tasmania. per haul at each site.

| Species | Average abundance | | | |
|---|---|---|---|---|
| | Group 1 | Group 2 | Group 3 | Group 4 |
| *Nesogobius* sp.1 | 0 | 8.5 | 55.3 | 5.1 |
| *Atherinosoma microstoma* | 0.1 | 0 | 11.5 | 86.1 |
| *Leptatherina presbyteroides* | 11.5 | 66.6 | 43.6 | 1.3 |
| *Neoodax balteatus* | 0 | 0 | 9.7 | 0 |
| *Pseudaphritis urvillii* | 0 | 0.1 | .4 | 11.0 |
| *Aldrichetta forsteri* | 14.6 | 5.5 | 2.6 | 38.6 |
| *Arripis trutta* | 19.1 | 16.6 | 4.0 | .25 |
| *Rhombosolea taparina* | 2.3 | 7.9 | 19.6 | 6.0 |
| *Gymnapistes marmoratus* | 0 | 0.2 | 3.1 | 4.1 |
| *Ammotretis rostratus* | 3.8 | 8.8 | 3.9 | 0 |

## 5.6 Snapshot survey of Tasmanian inshore habitats

### 5.6.1 Introduction

Extensive sampling of coastal soft-sediment was reported in the previous chapters, however, most of this was concentrated in southern and eastern Tasmania, particularly within the three main survey areas of Norfolk Bay, Prosser Bay and Georges Bay. While these areas were chosen to represent a range of coastal environment types the significance of these areas can only be assessed after sampling of similar habitats on a broader spatial scale. Therefore, the aim of this component of the study is to provide some appreciation of the geographical variation of soft-bottom demersal fish assemblages throughout Tasmania and to evaluate the conclusions on the significance of such habitats as nursery areas for commercial species an a state wide scale.

### 5.6.2 Methods

Full details of sampling gear and regime, distribution of survey sites and site characteristics is presented in Chapter 4.2.5. In brief, beam trawl sampling was conducted at twenty-four unvegetated and seagrass sites in five regions around Tasmania (see Fig. 4.7, Table 4.4). Up to six sites representative of subtidal (2-8 m) soft-sediment habitats were sampled in each region, although deeper beds (up to 20 m) of *Posidonia australis* were sampled on Flinders Island and on the north-east coast.

### 5.6.3 Results and Discussion

Full details of species composition in each of the five survey regions is presented in Table 5.6.1 and a full list of scientific and common names is presented in Appendix 1. Throughout south-east Tasmania, both *Acanthaluteres spilomelanurus* and *Neoodax balteatus* were the most abundant species in the *Heterozostera* sites, while *Gymnapistes marmoratus* and *Stigmatopora argus* were abundant at two and three sites respectively. Few species were in common with unvegetated sites, which were dominated by *Nesogobius hinsbyi*, *Ammotretis rostratus*, *Rhombosolea tapirina* and *Platycephalus bassensis*.

In eastern Tasmania, abundances were low at unvegetated sites with *Cristaceps australis*, *Vanacampus phillipi* and *Crapulatus munroi* dominating. The two *Heterozostera* sites were dominated by different species, with *A. spilomelanurus* in highest abundance in Promise Bay and several species of gobies and sygnathids dominating in Little Swanport. Excluding the high abundance of atherinids, which show patchy distribution, the *Amphibolis* site was dominated by *Cristiceps australis*, *Acanthaluteres vittiger* and *Neoodax balteatus* in low numbers.

The species composition of unvegetated sites in north-east Tasmania was more variable, possibly reflecting the influence of different sediment and estuarine characteristics. The oceanic sand site at Tomahawk was dominated by *A. vittiger* and several species Odacidae,

while the mud site in Port Sorrell showed low abundances of *P. bassensis* and flounders. While *Acanthaluteres vittiger* and *A. spilomelanurus* were present in high abundances at *Posidonia* sites, *Dotolabrus aurantiacus* dominated the Waterhouse Island site. In contrast, sygnathids were the dominant group at the *Heterozostera* site.

All Flinders Island sites were primarily comprised of *Posidonia* beds, although small amounts of *Amphibolis* were also present at some sites. While there was some between site differences, the community was dominated by monocanthids (primarily *A. spilomelanurus* and *Meuschenia freycineti*) with small numbers of *N. balteatus* also present. The scorpaenid, *Gymnapistes marmoratus* was also highly abundant at one site.

Sites on the north-west coast were characterised by low abundances, particularly at unvegetated sites which had only small numbers of gobies and *Tetractenos glaber*. The community composition of both the *Heterozostera* and *Posidonia* sites were similar with monocanthids and sygnathids dominant.

In general, the community composition of habitats over this large spatial scale was similar to that found from the more detailed beam trawl sampling outlined in the previous chapters. In addition, because the temporal patterns described earlier revealed distinct seasonality primarily at seagrass sites, it is not valid to compare sites based on a single snapshot survey. However, this survey does indicate the range of abundances and community composition that occurs within and between habitats over a large geographic range.

Of particular note is that seagrass sites across this large spatial scale showed a similar trend to sites sampled in the main survey areas of Norfolk Bay, Prosser Bay, Georges Bay and the Tamar River. That is, seagrass beds do not play a particularly significant role as a nursery area for fish species throughout Tasmania. In addition, while juveniles of species of commercial and recreational importance, (*Ammotretis rostratus*, *Rhombosolea tapirina, Arripis trutta* and *Platycephalus bassensis*), were caught primarily at unvegetated sites, they made up a much lower proportion of the overall community than shallow unvegetated sites surveyed by beach seine (see Chapter 5.5).

Some of these differences may reflect the relative efficiency of the beam trawl, which varies depending on several factors including swimming speed, habit and escape behaviour when compared with beach seine. Faster moving pelagic species are not captured reliably by beam trawl (Gray and Bell 1986), and consequently results from beam trawl samples reflect relative abundances of slower moving demersal species. While this may explain the low abundances of *A. trutta* in unvegetated sites this survey, further work is needed to examine the depth distribution of juveniles of this species. In contrast, flounder larvae have been shown to recruit to shallow (~1 m deep) sheltered sandflats indicating deeper subtidal areas are of less importance as a nursery habitat for these species (Crawford 1984).

A detailed examination of the habitat preference of juvenile *P. bassensis* is presented in Chapter 5.9.

Table 5.6.1 Total number of individuals and percentage of total individuals for the fish taxa collected using the beam trawl from soft-sediment sites in five regions around Tasmania.

| | South East Coast | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Location | Cloudy Lagoon | | Simpsons Bay | | Dru Pt. | | Trial Bay | | Lime Bay | | | |
| Habitat | *H. tasmanica* | | Sand | | *H. tasmanica* | | *H. tasmanica* | | *H. tasmanica* | | Sand | |
| Species | n | % | n | % | n | % | n | % | n | % | n | % |
| **Urolophidae** | | | | | | | | | | | | |
| *Urolophus cruciatus* | 0 | 0 | 2 | 0.86 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Gobiesocidae** | | | | | | | | | | | | |
| *Alabes dorsalis* | 0 | 0 | 1 | 0.43 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Sygnathidae** | | | | | | | | | | | | |
| *Mitotichthys semistriatus* | 0 | 0 | 0 | 0 | 1 | 0.41 | 0 | 0 | 2 | 0.24 | 0 | 0 |
| *Stigmatopora argus* | 20 | 12.2 | 18 | 7.76 | 52 | 21.5 | 44 | 10.1 | 0 | 0 | 0 | 0 |
| *Stigmatopora nigra* | 0 | 0 | 0 | 0 | 9 | 3.72 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Vanacampus phillipi* | 1 | 0.61 | 2 | 0.86 | 13 | 5.37 | 2 | 0.46 | 6 | 0.73 | 0 | 0 |
| **Scorpaenidae** | | | | | | | | | | | | |
| *Gymnapistes marmoratus* | 32 | 19.5 | 0 | 0 | 37 | 15.3 | 0 | 0 | 1 | 0.12 | 0 | 0 |
| **Triglidae** | | | | | | | | | | | | |
| *Lepidotrigla papilio* | 0 | 0 | 0 | 0 | 1 | 0.41 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Platycephalidae** | | | | | | | | | | | | |
| *Platycephalus bassensis* | 1 | 0.61 | 13 | 5.6 | 12 | 4.96 | 0 | 0 | 0 | 0 | 2 | 16.7 |
| **Apogonidae** | | | | | | | | | | | | |
| *Vincentia conspersa* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.23 | 12 | 1.45 | 0 | 0 |
| **Odacidae** | | | | | | | | | | | | |
| *Neoodax balteatus* | 92 | 56.1 | 27 | 11.6 | 40 | 16.5 | 143 | 32.9 | 255 | 30.9 | 1 | 8.33 |
| **Bovichtidae** | | | | | | | | | | | | |
| *Pseudaphritis urvillii* | 1 | 0.61 | 0 | 0 | 1 | 0.41 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Clinidae** | | | | | | | | | | | | |
| *Cristaceps australis* | 1 | 0.61 | 1 | 0.43 | 2 | 0.83 | 2 | 0.46 | 3 | 0.36 | 0 | 0 |
| *Heteroclinus perspicillatus* | 0 | 0 | 1 | 0.43 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Gobiidae** | | | | | | | | | | | | |
| *Favonigobius tamarensis* | 0 | 0 | 0 | 0 | 4 | 1.65 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nesogobius hinsbyi* | 0 | 0 | 101 | 43.5 | 12 | 4.96 | 10 | 2.3 | 0 | 0 | 2 | 16.7 |
| *Nesogobius pulchellus* | 3 | 1.83 | 0 | 0 | 1 | 0.41 | 2 | 0.46 | 1 | 0.12 | 0 | 0 |
| *Pseudogobius olorum* | 0 | 0 | 1 | 0.43 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nesogobius* sp.1 | 0 | 0 | 5 | 2.16 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nesogobius* sp.3 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.23 | 0 | 0 | 0 | 0 |
| *Tasmanogobius lasti* | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.46 | 0 | 0 | 0 | 0 |
| **Pleuronectidae** | | | | | | | | | | | | |
| *Ammotretis rostratus* | 0 | 0 | 0 | 0 | 3 | 1.24 | 0 | 0 | 0 | 0 | 3 | 25 |
| *Rhombosolea tapirina* | 1 | 0.61 | 1 | 0.43 | 1 | 0.41 | 1 | 0.23 | 0 | 0 | 3 | 25 |
| **Monocanthidae** | | | | | | | | | | | | |
| *Brachaluteres jacksonianus* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.23 | 0 | 0 | 0 | 0 |
| *Meuschenia freycineti* | 0 | 0 | 1 | 0.43 | 1 | 0.41 | 2 | 0.46 | 1 | .12 | 0 | 0 |
| *Acanthaluteres spilomelanurus* | 12 | 7.32 | 58 | 25 | 52 | 21.5 | 223 | 51.4 | 516 | 62.5 | 0 | 0 |

Table 5.6.1 (cont).

| | East Coast | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Location | Promise Bay | | | | Booming Bay | | Little Swanport | |
| Habitat | Sand | | *H. tasmanica* | | *A. antarctica* | | *H. tasmanica* | |
| Species | n | % | n | % | n | % | n | % |
| **Urolophidae** | | | | | | | | |
| *Urolophus paucimaculatus* | 0 | 0 | 1 | 0.8 | 0 | 0 | 0 | 0 |
| **Atherinidae** | | | | | | | | |
| *Atherinason hepsetoides* | 0 | 0 | 0 | 0 | 38 | 71.7 | 0 | 0 |
| **Sygnathidae** | | | | | | | | |
| *Stigmatopora argus* | 1 | 8.33 | 6 | 4.8 | 0 | 0 | 1 | 1.89 |
| *Stigmatopora nigra* | 0 | 0 | 4 | 3.2 | 0 | 0 | 8 | 15.1 |
| *Vanacampus phillipi* | 2 | 16.7 | 3 | 2.4 | 0 | 0 | 1 | 1.89 |
| **Scorpaenidae** | | | | | | | | |
| *Scorpaena papillosa* | 1 | 8.33 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Gymnapistes marmoratus* | 0 | 0 | 2 | 1.6 | 0 | 0 | 9 | 17 |
| **Platycephalidae** | | | | | | | | |
| *Platycephalus bassensis* | 1 | 8.33 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Apogonidae** | | | | | | | | |
| *Siphaemia cephalotes* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Vincentia conspersa* | 0 | 0 | 1 | 0.8 | 0 | 0 | 0 | 0 |
| **Labridae** | | | | | | | | |
| *Pseudolabrus psittaculus* | 0 | 0 | 0 | 0 | 1 | 1.89 | 0 | 0 |
| **Odacidae** | | | | | | | | |
| *Siphonognathus attenuatus* | 0 | 0 | 3 | 2.4 | 0 | 0 | 0 | 0 |
| *Neoodax balteatus* | 1 | 8.33 | 10 | 8 | 3 | 5.66 | 0 | 0 |
| **Leptoscopidae** | | | | | | | | |
| *Crapulatus munroi* | 2 | 16.7 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Bovichtidae** | | | | | | | | |
| *Pseudaphritis urvillii* | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 3.77 |
| **Clinidae** | | | | | | | | |
| *Cristaceps australis* | 2 | 16.7 | 6 | 4.8 | 4 | 7.55 | 0 | 0 |
| *Heteroclinus puellarum* | 0 | 0 | 0 | 0 | 1 | 1.89 | 0 | 0 |
| **Gobiidae** | | | | | | | | |
| *Arenigobius bifrenatus* | 0 | 0 | 0 | 0 | 0 | 0 | 16 | 30.2 |
| *Pseudogobius olorum* | 0 | 0 | 0 | 0 | 0 | 0 | 16 | 30.2 |
| *Tasmanogobius lasti* | 0 | 0 | 2 | 1.6 | 0 | 0 | 0 | 0 |
| **Bothidae** | | | | | | | | |
| *Pseudorhombus jenynsii* | 1 | 8.33 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Pleuronectidae** | | | | | | | | |
| *Ammotretis rostratus* | 0 | 0 | 0 | 0 | 1 | 1.89 | 0 | 0 |
| **Monocanthidae** | | | | | | | | |
| *Acanthaluteres vittiger* | 0 | 0 | 0 | 0 | 3 | 5.66 | 0 | 0 |
| *A. spilomelanurus* | 1 | 8.33 | 84 | 67.2 | 2 | 3.77 | 0 | 0 |
| *Meuschenia freycineti* | 0 | 0 | 2 | 1.6 | 0 | 0 | 0 | 0 |

Table 5.6.1 (cont).

| | North East Coast | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Location | Tomahawk | | Waterhouse Is. | | Little Musselroe Bay | | Port Sorell | | | |
| Habitat | Sand | | *P. australis* | | *P. australis* | | *H. tasmanica* | | Mud | |
| Species | n | % | n | % | n | % | n | % | n | % |
| **Atherinidae** | | | | | | | | | | |
| *Atherinason hepsetoides* | 2 | 4.55 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Sygnathidae** | | | | | | | | | | |
| *Phyllopteryx taeniolatus* | 0 | 0 | 1 | 1.04 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Hippocampus abdominalis* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.36 | 0 | 0 |
| *Histiogamphelus briggsii* | 2 | 4.55 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Stigmatopora argus* | 1 | 2.27 | 0 | 0 | 2 | 2.94 | 0 | 0 | 0 | 0 |
| *Stigmatopora nigra* | 0 | 0 | 0 | 0 | 0 | 0 | 162 | 57.7 | 1 | 20.0 |
| *Vanacampus phillipi* | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 1.78 | 0 | 0 |
| *Vanacampus poecilolaemus* | 0 | 0 | 1 | 1.04 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Scorpaenidae** | | | | | | | | | | |
| *Scorpaena papillosa* | 0 | 0 | 3 | 3.13 | 0 | 0 | 3 | 1.07 | 0 | 0 |
| **Platycephalidae** | | | | | | | | | | |
| *Platycephalus bassensis* | 0 | 0 | 0 | 0 | 0 | 0 | 11 | 3.91 | 2 | 40.0 |
| **Labridae** | | | | | | | | | | |
| *Dotalabrus aurantiacus* | 8 | 18.2 | 44 | 45.8 | 1 | 1.47 | 0 | 0 | 0 | 0 |
| **Odacidae** | | | | | | | | | | |
| *Haletta semifasciata* | 0 | 0 | 1 | 1.04 | 1 | 1.47 | 0 | 0 | 0 | 0 |
| *Siphonognathus attenuatus* | 1 | 2.27 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Neoodax balteatus* | 1 | 2.27 | 2 | 2.08 | 0 | 0 | 3 | 1.07 | 0 | 0 |
| *Siphonognathus beddomei* | 1 | 2.27 | 1 | 1.04 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Siphonognathus radiatus* | 4 | 9.09 | 7 | 7.29 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Leptoscopidae** | | | | | | | | | | |
| *Crapulatus munroi* | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 1.78 | 0 | 0 |
| **Clinidae** | | | | | | | | | | |
| *Cristaceps australis* | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 1.78 | 0 | 0 |
| *Heteroclinus adelaidae* | 0 | 0 | 5 | 5.21 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Heteroclinus tristis* | 0 | 0 | 1 | 1.04 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Heteroclinus perspicillatus* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.36 | 0 | 0 |
| **Gobiidae** | | | | | | | | | | |
| *Nesogobius* sp.1 | 0 | 0 | 0 | 0 | 0 | 0 | 9 | 3.2 | 0 | 0 |
| **Pleuronectidae** | | | | | | | | | | |
| *Ammotretis rostratus* | 0 | 0 | 0 | 0 | 0 | 0 | 9 | 3.2 | 1 | 20.0 |
| *Rhombosolea tapirina* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.36 | 1 | 20.0 |
| **Monocanthidae** | | | | | | | | | | |
| *Acanthaluteres vittiger* | 21 | 47.7 | 26 | 27.1 | 15 | 22.1 | 64 | 22.8 | 0 | 0 |
| *A. spilomelanurus* | 3 | 6.82 | 4 | 4.17 | 49 | 72.1 | 1 | 0.36 | 0 | 0 |
| **Tetraodontidae** | | | | | | | | | | |
| *Tetractenos glaber* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.36 | 0 | 0 |

Table 5.6.1 (cont).

| | Flinders Island Area | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Location | Franklin Sound A | | Franklin Sound F | | Prime Seal Island | | Kent Bay | |
| Habitat | *P. australis* | | *P. australis* | | *P. australis* | | *P. australis* | |
| Species | n | % | n | % | n | % | n | % |
| **Ophidiidae** | | | | | | | | |
| *Genypterus tigerinus* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.37 |
| **Sygnathidae** | | | | | | | | |
| *Hypselognathus rostratus* | 2 | 2.08 | 3 | 0.51 | 0 | 0 | 0 | 0 |
| *Leptoichthys fistularius* | 0 | 0 | 0 | 0 | 1 | 0.7 | 0 | 0 |
| *Stigmatopora argus* | 3 | 3.13 | 1 | 0.17 | 2 | 1.4 | 4 | 1.49 |
| *Vanacampus poecilolaemus* | 0 | 0 | 0 | 0 | 2 | 1.4 | 0 | 0 |
| **Scorpaenidae** | | | | | | | | |
| *Neosebastes thetidis* | 0 | 0 | 0 | 0 | 2 | 1.4 | 0 | 0 |
| *Scorpaena papillosa* | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.75 |
| *Gymnapistes marmoratus* | 1 | 1.04 | 238 | 40.1 | 7 | 4.9 | 1 | 0.37 |
| **Aploactinidae** | | | | | | | | |
| *Aploactisoma milesii* | 0 | 0 | 0 | 0 | 1 | 0.7 | 0 | 0 |
| **Apogonidae** | | | | | | | | |
| *Siphaemia cephalotes* | 5 | 5.21 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Vincentia conspersa* | 0 | 0 | 2 | 0.34 | 0 | 0 | 41 | 15.3 |
| **Mullidae** | | | | | | | | |
| *Upeneichthys vlamingii* | 0 | 0 | 0 | 0 | 1 | 0.7 | 0 | 0 |
| **Labridae** | | | | | | | | |
| *Dotalabrus aurantiacus* | 0 | 0 | 0 | 0 | 8 | 5.59 | 0 | 0 |
| **Odacidae** | | | | | | | | |
| *Haletta semifasciata* | 0 | 0 | 29 | 4.88 | 0 | 0 | 11 | 4.1 |
| *Siphonognathus attenuatus* | 0 | 0 | 0 | 0 | 2 | 1.4 | 0 | 0 |
| *Neoodax balteatus* | 9 | 9.38 | 62 | 10.4 | 2 | 1.4 | 14 | 5.22 |
| *Siphonognathus beddomei* | 0 | 0 | 0 | 0 | 2 | 1.4 | 0 | 0 |
| *Siphonognathus radiatus* | 1 | 1.04 | 0 | 0 | 11 | 7.69 | 2 | 0.75 |
| **Clinidae** | | | | | | | | |
| *Cristaceps australis* | 0 | 0 | 1 | 0.17 | 1 | 0.7 | 1 | 0.37 |
| **Monocanthidae** | | | | | | | | |
| *Acanthaluteres vittiger* | 0 | 0 | 0 | 0 | 3 | 2.1 | 6 | 2.24 |
| *A. spilomelanurus* | 68 | 70.8 | 247 | 41.6 | 97 | 67.8 | 183 | 68.3 |
| *Meuschenia scaber* | 0 | 0 | 2 | 0.34 | 0 | 0 | 0 | 0 |
| *Scobinichthys granulatus* | 0 | 0 | 0 | 0 | 1 | 0.7 | 0 | 0 |
| *Meuschenia freycineti* | 7 | 7.29 | 8 | 1.35 | 0 | 0 | 2 | 0.75 |
| **Tetraodontidae** | | | | | | | | |
| *Contusus brevicaudus* | 0 | 0 | 1 | 0.17 | 0 | 0 | 0 | 0 |

Table 5.6.1 (cont).

| | North West Coast | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Location | Robbins Island | | | | Stanley Beach | | | | West Inlet | |
| Habitat | Sand | | *P. australis* | | Sand | | Sand | | *H. tasmanica* | |
| Species | n | % | n | % | n | % | n | % | n | % |
| **Sygnathidae** | | | | | | | | | | |
| *Mitotichthys semistriatus* | 0 | 0 | 1 | 2.38 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Stigmatopora argus* | 0 | 0 | 2 | 4.76 | 0 | 0 | 0 | 0 | 74 | 17.5 |
| *Stigmatopora nigra* | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 12 | 9 | 2.13 |
| *Vanacampus phillipi* | 0 | 0 | 3 | 7.14 | 0 | 0 | 0 | 0 | 7 | 1.66 |
| *Hippocampus breviceps* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.47 |
| **Scorpaenidae** | | | | | | | | | | |
| *Scorpaena papillosa* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.24 |
| *Gymnapistes marmoratus* | 0 | 0 | 6 | 14.3 | 0 | 0 | 0 | 0 | 1 | 0.24 |
| **Mullidae** | | | | | | | | | | |
| *Upeneichthys vlamingii* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.24 |
| **Odacidae** | | | | | | | | | | |
| *Neoodax balteatus* | 0 | 0 | 7 | 16.7 | 0 | 0 | 0 | 0 | 6 | 1.42 |
| **Clinidae** | | | | | | | | | | |
| *Cristaceps australis* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.47 |
| *Heteroclinus perspicillatus* | 0 | 0 | 1 | 2.38 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Gobiidae** | | | | | | | | | | |
| *Nesogobius pulchellus* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.24 |
| *Nesogobius* sp.1 | 1 | 100 | 1 | 2.38 | 0 | 0 | 10 | 40 | 11 | 2.61 |
| *Nesogobius* sp.6 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 4 | 0 | 0 |
| **Pleuronectidae** | | | | | | | | | | |
| *Ammotretis rostratus* | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 12 | 0 | 0 |
| *Rhombosolea tapirina* | 0 | 0 | 0 | 0 | 0 | 0 | 8 | 32 | 1 | 0.24 |
| *Taratretis derwentensis* | 0 | 0 | 0 | 0 | 1 | 5.26 | 0 | 0 | 0 | 0 |
| **Monocanthidae** | | | | | | | | | | |
| *Acanthaluteres vittiger* | 0 | 0 | 2 | 4.76 | 0 | 0 | 0 | 0 | 52 | 12.3 |
| *A. spilomelanurus* | 0 | 0 | 17 | 40.5 | 0 | 0 | 0 | 0 | 246 | 58.3 |
| *Eubalichthys gunnii* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0.71 |
| *Meuschenia freycineti* | 0 | 0 | 1 | 2.38 | 0 | 0 | 0 | 0 | 2 | 0.47 |
| **Tetraodontidae** | | | | | | | | | | |
| *Tetractenos glaber* | 0 | 0 | 1 | 2.38 | 18 | 94.7 | 0 | 0 | 1 | 0.24 |
| **Diodontidae** | | | | | | | | | | |
| *Diodon nicthemerus* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.47 |

## 5.7 Tasmanian seagrass distribution

Survey sites in the present project were selected to cover a wide range of seagrass and unvegetated habitats representative of different coastal environment types throughout Tasmania. During the course of this project, particularly during the snapshot survey (Chapter 5.6), it became clear that there are considerable areas of unmapped seagrass beds along the north coast of Tasmania. Previous mapping of seagrass beds around Tasmania estimated the area of habitat to be around 500 $km^2$ (Rees 1993). However, substantial areas along the north coast were not mapped due to the limited coverage of suitable aerial photographs.

Due to the limitations of the study by Rees (1993), some limited mapping of seagrass beds was undertaken in the present project to update the Tasmania seagrass distributions. Around 530 $km^2$ of previously undocumented seagrass beds were identified along the north coast of Tasmania (particularly Musselroe Bay and Ringarooma Bay), and on the western shore of Flinders Island (Fig. 5.7.1). These beds were dominated by single species stands of *Posidonia australis,* although many beds consisted of a mixture of *P. australis*, *Amphibolis antarctica* and a range of species of macroalgae. Of particular interest was that adjacent to Flinders Island extensive beds of *P. australis* occurred in depths of up to 20 m. While this was the limit of surveying, the high standing stock of *P. australis* in 20 m suggests that the beds extended into even deeper water. Also of note was the presence of *Posidonia angustifolia* in the Flinders Island beds, extending the eastern boundary of its distribution from South Australia.

Broad scale mapping of marine habitats is currently being conducted around Tasmania in relation to a regional classification of Tasmanian coastal waters. The objective of the project is to use Landsat TM imagery and aerial photography to systematically map and ground truth shallow subtidal habitats around Tasmania and enter these data on a GIS. The physical aspects and vegetation of Tasmania's inshore marine environment are being classified into 8 categories and processed to produce maps of 1:100,000 scale showing distribution of selected substrate types and associated dominant vegetation types.

As of June 1998, all Tasmanian coastal waters have been ground truthed from the FRV *Challenger,* except the North Coast from Musselroe Bay to Stanley. The North Coast is planned to be mapped during late 1998. Satellite imagery and aerial photographs have been assessed for the west coast, King Island, Flinders Island and the east coast, and maps have been produced and digitised. Electronic copies of the maps are available in ArcInfo format from the Tasmanian Department of Environment and Land Management. They are to be transferred to ERIN's Blue Pages at a later date. However, while this project is conducting broad scale mapping, it is evident that there is clearly a lack of information available on the distribution of seagrass habitats at the appropriate spatial scale for effective management and monitoring.

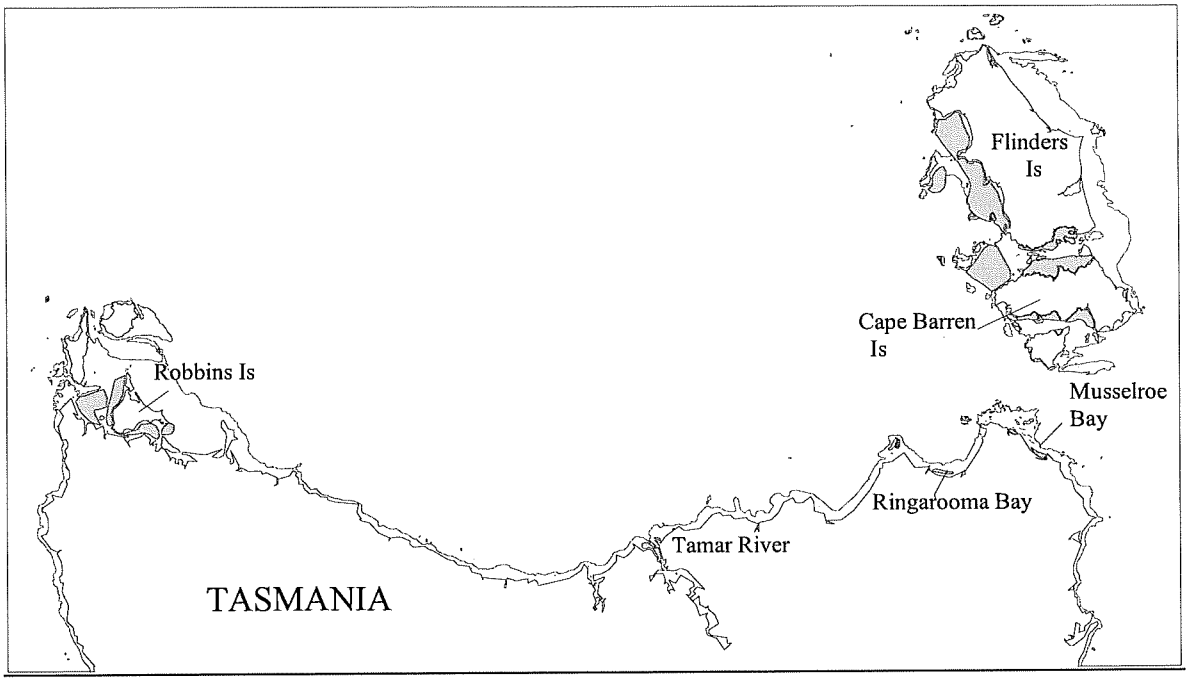



Fig. 5.7.1 Distribution of seagrass beds (green shading) (consisting primarily of *Posidonia australis* and *Amphibolis antarctica)* around the north coast of Tasmania. Blue contour line presents the 20 m depth contour.

## 5.8 General Discussion - Community composition

The primary objective of this component of the study was to examine the both community composition and habitat associations of different life-history stages of economically important scalefish species in seagrass and unvegetated habitats throughout Tasmania. The main survey areas of Norfolk Bay, Georges Bay and Prosser Bay were sampled in order to examine the significance of subtidal beds of *Heterozostera* and unvegetated areas at a range of spatial and temporal scales throughout Tasmania. Such information is necessary in order to provide information at a spatial scale relevant to management of such habitats throughout the state.

Firstly, the finding that fish communities in *Heterozostera* beds are more similar within an estuary than between estuaries, with all areas showing a high number of unique species clearly indicates that management of such habitats should be at the scale of individual estuaries, as each estuary can be described as having some unique community. In addition, management of such habitats is required at this large spatial scale if preservation of rarer species is of high importance.

However, detailed sampling in Norfolk Bay also found that *Heterozostera* sites within a single bay showed differences in abundance and community composition despite sites having similar physical characteristics. It appears that such differences may be related to variations in habitat structure between sites and/or patchiness in larval supply. As abundances were highest in the densest seagrass beds it appears that active selection may take place for such beds. This is consistent with previous studies where habitat structure was found to be an important factor influencing the spatial and temporal patterns of recruitment of fish into nearshore soft-sediment habitats (Orth *et al.* 1984, Bell *et al.* 1988). However, larval supply has also been shown to influence recruitment patterns of seagrass associated fish (Bell and Westoby 1986b, Bell *et al.* 1988, Jenkins *et al.* 1996). The fact that the dominant seagrass species in Norfolk Bay are resident species that spawn widely within the bay suggests that settlement patterns may be less influenced by the current regimes. Detailed studies of the hydrodynamics of such coastal bays will be required before the significance of each factor can be further examined.

The fact that there was a consistent relationship between fish abundances and seagrass density has implications for the management of such habitats. In order to maximise abundance and diversity of seagrass associated fish species, a high priority should be directed at areas with densest seagrass. In addition, it appears that in areas of low seagrass density like Prosser Bay, the seasonal presence of seagrass is in itself either not sufficient to attract high numbers of fish, post-settlement mortality is high or abundances of pre-recruits is low. As few seagrass beds exist for some tens of kilometres from the beds in Prosser Bay it is possible that larval supply may be limiting. If this is so, declines in

density in isolated seagrass beds may result in a consistent decrease in abundance of seagrass associated species.

The temporal trends in the community dynamics of fish populations in *Heterozostera* habitats in south-east Tasmania vary considerably from seagrass habitats elsewhere in temperate Australia. Fish abundances peaked in winter, apparently related to winter die-back of seagrass beds reducing overall area of available habitat. It is therefore important to minimise impacts on seagrass beds throughout the year, as the seagrass that remain present throughout the colder months are particularly important for sustaining the resident populations.

A comparison between *Heterozostera* and *Posidonia* habitats in the Tamar River found no difference in the abundance or number of fish species in each habitat type. However, distinctly different fish communities were present in each habitat, with a large number of species unique to each seagrass species. The significance of this finding is that while both species of seagrass may be present in a single estuary, each species should be managed as discrete habitats and management actions should be directed at minimising impacts on both habitat types. In addition, no economically important species were found to be utilising either *Heterozostera* or *Posidonia* beds in the Tamar River as a nursery area.

In the comparison between *Heterozostera* beds and subtidal unvegetated habitats, consistently higher fish abundances and diversity were present at seagrass sites, with a large number of species found only in one of the habitat types. The majority of fish within both habitats were small demersal resident species, caught regularly as juveniles and adults in beam trawl samples. In addition, larger resident species that occurred in beam-trawl samples as juveniles and gillnet samples as adults made up a far greater proportion of the catch in unvegetated habitats (31%) than in *Heterozostera* beds (1.5%).

The dominance of a particular fish species in each habitat type is closely related to their morphology and behaviour, with many species in *Heterozostera* beds cryptic and associated closely with, and in some cases mimicking the seagrass blades. Dorso-ventral flattening, a close association with the substratum, and sand coloration are common characteristics of families abundant in beam trawl catches from unvegetated habitats (Gobidae, Pleuronectidae, Platycephalidae). While such differences in the dominant demersal species resulted in distinct communities in each habitat, such differences were not consistent in areas of marginal seagrass habitat such as Prosser Bay. In addition, a distinct habitat preference becomes less consistent with increasing size, with larger mobile fish often caught over both habitat types.

While *Heterozostera* beds throughout southern and eastern Tasmania were found to be an important habitat for production of small, resident fishes, they were not an important nursery habitat for commercial and recreational species. This observation was also supported by the results of the beach seine survey of shallow intertidal habitats where

species of commercial and recreational importance (*A. rostratus*, *A. lituratus*, *R. tapirina*, *A. trutta*, *Sillago flindersi*, *Platycephalus bassensis* and *A. forsteri*) dominated the overall catch at most unvegetated sites compared to seagrass sites. This contrasts the majority of studies throughout temperate Australia that have found seagrass beds to be important nursery areas for larger fish species, with larvae recruiting to seagrass beds, and juveniles or adults then moving into deeper water habitats (see Bell and Pollard 1989). While unvegetated habitats were found to be a more important nursery area for juveniles of economically important species than seagrass beds, management should be directed as minimising impacts on both habitats throughout the coastal zone. This is particularly important as seagrass beds do provide high levels of primary and secondary productivity in coastal waters, with most seagrass production not utilised *in situ* but exported from the beds.

It is also evident that there is clearly a lack of information available on the full extent of seagrass habitats throughout Tasmania. Around 530 $km^2$ of previously undocumented seagrass beds were identified along the north coast of Tasmania and on the western shore of Flinders Island. These beds were dominated by single species stands of *Posidonia australis,* although many beds consisted of a mixture of *P. australis*, *Amphibolis antarctica* and a range of species of macroalgae. This additional area indicates that around 1000 $km^2$ of seagrass beds are present around the state, close to twice the area in New South Wales and Victoria combined. Further detailed mapping at a range of spatial scales is required throughout the state for effective management and monitoring of such habitats.

# Commercial species

## 5.9 Sand flathead (*Platycephalus bassensis*)

### 5.9.1 Introduction

*Platycephalus bassensis* is found from the central coast of New South Wales to eastern South Australia, but is most common in southern New South Wales, Victoria and Tasmania (Gomon *et al.* 1994). The species occurs on sandy and muddy substrates down to 100 m, but are most common in shallow coastal waters less than 65 m. Commercial fishing for *P. bassensis* occurs in eastern Bass Strait, several Victorian bays and inlets, and around Tasmania. Catch records from these areas since the early 1960's show a reasonably stable level of landings averaging about 400 tonnes year$^{-1}$ (Kailola *et al.* 1993). Recent annual commercial landings from Victoria are around 30 tonnes (Neira *et al.* 1997), with a significant landings also being taken by recreational fishers (Hall and MacDonald 1986).

The commercial catch of *P. bassensis* in Tasmanian waters is unknown, as both *P. bassensis* and *Neoplatycephalus richardsoni* are pooled on commercial catch records. Up until the late 1980's the total flathead catch ranged between 20 and 50 tonnes; however since 1990 landings have been relatively stable at around 120 tonnes (Jordan 1994a). In Tasmania, *P. bassensis* are caught mainly by otter trawling and Danish seining in open coastal waters, while small landings are made by gill nets in more sheltered waters. The commercial fishing of flathead in Tasmanian state waters occurs relatively evenly throughout the east and south-east coasts with highest catches in late spring to autumn. In addition, *P. bassensis* are the most important recreational species in the state, with targeting by anglers mainly during summer (Lyle and Smith 1998).

Most species of platycephalids common in temperate estuarine and coastal waters of Australia show a preference for unvegetated habitats (Gomon *et al* 1994, Edgar and Shaw 1995a). The lack of studies detailing patterns of abundance and distribution in these waters in some way reflects the emphasis on studies of fish communities associated with seagrasses (Young 1981, Middleton *et al.* 1984, Bell *et al.* 1992, Ferrell *et al.* 1993), the analysis of patterns at the community level (Potter and Hyndes 1994, Ayvazian and Hyndes 1995), and low abundances of individual species (Gray *et al.* 1990, Edgar and Shaw 1995a, Gray *et al.* 1996, Jenkins *et al.* 1997). In addition, there is little published information on size compositions of platycephalids across their distribution.

Edgar and Shaw (1995a) found spatial variations in the abundance of *P. bassensis* in Western Port, Victoria, which they attributed to habitat type. In addition, *P. bassensis* showed no indication of a ontogenetic change in habitat preference in Western Port, with both juveniles and adults preferring unvegetated habitats (Edgar and Shaw 1995a). A similar habitat preference was noted for *P. bassensis* in coastal waters of Tasmania (Last 1983). However, both Last (1983) and Edgar and Shaw (1995a) primarily sampled

shallow (<3 m) habitats, with only limited sampling of deeper subtidal unvegetated and seagrass habitats. Such deeper habitats form a substantial part of the coastal region of Tasmania, with beds of *Heterozostera tasmanica* common in depths down to 7 m and unvegetated habitats dominant in marine embayments and estuaries.

The only previous ageing study on *P. bassensis* estimated maximum ages from whole sagittae to be 7 years for males and 9 years for females (Brown 1978). With the population dominated by two distinct modes, the first representing 2 year old fish and the second, 3 and 4 year olds. No studies have detailed the age, growth and age composition of *P. bassensis* in Tasmanian waters. In addition, while *P. bassensis* are known to spawn in Port Phillip Bay (Brown 1978), the lack of detailed early life-history studies on platycephalids has resulted in little information on patterns of spawning and larval distribution throughout their range.

Given the lack of information on the reproductive biology, early life-history, habitat preference, size composition and age and growth of *P. bassensis* from Tasmanian waters, the aim of this chapter is to (1) estimate the size at sexual maturity for male and female *P. bassensis*, (2) describe the pattern of gonadal and larval development, (3) examine spatial and temporal patterns of adults and juveniles, (4) determine the validated age and describe the growth of *P. bassensis* from sectioned sagittal otoliths, and (5) examine the interannual and spatial trends in age composition from inshore waters of southern and eastern Tasmania.

## 5.9.2 Methods

### *5.9.2.1 Survey area and sampling regime*

*Platycephalus bassensis* were routinely sampled from three inshore areas in eastern and southern Tasmania (Norfolk Bay, Georges Bay and Prosser Bay). Full details of sampling gear and regime are presented in Chapter 4. In brief, in each area, sites in the 1-12m depth range were chosen to be representative of soft-sediment unvegetated (mud and sand) and seagrass habitats (Table 5.9.1). The abundance of *P. bassensis* was analysed from six sites in Norfolk Bay sampled every two months from February 1995 to December 1996 and four sites in Georges Bay and two in Prosser Bay seasonally from February 1995 to October 1995. At each site, three non-overlapping 3-min beam trawls were conducted at a tow speed of 2 knots. In addition, the abundance of larger *P. bassensis* were analysed from a single seagrass and unvegetated site in each area where two multi-panel gillnets were set overnight on each sampling occasion. Catch rates were calculated as the number of fish per tow for beam trawls and number of fish per hour for gillnets. In addition, *P. bassensis* were sampled monthly from nearshore beach habitats between December 1996 and February 1997 at 26 sites throughout south-eastern Tasmania. The distribution of nearshore beach sampling sites, sampling regime and site characteristics are detailed in Chapter 4.2.4.

Table 5.9.1 Site and habitat characteristics of sites sampled in Norfolk Bay, Georges Bay and Prosser Bay

| Area/Site | Habitat | Seagrass Density | Depth (m) | Gear deployed |
|---|---|---|---|---|
| **Norfolk Bay** | | | | |
| Cascade Bay | Mud | | 8 - 12 | BT, GN |
| Prices Bay | *H. tasmanica* | Medium | 3 - 6 | BT |
| Prices Bay | Mud | | 8 - 12 | BT |
| Lime Bay | *H. tasmanica* | Medium | 3 - 6 | BT, GN |
| Lime Bay | Sand | | 1 - 3 | BT |
| Smooth Island | *H. tasmanica* | Low | 4 - 6 | BT |
| **Georges Bay** | | | | |
| Steiglitz Beach | *H. tasmanica* | High | 2 - 5 | BT, GN |
| McDonalds Pt. | Mud | | 8 - 12 | BT |
| Moulting Bay Nth | Mud | | 3 - 5 | BT, GN |
| Moulting Bay SW. | *H. tasmanica* | Low | 2 - 4 | BT |
| **Prosser Bay** | | | | |
| Paddys Point | *H. tasmanica* | Low | 3 - 5 | BT, GN |
| Raspins Beach | Sand | | 2 - 4 | BT, GN |

The inshore distribution of platycephalid larvae was assessed during ichthyoplankton sampling conducted in November 1996 at four stations in Norfolk Bay (see Fig. 4.2). Samples were collected with a 100 cm diameter ring net with 500 μm mesh. Each station consisted of a surface and oblique tow to a maximum depth of 15 m (bottom depth permitting), at a tow speed of ~3 knots. During inshore surveys, surface and bottom temperatures were recorded with a temperature/depth probe ($^{\pm}$0.1° C, 0.1ppt).

*5.9.2.2 Laboratory analysis*

All juvenile and adult *P. bassensis* were retained and processed for biologicals including fork length (FL) (to the nearest millimetre), total weight (to the nearest gram) and sex, gonad stage, and gonad weight (to the nearest gram). Gonads were staged macroscopically (see Table 4.5) and sagittal otoliths were removed from fish and weighed whole (to the 0.01 gram).

Platycephalid larvae were sorted from plankton samples in a rotatable sorting ring under a dissecting microscope. All unspecified body lengths refer to notochord length (NL) in preflexion and flexion larvae (tip of the snout to the posterior end of the notochord), and to standard length (SL) (i.e. tip of the snout to the posterior region of the hypural plate) in postflexion larvae and juveniles. All measurements are expressed as mean percentage of body length. Pre-anal length is defined as the horizontal distance from the tip of the snout to the anterior origin of the anal fin or anal-fin anlagen. Pectoral-fin length is defined as the distance from the pectoral-fin base to the posterior tip of the longest pectoral ray. Body depth at pectoral is equivalent to 'body depth' of Leis and Rennis (1983). Other definitions, such as body shape, follow Leis and Trnski (1989). Nomenclature of head spination follows that of Moser and Ahlstrom (1978). Larval measurements were made

using an ocular micrometer, while juveniles were measured with vernier calipers. Larval drawings were made with the aid of a camera lucida.

Age estimates of *P. bassensis* were derived from sagittae that were transversely sectioned through a three stage process of embedding into polyester resin, sectioning to ~300mm thick and mounting on glass slides. Transverse sections of sagittae from *P. bassensis* <12 cm were made by mounting the central part of the sagittae on the edge of 1 mm thick glass slide with resin and grinding from anterior and posterior ends until a 1 mm section was obtained. Sections were then mounted on the surface of a glass slide and both surfaces ground with sequentially finer grades of carborundum paper until ~300mm thick and viewed at either 12, 25 or 50 times magnification using a dissecting microscope with transmitted light and displayed on a personal computer. A customised image analysis system was used to enable on-screen digitising and enhancing of each section.

Age was estimated by counting the presumed annual increments (opaque or dark zones) from the primordium to the edge of the otolith section on the ventral sector of the proximal side. Along this same axis the distance from the primordium to the outer edge of each opaque zone and the edge of the section was measured to the nearest 0.1 mm. The section with the clearest increments and most discernible primordium was used for counts and measurements. The opaque zones considered true annuli were distinguished from false checks as they extended down both the ventral and dorsal sides of the medial groove and were continuous from the ventral edge to the sulcus. An increment was considered complete when a distinct opaque zone was visible across the proximal face of the otolith section immediately inside a narrow discernible edge of translucent material. All counts and measurements of increments were made without knowledge of fish size, sex or date at capture.

#### *5.9.2.3 Statistical analysis*

Spatial and temporal variations in the distribution of *P. bassensis* in all areas was assessed using analysis of variance (ANOVA). Variations in abundance in gillnets from Norfolk Bay were analysed using a two-way ANOVA with habitat considered fixed and time a random factor. Time was considered random, as there was no *a-priori* reason for choosing sampling dates which were chosen to give an even spread of samples throughout the year. In Georges Bay and Prosser Bay, gillnet abundance was analysed using a two-way ANOVA with season and habitat considered fixed factors. Analysis of variations in the abundance of *P. bassensis* from the beam trawl in Norfolk Bay was restricted to fish <18.0 cm using a two-way ANOVA with habitat considered fixed and time a random factor. Due to low and patchy abundance of *P. bassensis* <18.0 cm in Georges Bay and Prosser Bay, no statistical analyses were done.

*5.9.2.4 Age Validation*

The periodicity of increment formation was determined from analysis of the temporal pattern of marginal increment development (distance between the outer edge of the outermost opaque zone and the otolith periphery). This was calculated as the index of completion [C] using the formula of Tanaka *et al.* (1981):

$$C = W_n/W_{n-1}$$

where $W_n$ = marginal increment and $W_{n-1}$ = previous complete increment. Mean monthly index of completion values were plotted separately for otoliths with 1-8 and >8 opaque zones with months pooled over different years.

*5.9.2.5 Precision of age estimates*

To compare the precision of age estimates a random subsample of 275 sagittae were read a second time by the main reader, and a second subsample of 100 sagittae by a second reader. The average percent error (APE) was calculated for both the within and between reader age estimates using the formulae of Beamish and Fournier (1981). In addition, the percentage agreement of the within and between reader age estimates was calculated as another means of evaluating precision.

*5.9.2.6 Growth*

An absolute age was assigned to *P. bassensis* using a birth date of 1 December, which corresponds to the mid-point of the spawning season. Von Bertalanffy growth curves were fitted to the individual length-at-age data for males and females separately and combined by direct non-linear least-squares estimation using *Genstat* statistical package. For both males and females, juveniles were ranked by size, with successive juveniles assigned an alternate sex. This was done to eliminate the bias of excluding slow growing juveniles that take longer to reach a size that can be sexed. The von Bertalanffy equation is defined as:

$$L_t = L_\infty\{1 - \exp[-K(t-t_0)]\}$$

where $L_t$ is the length at age $t$ (years), $L_\infty$ is the asymptotic length, K is the growth coefficient and $t_0$ is the hypothetical age at which the fish would have zero length if growth had followed that predicted by the equation. The growth curve derived for males and females was compared using an F-test on the ratio of the mean square for the combined fit and the sum of the error mean square for males and females fitted separately (Ratkowsky 1983).

*5.9.2.7 Age composition*

Using the estimated ages, mean lengths-at-age were calculated for males and females separately and combined. The age composition of the *P. bassensis* population was estimated for the 1995 samples, with the number of fish aged proportional to the number in

each 2 cm size-class from the size composition of the total population sample. Year-class distributions were also examined for 1995, with the year-class referring to the year in which the fish was spawned.

### 5.9.3 Spawning, early life history and recruitment

#### *5.9.3.1 Size at maturity*

*Platycephalus bassensis* were considered mature if macroscopic staging showed at least the presence of developing oocytes in females and developing testes in males (Stage 3 or greater). The smallest male and female *P. bassensis* to reach maturity were 19.0 and 20.0 cm, respectively (Fig. 5.9.1). All males larger than 24.5 cm and all females larger than 29.5 cm were mature. A comparison of the proportion of mature fish in each 0.5 cm size-class collected over the spawning season reveals that 50% of males and females were mature by 21.0 and 23.5 cm, respectively.

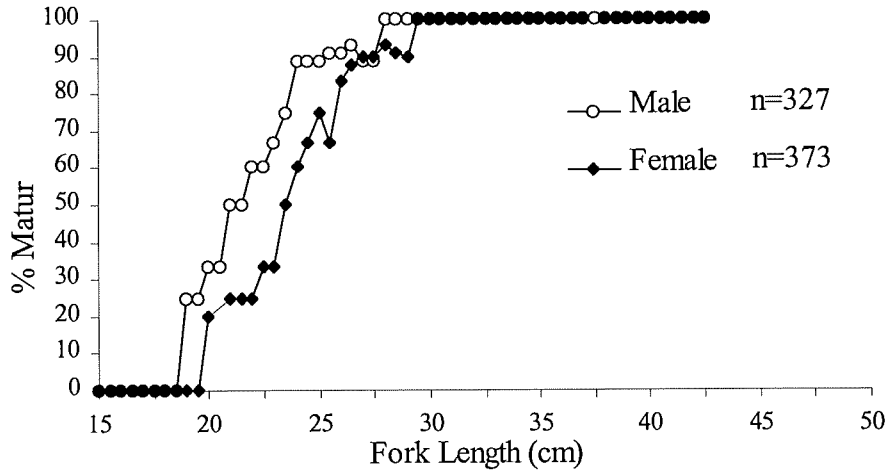



Fig. 5.9.1 Proportion of mature male and female *Platycephalus bassensis* by 0.5 cm size-classes. n is sample size.

#### *5.9.3.2 Gonadal development*

Trends in mean gonadosomatic (GSI) for male and female *P. bassensis* were analysed monthly from inshore regions from February 1995 to February 1997. Monthly mean GSI's showed the same overall trend for both males and females (Fig. 5.9.2). Mean female GSI's rose from a low in May to a peak in October in both years before declining through to low values by March. Mean male GSI's exhibited similar trends. While values decreased rapidly from October to November before increasing again in December, this trend reflects the small sample size in November in both years rather than indicating a period of reduced spawning activity in that month. Gonads reached a maximum of 10.2% and 18.7% of total body weight for males and females, respectively, during the spawning season.

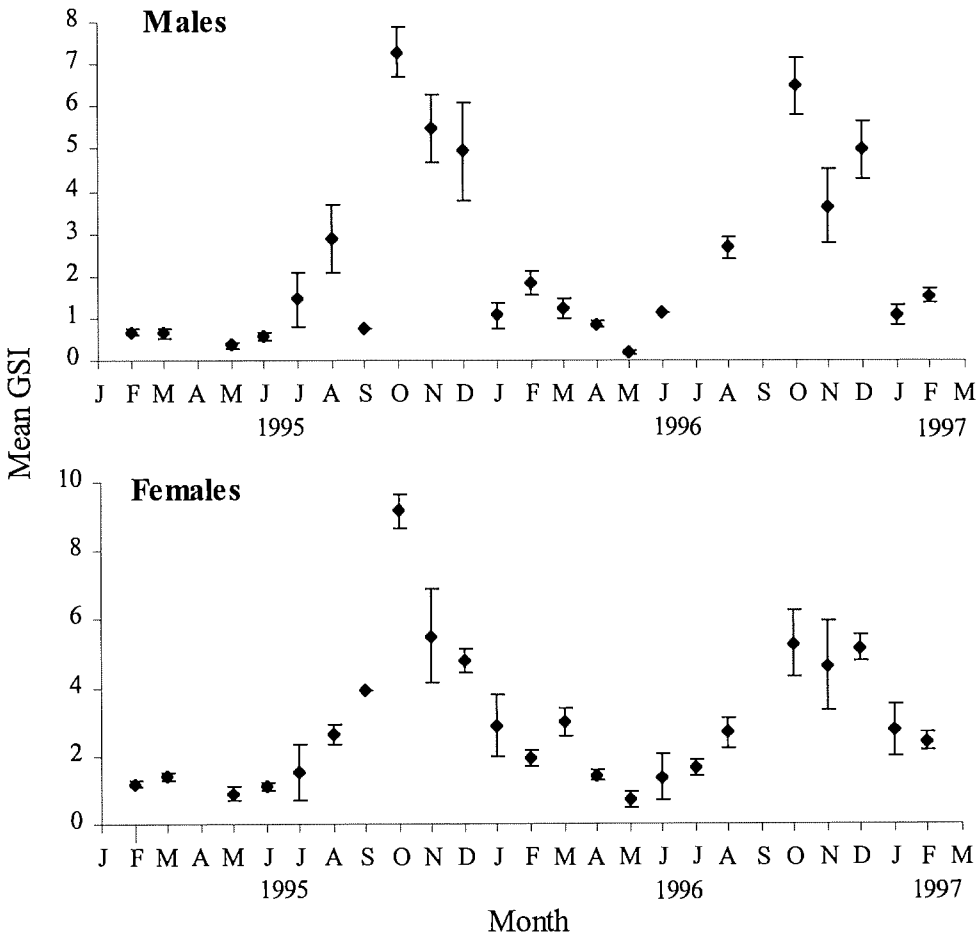



Fig. 5.9.2 Mean gonadosomatic indices (GSI) for male and female *Platycephalus bassensis* caught inshore between February 1995 and February 1997. Error bars are standard error.

The temporal patterns of spawning from the GSI's is also reflected in the monthly trend in gonad stages with all males and females in the resting phase (stage 2) from April to June, and ripe, running ripe and spent fish (≥ stage 5) from October to March (Fig. 5.9.3). The decrease in GSI's from October through to March reflects the increasing proportion of spent (stage 7) and recovering (stage 2) fish through these months and indicates that an increasing proportion of the population completes spawning between January and March.

### *5.9.3.3 Hydrography*

Sea-surface temperatures in Norfolk Bay between February 1995 and 1997 reflected the seasonal cycle of warming and cooling. Mean temperatures reached a minimum of 8.2° C in August 1995, rising to a maximum of 17.4° C in February 1996 (Fig. 5.9.4). There was no indication of differences between years in either minimum or maximum temperatures.

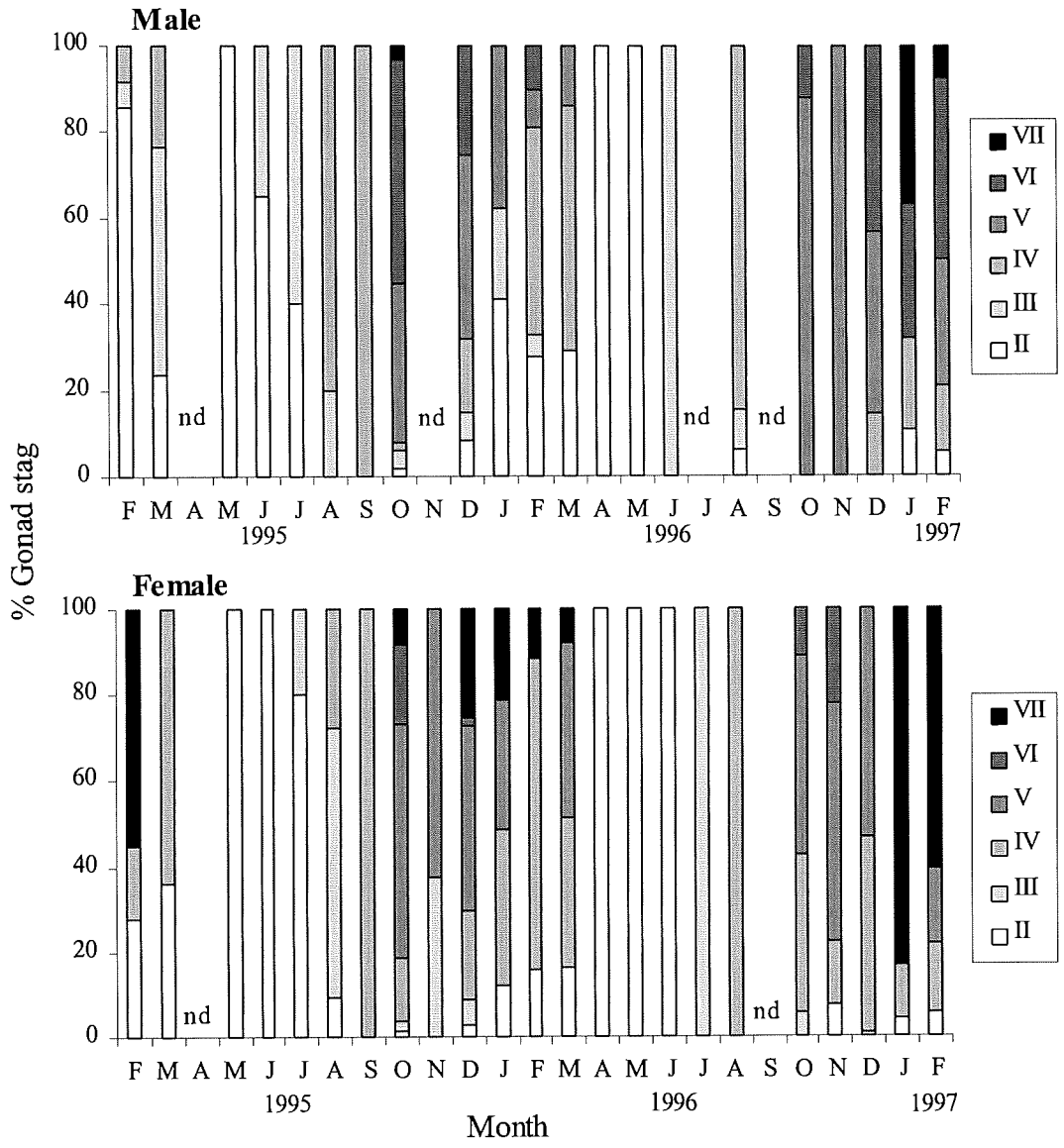



Fig. 5.9.3 Monthly percentage of gonad stages for male and female *Platycephalus bassensis* caught inshore between February 1995 and February 1997. nd represents months with no data.

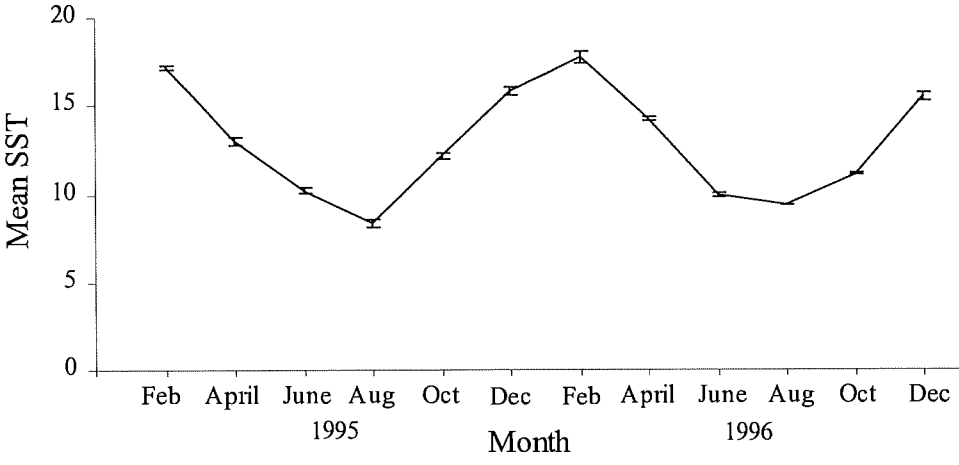



Fig. 5.9.4 Mean sea surface temperature (SST) (° C) of sampling sites in Norfolk Bay between February 1995 and December 1996. Error bars are standard error.

*5.9.3.4 Larval development*

Identification of larvae to the family Platycephalidae was based on a combination of characters including a large and wide head with extensive spination, moderate to large, fan shaped pectoral fins and the presence of 26-28 myomeres (Neira and Miskiewicz 1998). Identification to species using meristic characters is difficult as there are few differences between species (Table 5.9.2). Larvae were identified as those of *P. bassensis* by comparison with adult features, comparison of known adult distributions and spawning times, and the establishment of a developmental series.

Despite extensive sampling of southern and eastern Tasmania shelf waters (Jordan 1997), and inshore waters in the present study, no juvenile or adult specimens of *Neoplatycephalus aurimaculatus, Platycephalus speculator* and *P. laevigatus* were recorded, and these species appear to be restricted to waters of Bass Strait and northern Tasmania. While *N. richardsoni* are common on the shelf of southern and eastern Tasmania, spawning occurs during summer with no evidence of spring spawning (Jordan 1997). A single series of platycephalid larvae were present in samples taken in Norfolk Bay during the period of peak spawning activity of *P. bassensis* (November), strongly suggesting they are the larvae of *P. bassensis*. A second developmental series of platycephalid larvae taken in the eastern Tasmanian shelf surveys (Marshall and Jordan 1989) was characterised by melanophores on the dorsal surface of the trunk in all stages, and the presence of large teeth on the lower jaw and roof of the mouth in flexion and postflexion larvae. The presence of strong teeth is a diagnostic character of the genus *Neoplatycephalus* (Gomon *et al.* 1994), indicating that this second series were larvae of this genus.

Table 5.9.2 Meristic characters of platycephalid species present in Tasmanian waters. Collated from Gomon *et al.* (1994).

| | D | A | P1 | P2 | C | Vertebrae |
|---|---|---|---|---|---|---|
| *Neoplatycephalus richardsoni* | VIII-IX,14 | 14 | 19-20 | I,5 | 15 | - |
| *Neoplatycephalus aurimaculatus* | IX,14 | 14 | 16-20 | I,5 | 15 | - |
| *Platycephalus bassensis* | VIII-IX,14 | 14 | 19-20 | I,5 | 15 | 27 |
| *Platycephalus speculator* | VIII,14 | 14 | 19-21 | I,5 | 15 | 27 |
| *Platycephalus laevigatus* | IX,14-15 | 14-15 | 18-21 | I,5 | 15 | 27 |

Larvae of *P. bassensis* are pelagic. The smallest *P. bassensis* larvae examined (3.0 mm) had a functional mouth and coiled gut with yolk absorption complete. The head is small and compressed in preflexion larvae (HL = 24%), but becomes moderate during flexion (Table 5.9.3, Fig. 5.9.5A-D). The mouth is large, reaching to approximately the centre of the eye in all larval stages, while the snout increases in length and becomes flatter during flexion. There are no strong teeth on the roof of the mouth or lower jaw in any stage. A small gas bladder was inflated and visible above the foregut in preflexion and flexion

larvae. The body depth is moderate (BD=19-21%) with little change in body shape during larval development. Pectoral fins are moderate and fan shaped increasing in size during flexion. Notochord flexion commences at 6.0 mm and was almost complete in the largest larvae examined (8.4 mm). Larvae have 27 myomeres (10~11+16~17).

Table 5.9.3 Body proportions of *Platycephalus bassensis* larvae (expressed as mean percentage of body length, with standard deviations in parentheses; n = number of individuals). Specimens below dashed lines are undergoing notochord flexion.

| Size range (mm) | n | Pre-anal length | Body depth at pectoral | Head length | Pectoral-fin length |
|---|---|---|---|---|---|
| 3.01-4.00 | 7 | 43.5 (3.5) | 21.2 (2.8) | 24.0 (1.3) | 11.3 (0.9) |
| 4.01-5.00 | 25 | 46.1 (3.0) | 19.6 (0.9) | 26.3 (1.7) | 12.8 (1.1) |
| 5.01-6.00 | 18 | 49.4 (2.0) | 18.9 (0.4) | 27.5 (1.5) | 14.9 (1.0) |
| 6.01-8.39 | 15 | 51.0 (1.4) | 20.9 (1.4) | 30.5 (1.8) | 17.5 (0.9) |

Development of the pectoral fins was precocious with 1-2 incipient rays present in the smallest larvae examined (3.0 mm), ossification commencing in late preflexion larvae (5.4 mm) (Fig. 5.9.5). The pectorals have a full complement of 19-20 rays and reaching up to 17.5 % of body length during flexion (7.4 mm). Pelvic fin buds are visible in 5.9-6.0 mm larvae as small swellings either side of the gut. The pelvics develop rapidly, having a full complement of I,5 rays by 8.4 mm. Anlagen of both anal and second dorsal fins appear early during flexion with distinct bases present by 7.0 mm. Incipient rays first appear by 7.4 mm with up to 12 rays ossified in the largest larva examined (8.4 mm). The first dorsal fin anlagen first appears by 7.4 mm with 5 spines ossified by 8.4 mm. The caudal fin anlagen first appears on the ventral surface of the notochord immediately prior to flexion (5.8 mm) with a total of 10 rays ossified by 8.4 mm.

One small anterior preopercular spine is present in the smallest larvae examined (3.0 mm), with two present by 3.5 mm (Fig. 5.9.5). A single posterior preopercular spine is present by 3.5 mm, increasing to four immediately prior to flexion (5.9 mm), with the second and third spines becoming the longest. A single parietal spine develops at about 4.0 mm, with a further small spine appearing on the anterior portion of the spine by 5.1 mm. A small supraocular spine is visible by 5.8 mm and remains small after settlement.

Pigment appears at the tip of the upper and lower jaws and snout by 3.5 mm and remains moderate during flexion (Fig. 5.9.5). Several scattered melanophores appear on the preopercle by 5.2 mm and on the dorsal surface of the head by about 5.9 mm. A single row of 13-17 melanophores is present on the ventral surface of the tail and 5-9 small melanophores on the ventral surface of the gut in all larval stages. Numerous small melanophores are present on the posterior portion of the gut in preflexion larvae, increasing in number during flexion. Pigment on the pectoral fin appears in early preflexion larvae and is restricted to the upper fin rays, with the lower rays remaining unpigmented. Internal pigment is present on the dorsal surface of the gas bladder during all larval stages.

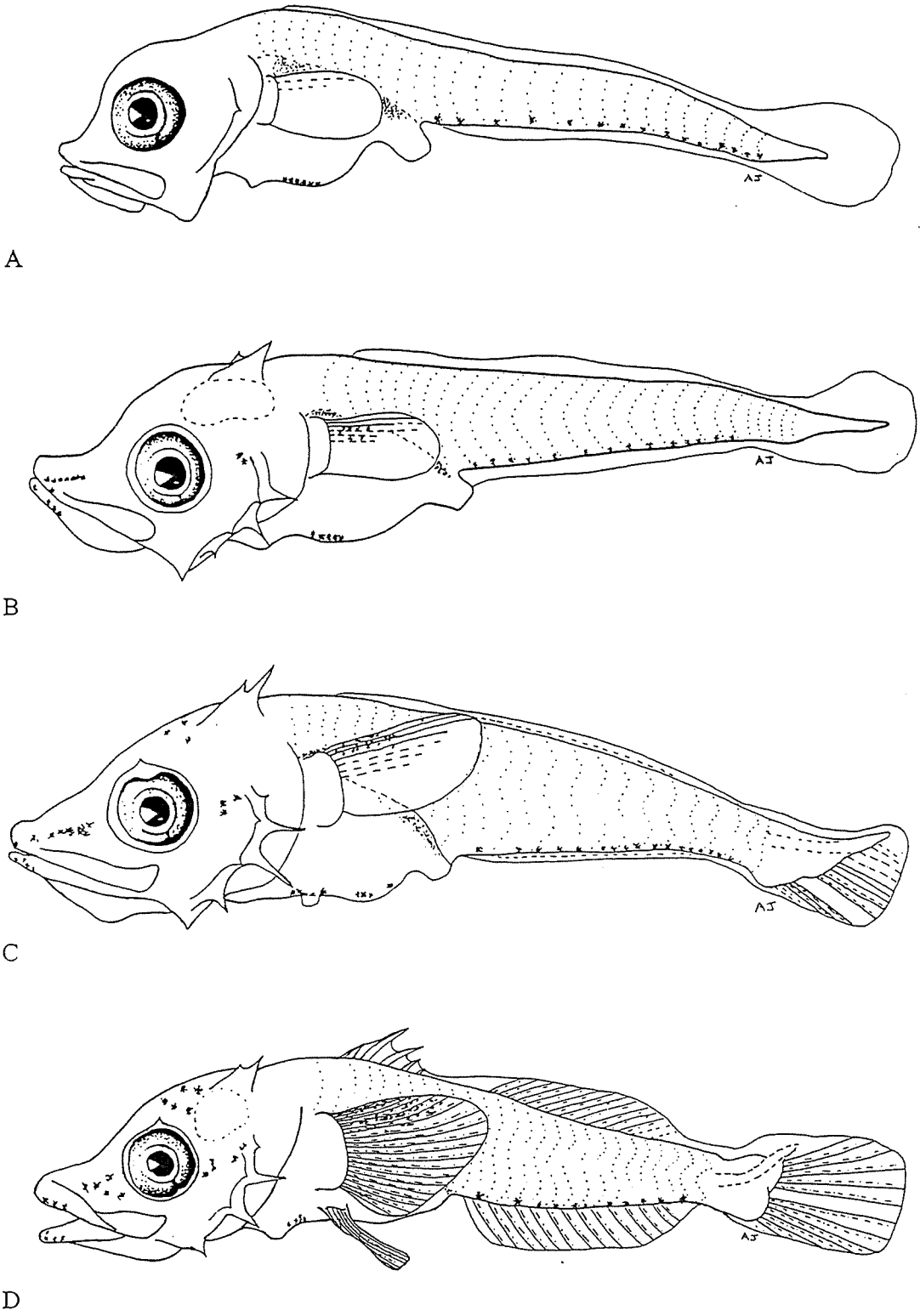


Fig. 5.9.5 Developmental stages of *Platycephalus bassensis* larvae: (A) 3.0 mm, (B) 5.7 mm, (C) 7.1 mm, (D) 8.5 mm.

*5.9.3.5 Inshore larval distribution*

Despite sampling during the three months of peak spawning activity (Oct-Dec), *P. bassensis* larvae were only caught in November 1996. At that time larvae were present at all four stations, although densities were highest in the middle of the bay, peaking at 165 larvae.1000m$^{-3}$ (Fig. 5.9.6). *P. bassensis* larvae were restricted to oblique tows with no larvae present in surface tows.

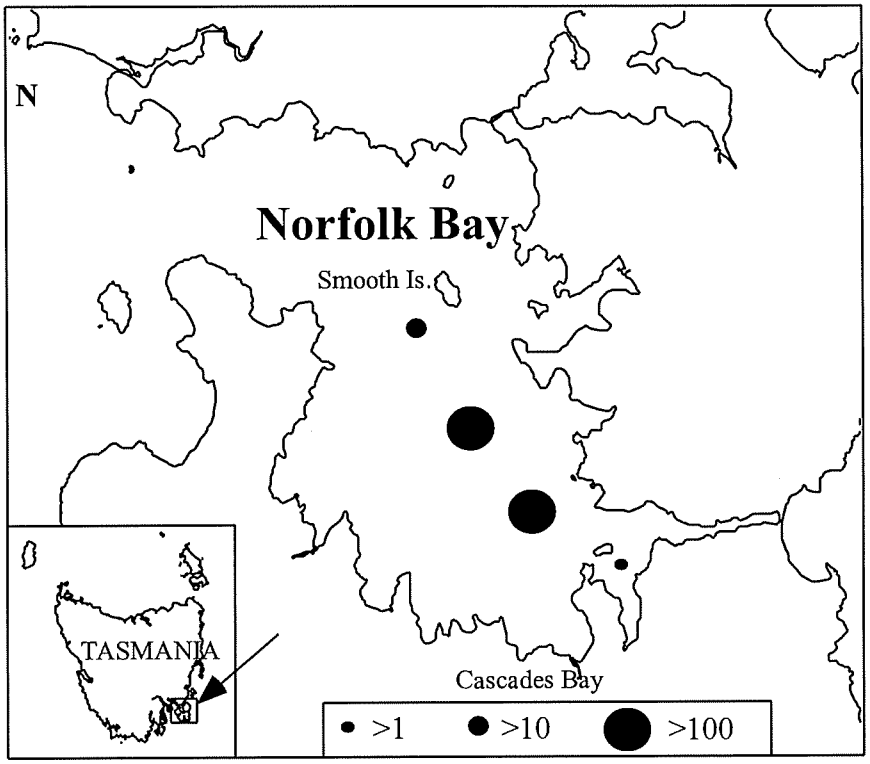



Fig. 5.9.6 *Platycephalus bassensis* larval concentrations (no.1000m$^{-3}$) during November 1996 in Norfolk Bay.

*5.9.3.6 Recruitment*

Length-frequency distributions of juvenile *P. bassensis* are dominated by a single size-class from March to December 1996, although in some months there is some evidence of bimodal distribution (Fig. 5.9.7). This cohort had a mean length of 7.6 cm in March 1996 and represents 0+ fish from spawning that took place the previous spring and summer. The broad range of lengths (5.6-9.6 cm) suggests that settlement occurred over an extended period. The earliest month settlement was recorded was January, although lengths ranged from 2.3 to 7.4 cm in that month suggesting settlement had begun some time earlier. The lack of new recruits in December may reflect the smaller sample size in that month. Two additional cohorts with mean lengths of 13.9 cm and 19.0 cm were present in January 1997, representing the 1+ and 2+ age-classes. Modal progressions of the 0+ age-class in both years indicates growth is rapid until around May, with little increase in length until October when growth resumes.

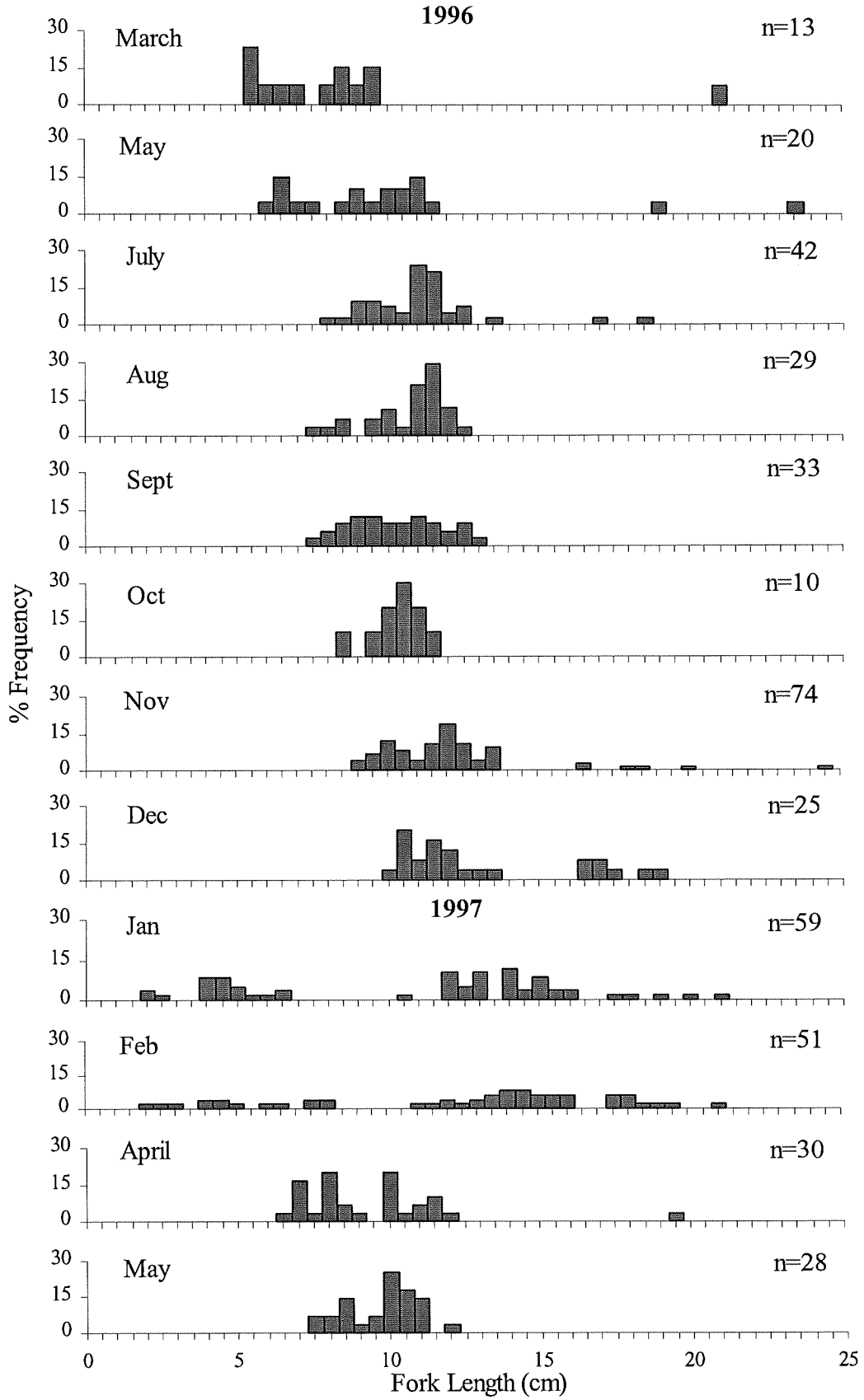



Fig. 5.9.7 Length-frequency distributions of juvenile *Platycephalus bassensis* from North West Bay between March 1996 and May 1997. n is sample size.

## 5.9.4 Spatial and temporal patterns in abundance and distribution

### *5.9.4.1 Catch rates*

Abundances of *Platycephalus bassensis* from gillnets in Norfolk Bay varied significantly between sample dates, and there was also a significant habitat and date interaction (Table 5.9.4, Fig. 5.9.8). Post-hoc tests indicated that in terms of sample dates, abundance was significantly higher in *Heterozostera* compared to unvegetated habitat in October and December, 1995, but not significantly different as all other dates. In terms of habitats, post-hoc tests indicate that abundance in unvegetated habitats was significantly higher in February 1995, April 1995 and October 1996, than June 1995, August 1995 and June 1996, but not significantly different as all other dates. Abundance in *Heterozostera* was significantly higher in October 1995, December 1995 and December 1996, than June and August in both 1995 and 1996, but not significantly different as all other dates.

Table 5.9.4 Analysis of variance of ln(x+1) transformed abundance ($N.hr^{-1}$) of *Platycephalus bassensis* in gillnets in *Heterozostera tasmanica* and unvegetated habitats in Norfolk Bay.

| Factor | Hypothesis | DF | MS | F | P |
|---|---|---|---|---|---|
| Habitat | a/ab | 1 | 0.155 | 1.918 | 0.147 |
| Date | b/r | 11 | 0.131 | 4.210 | 0.002 |
| Habitat*Date | ab/r | 11 | 0.081 | 2.600 | 0.024 |
| Residual | r | 24 | 0.031 | | |

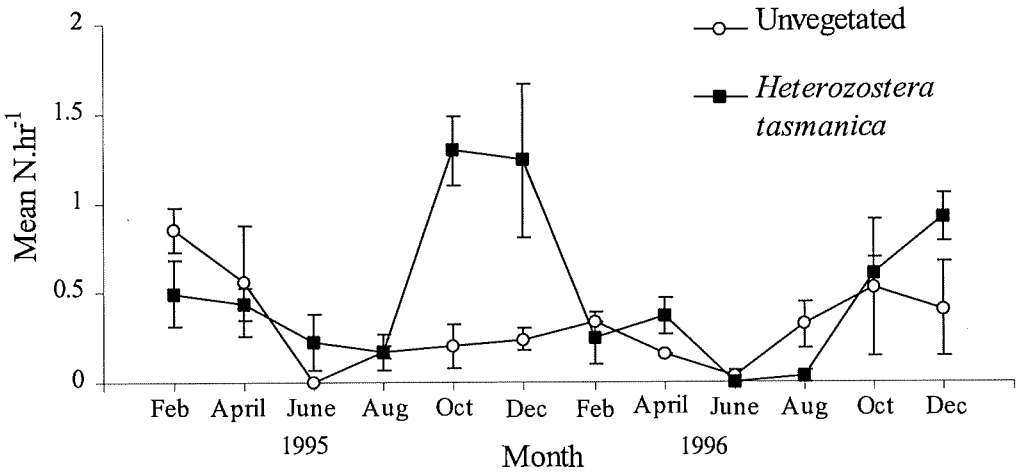



Fig. 5.9.8 Mean abundance ($N.hr^{-1}$) of *Platycephalus bassensis* collected in gillnets from *Heterozostera tasmanica* and unvegetated habitats sampled every two months in Norfolk Bay. Error bars are standard error.

Abundance of 0+ and 1+ *P. bassensis* in Norfolk Bay was significantly higher in unvegetated compared to *Heterozostera* habitats throughout all sampling dates (Table 5.9.5, Fig. 5.9.9).

Table 5.9.5 Analysis of variance of ln(x+1) transformed abundance (N.tow$^{-1}$) of *Platycephalus bassensis* <18.0 cm in *Heterozostera tasmanica* and unvegetated habitats in Norfolk Bay.

| Factor | Hypothesis | DF | MS | F | P |
|---|---|---|---|---|---|
| Habitat | a/ab | 1 | 11.089 | 29.619 | <0.001 |
| Date | b/r | 11 | 0.667 | 1.880 | 0.066 |
| Habitat*Date | ab/r | 11 | 0.374 | 1.060 | 0.415 |
| Residual | r | 48 | 0.354 | | |

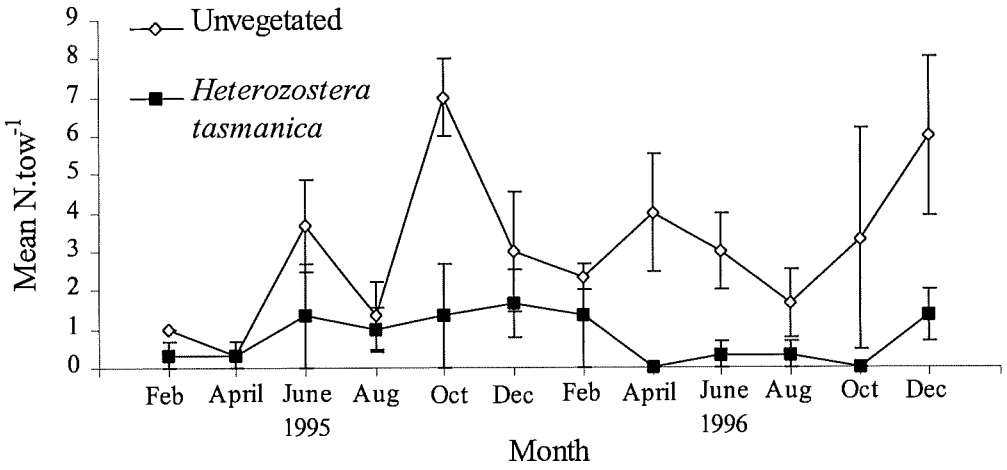



Fig. 5.9.9 Mean abundance (N.tow$^{-1}$) of 0+ and 1+ *Platycephalus bassensis* collected by beam trawl from *Heterozostera tasmanica* and unvegetated habitats sampled every two months in Norfolk Bay. Error bars are standard error.

Abundance of *P. bassensis* in Georges Bay indicates significant seasonal variability, with a significant habitat and season interaction (Table 5.9.6, Fig. 5.9.10). Post-hoc tests indicated that abundance was significantly higher in *Heterozostera* compared to unvegetated habitats in spring, but not significantly different in all other seasons. In terms of seasons, abundance was significantly higher in spring than all other seasons in both *Heterozostera* and unvegetated habitats.

Table 5.9.6 Analysis of variance of ln(x+1) transformed abundance (N.hr$^{-1}$) of *Platycephalus bassensis* in gillnets in *Heterozostera tasmanica* and unvegetated habitats in Georges Bay.

| Factor | Hypothesis | DF | MS | F | P |
|---|---|---|---|---|---|
| Season | a/r | 3 | 0.076 | 37.540 | <0.001 |
| Habitat | b/r | 1 | 0.001 | 0.450 | 0.523 |
| Season*Habitat | ab/r | 3 | 0.013 | 6.390 | 0.016 |
| Residual | r | 8 | 0.002 | | |

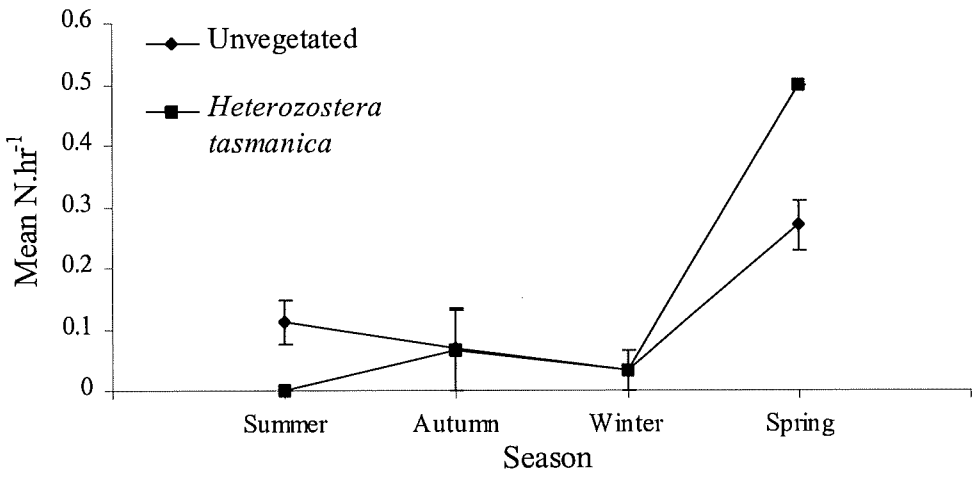



Fig. 5.9.10 Mean abundance ($N.hr^{-1}$) of *Platycephalus bassensis* collected in gillnets from *Heterozostera tasmanica* and unvegetated habitats sampled seasonally in Georges Bay. Error bars are standard error.

Abundance of *P. bassensis* in Prosser Bay indicates significant seasonal variability, with a significant habitat and season interaction (Table 5.9.7, Fig. 5.9.11). Post-hoc tests reveal that abundance was significantly higher in unvegetated relative to *Heterozostera* habitats in spring, but not significantly different in all other seasons. In terms of seasons, abundance was significantly higher in spring in *Heterozostera* than all other seasons. In unvegetated habitats, abundances were significantly higher in spring than summer and autumn, which were significantly higher than winter.

Table 5.9.7 Analysis of variance of ln(x+1) transformed abundance ($N.hr^{-1}$) of *Platycephalus bassensis* in gillnets in *Heterozostera tasmanica* and unvegetated habitats in Prosser Bay.

| Factor | Hypothesis | DF | MS | F | P |
|---|---|---|---|---|---|
| Season | a/r | 3 | 0.173 | 89.030 | <0.001 |
| Habitat | b/r | 1 | 0.007 | 3.580 | 0.095 |
| Season*Habitat | ab/r | 3 | 0.009 | 4.490 | 0.040 |
| Residual | r | 8 | 0.002 | | |

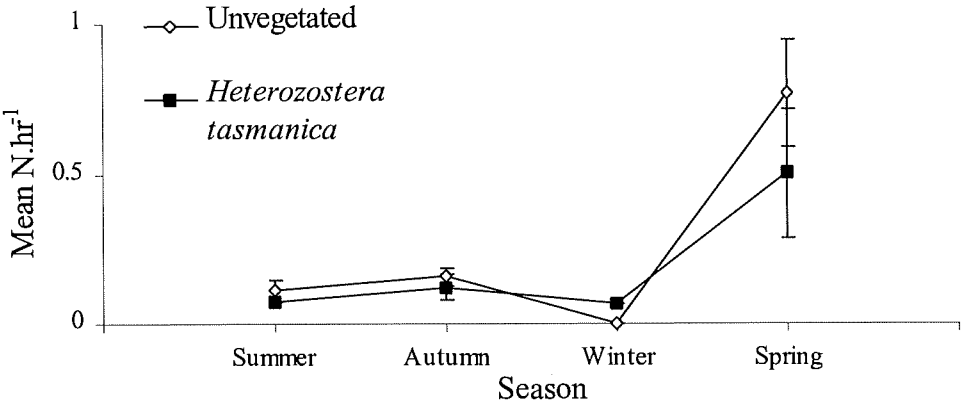



Fig. 5.9.11 Mean seasonal abundance ($N.hr^{-1}$) of *Platycephalus bassensis* collected in gillnets from *Heterozostera tasmanica* and unvegetated habitats in Prosser Bay. Error bars are s.e.

*5.9.4.2 Size composition*

*Platycephalus bassensis* in Norfolk Bay ranged from 2.1 to 46.6 cm, with evidence of two distinct modes in the distribution, one at around 9 cm and the other at 33 cm, with a smaller mode at 15 cm (Fig. 5.9.12). The size-classes > 23 cm represents fish caught by gillnet, with the increase in the proportion of fish > 28 cm reflecting the increased selectivity of the 64 mm gill-mesh for *P. bassensis* above that length. There was a considerable difference in the gillnet size compositions between habitats, with fish >35 cm making up 40% of the sample from *Heterozostera* compared to 8% from unvegetated habitats. There is little change in the seasonal size composition of the gillnet size-classes (> 23 cm), throughout the year (Fig. 5.9.13).

The smaller modes at 9 and 15 cm represents *P. bassensis* caught by beam trawl. Within this size range the overall size composition was similar for *Heterozostera* and unvegetated habitats, although fish < 6 cm were restricted to unvegetated sites (Fig. 5.9.12). Seasonal length-frequency distributions show progression of the smallest size-class, previously identified as the 0+ cohort, in both 1995 and 1996 (Fig. 5.9.13). The appearance of the smallest new recruit occurred in February of both years at a length of 2-3 cm. The 0+ cohort in 1995 had progressed to a mean size of 7.5 cm by August and 10.1 cm by December. Few 1+ and 2+ fish were present in beam trawl samples in Norfolk Bay in both years, although the 15 cm mode in the total length-frequency distributions represents the 1+ age-class. This size-at-age is consistent with that reported for *P. bassensis* from North West Bay (Fig. 5.9.7).

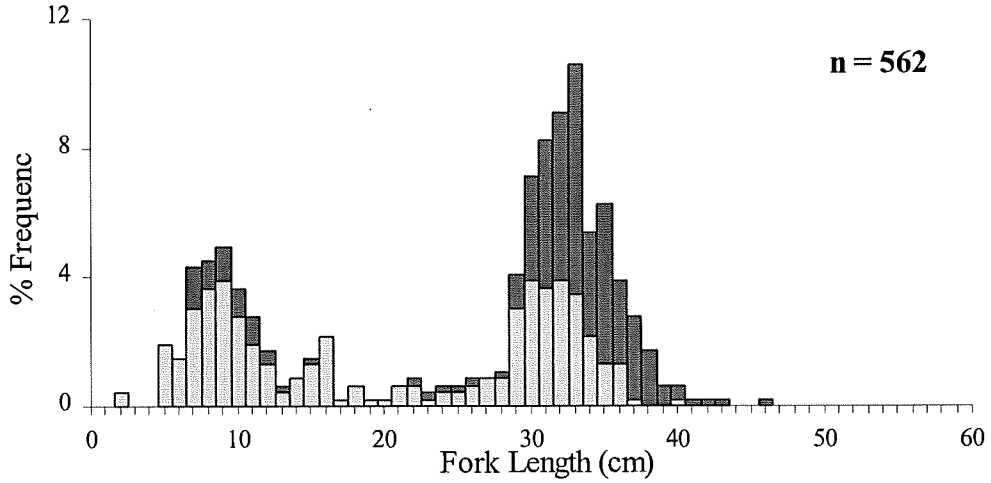



Fig. 5.9.12 Length-frequency distribution of *Platycephalus bassensis* collected with beam trawl and gillnets from *Heterozostera tasmanica* (dark bars) and unvegetated habitats (light bars) in Norfolk Bay between February 1995 and December 1996. n is sample size.

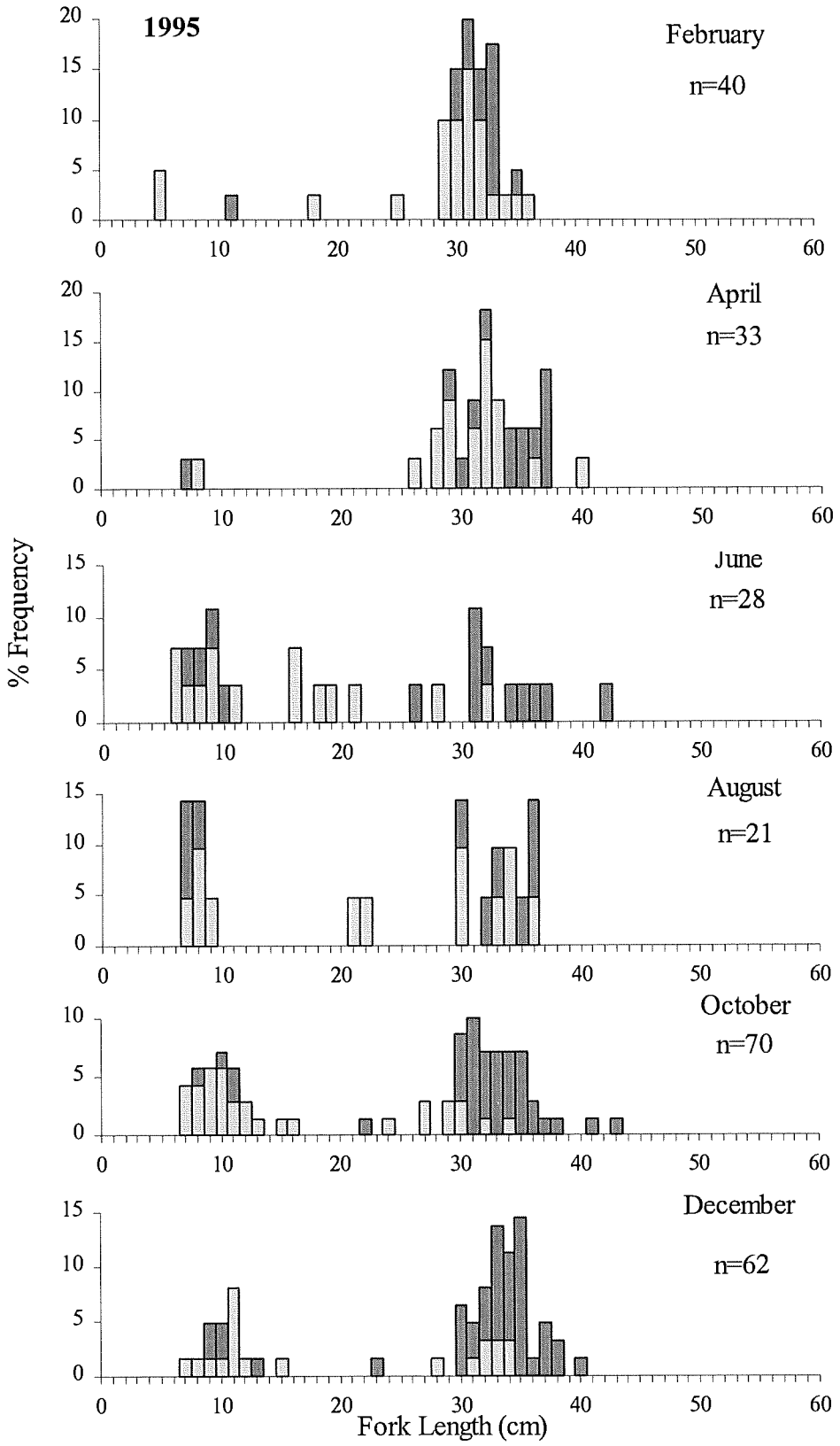



Fig. 5.9.13 Bi-monthly length-frequency distribution of *Platycephalus bassensis* collected with beam trawl and gillnets from *Heterozostera tasmanica* (dark bars) and unvegetated habitats (light bars) in Norfolk Bay between February 1995 and December 1996. n is sample size.

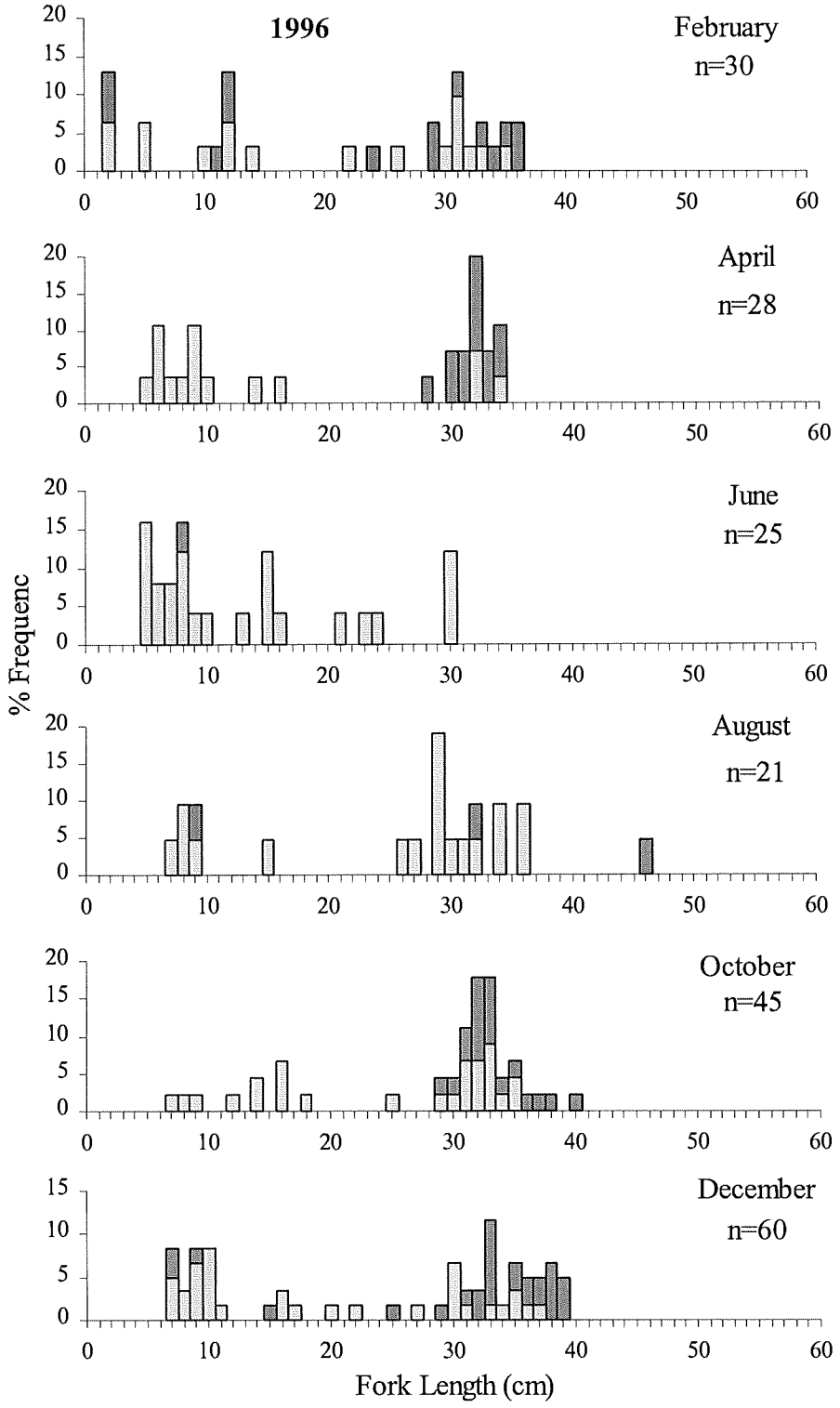



Fig. 5.9.13 (Cont). Bi-monthly length-frequency distribution of *Platycephalus bassensis* collected with beam trawl and gillnets from *Heterozostera tasmanica* (dark bars) and unvegetated habitats (light bars) in Norfolk Bay between February 1995 and December 1996. n is sample size.

*5.9.4.3 Nearshore beach survey*

Very few *P. bassensis* were caught in nearshore beach habitats in south-eastern Tasmania, occurring in only 8% of hauls (n=33). All fish were caught at unvegetated sites which were represented by different levels of exposure. Catches consisted exclusively of juveniles in the 0+ age-class, ranging in size from 4.4 to 9.7 cm (Fig. 5.9.13).

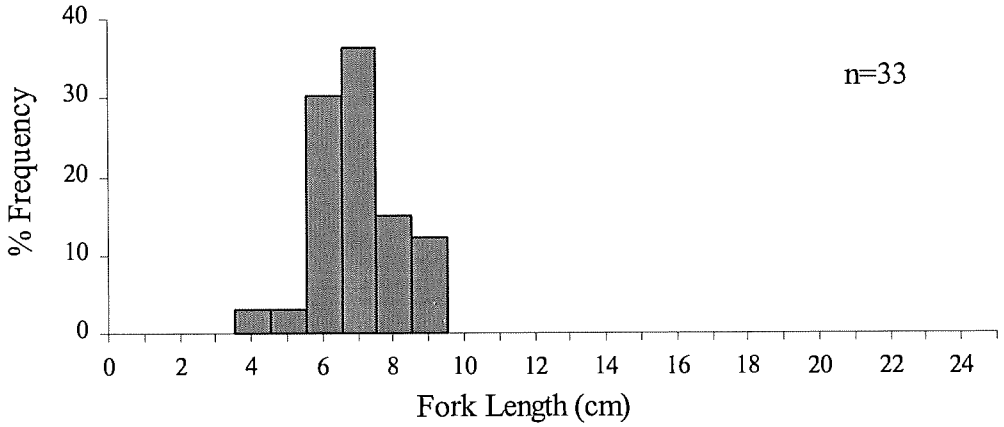



Fig. 5.9.14 Length-frequency distributions of *Platycephalus bassensis* sampled from nearshore beach habitats in south-east Tasmania. n is sample size.

5.9.5 Age, growth and sex/age composition

*5.9.5.1 Size and sex composition*

Length-frequency distributions were determined separately for male and female *P. bassensis* (Fig. 5.9.15). The overall distribution was dominated by a single mode at around 33 cm, although a broad range of smaller beam trawl caught fish around 15 to 20 cm was also evident. Lengths ranged from 12.6 to 42.7 cm (mean 28.5 cm) for males and 12.0 to 47.5 cm (mean 31.1 cm) for females. The small proportion of fish less than around 28 cm reflects the decreased selectivity of these size-classes in the gillnets.

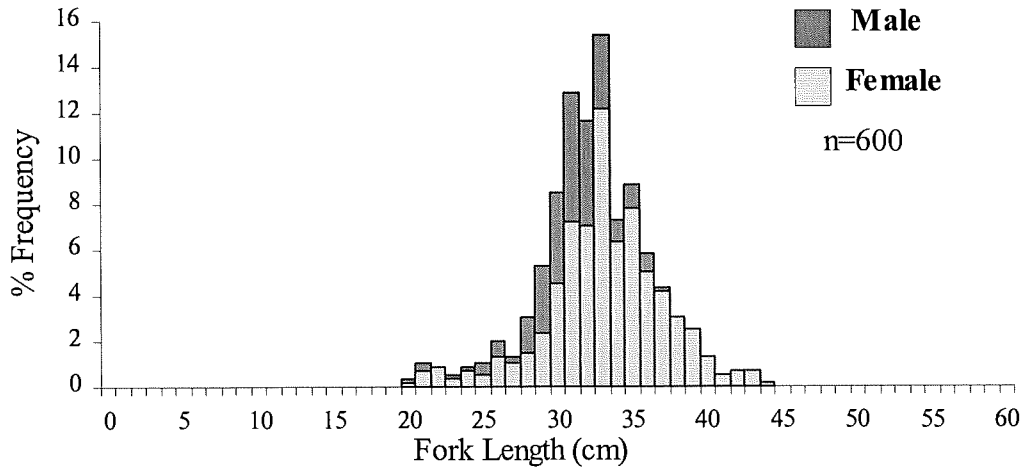



Fig. 5.9.15 Length-frequency distributions of male and female *Platycephalus bassensis* from inshore regions of southern and eastern Tasmania. n is sample size.

Sex ratios were determined for the inshore population of *P. bassensis* >20 cm, with the proportion of females significantly higher than males (Table 5.9.8). This pattern was consistent in all seasons except winter, where the sex ratio did not differ from 1:1. Sex ratios varied considerably by size, with females generally dominant in most size-classes, except between 27 and 31 cm where they were not significantly different (Chi-square, P>0.5) (Fig. 5.9.16). The dominance of females above 35 cm can be attributed to different growth rates between the sexes.

Table 5.9.8 Sex ratios of *Platycephalus bassensis* >20 cm, based on proportion of females (prop. F) by season. P is probability of sex ratios varying from 1:1 based on Chi-square tests. n is sample size.

| Season | Summer | Autumn | Winter | Spring |
|---|---|---|---|---|
| prop. F | 71.4 | 62.6 | 50.5 | 74.6 |
| n | 402 | 99 | 109 | 67 |
| P | p<0.001 | 0.01<p<0.05 | p>0.5 | p<0.001 |

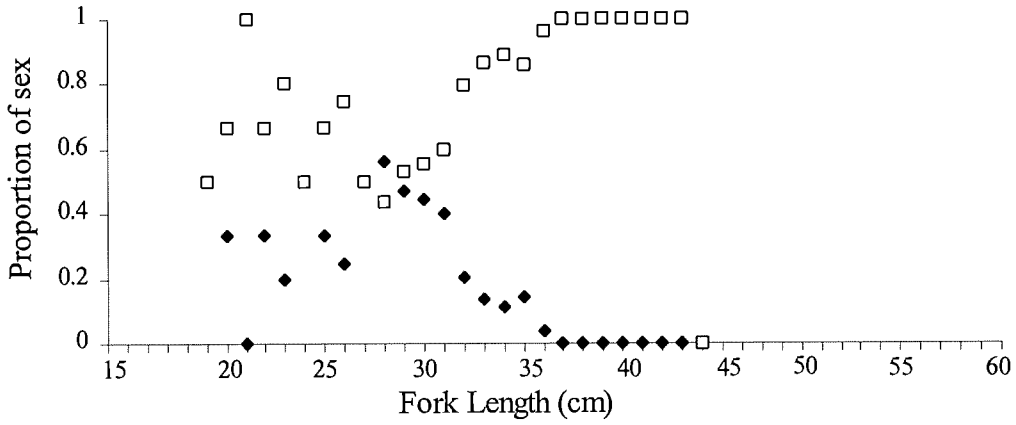
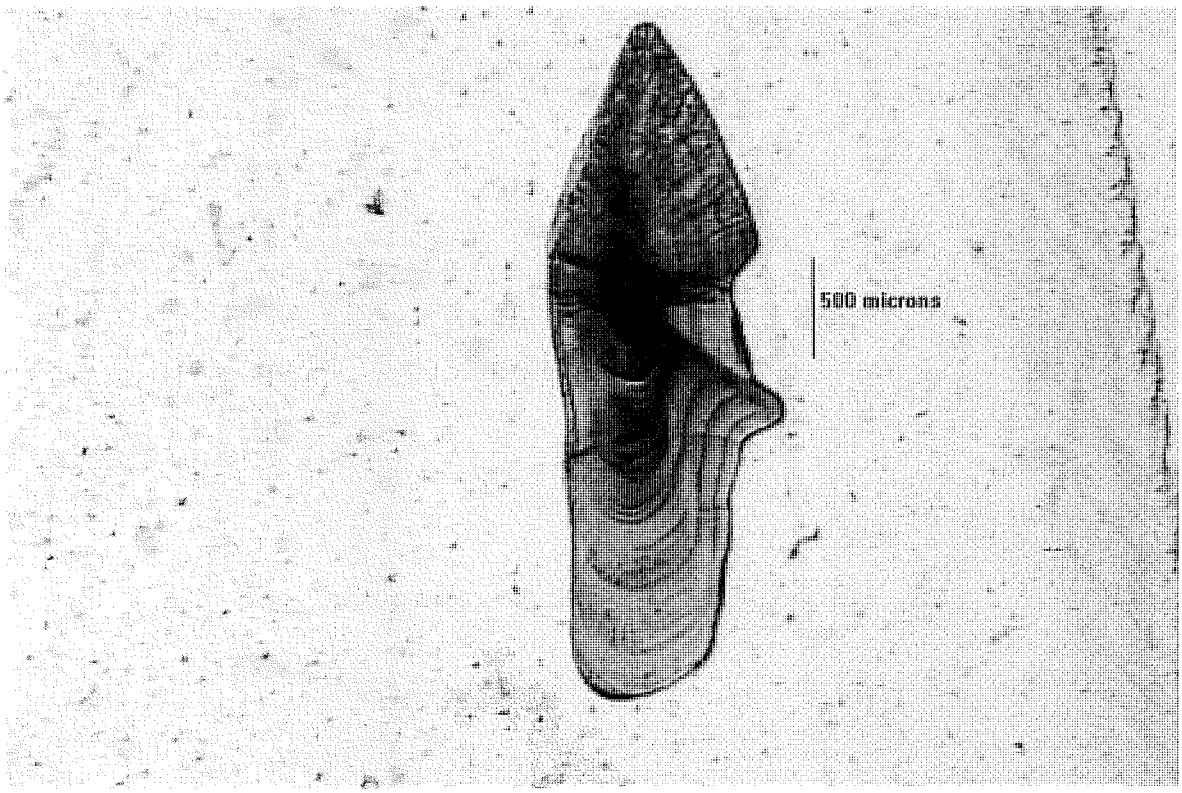



Fig. 5.9.16 Proportion of male (diamond) and female (squares) *Platycephalus bassensis* >20 cm by 1 cm length-class.

### *5.9.5.2 Otolith structure and interpretation*

Sagittal sections of *P. bassensis* showed clear and distinctive alternating opaque and translucent zones seen under transmitted light (Fig. 5.9.17A,B). A total of 12.6% of otoliths were rejected due to the poor quality of sections. The increment banding pattern remained relatively easy to read in older fish despite the narrowing of translucent zones. The primordial area of all otoliths consisted of an opaque region with no obvious increment structure. Immediately adjacent to this was a broad opaque zone with a mean radius (±s.d.) of 525 ±54 μm that occurred in 25.2% of all sagittae examined (Fig. 5.9.17B). This zone was characterised by being fainter than adjacent opaque zones and not being continuous around the distal face of the otolith. A second broad opaque zone with a mean radius of

A




B

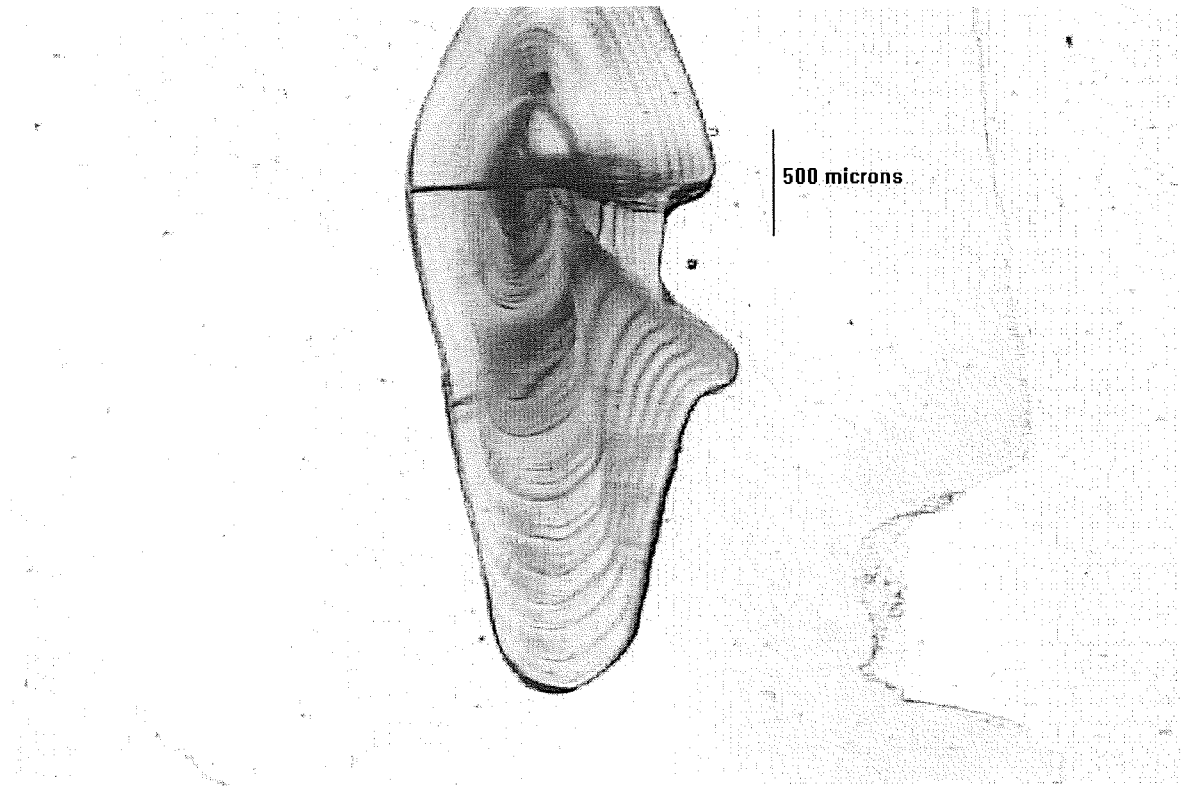



Fig. 5.9.17 Transverse section of sagittal otoliths of (A) 4 year old and (B) 8 year old *Platycephalus bassensis*

877 ±78 μm occurred in 72.2% of all sagittae. The structure of the otolith differed outside this second zone with all sagittae of sufficient radius having a consistent narrow opaque zone with a mean radius of 1200 ±87 μm. Beyond this, there were clear and distinctive opaque zones, initially 310 ±68 μm apart but generally decreasing in width towards the margin.

Given the variability in the structure and consistency of the opaque zones in the region of the primordium, the definition of the first annual increment was based on the relationship between the otolith radius and length of the 0+ cohort. Monthly progressions of juvenile size-compositions show *P. bassensis* first appeared in samples in January at around 2-7 cm and progressed rapidly through summer to around 9 cm by May (Fig. 5.9.18). This cohort had reached a mean length of around 11 cm by the following November. These lengths are consistent with that previously described as the 0+ age-class from spawning that peaked the previous spring and summer. By December this cohort progressed into the 1+ age-class, given the birth date of 1 December, which corresponds to the mid-point of the spawning season.

The monthly progression of the otolith radius of these two age-classes is shown in Fig. 5.9.19. The otolith radius of the smaller cohort increased from around 546 μm in February to 692 μm in May. By December the mean radius was 872 μm, a radius consistent with the second broad opaque zone visible in the primordial region of most otoliths, and hence defined as the first annual increment. Both the modal length and otolith radius increased in the larger cohort to around 16 cm and 1207 μm respectively by the following November. This is consistent with the radius of the first distinct narrow opaque zone seen in all otoliths, and hence defined as the second annual increment.

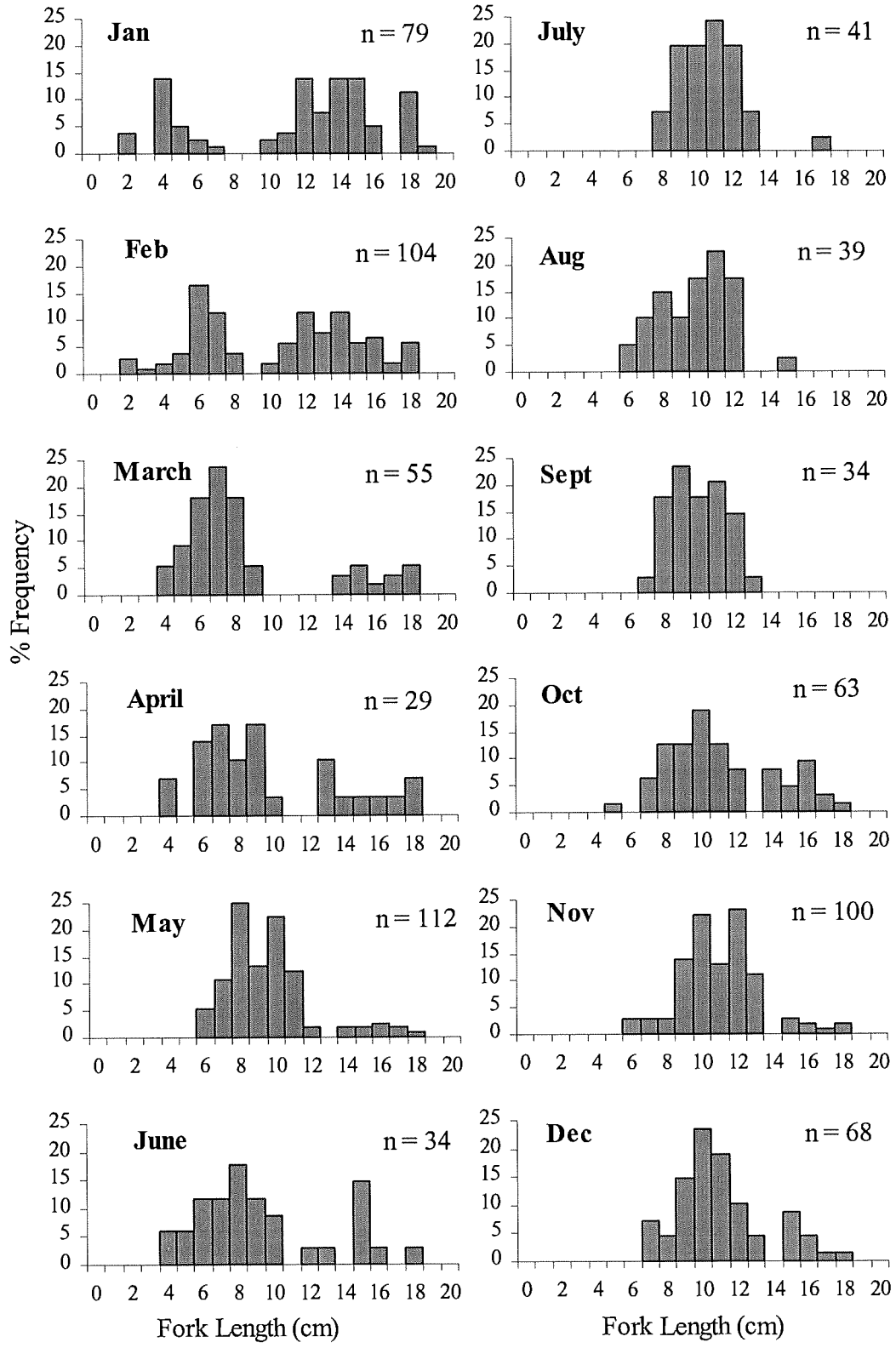



Fig. 5.9.18 Monthly length-frequency distributions of juvenile *Platycephalus bassensis* from southern and eastern Tasmania. n is sample size.

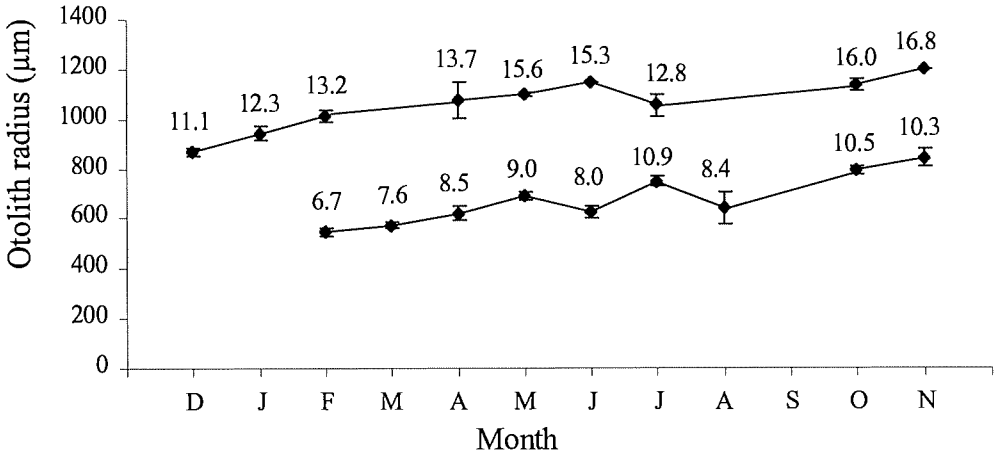



Fig. 5.9.19 Mean monthly progression of the otolith radius (µm) of juvenile *Platycephalus bassensis* from southern and eastern Tasmania. Values labels are corresponding mean fish lengths (cm). Error bars are standard error.

### *5.9.5.3 Validation*

Trends in the monthly pattern of marginal increment development was used to determine the periodicity of annulus formation. For otoliths with one opaque zone the marginal increment rose to a peak in November before decreasing rapidly in December and increasing again over the following months before levelling off in winter (Fig. 5.9.20). Similar monthly trends were apparent in otoliths with two or more opaque zones with marginal increments falling rapidly in December and January (summer). The rapid drop in marginal increments in summer indicates that translucent material has started to form at that time, with opaque material forming between about July and November. The above trends showing a decline in marginal increments to occur only once in a year indicate that the first eight opaque zones in sectioned otoliths of *P. bassensis* are formed annually. Given that the same trend was displayed in the data pooled for the ninth and subsequent opaque zones, these results indicate that zones are also formed annually.

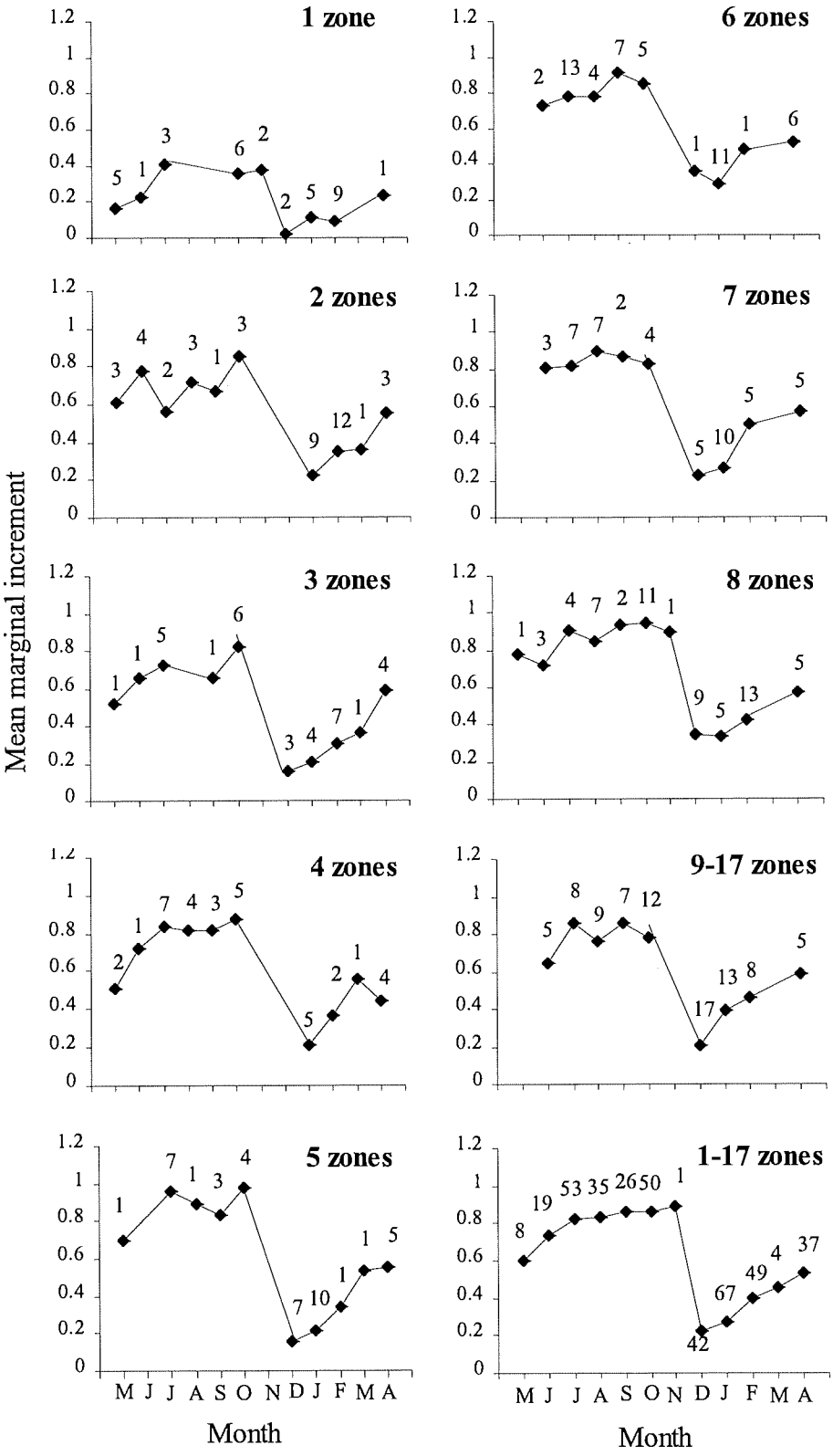



Fig. 5.9.20 Monthly trends in mean marginal increment (μm) for sagittal otoliths of *Platycephalus bassensis*. Value labels are sample size. Error bars are standard error.

*5.9.5.4 Precision of age estimates*

To compare the precision of age estimation, a random subsample of 275 sagittae were read a second time by the main reader, and a second subsample of 100 sagittae by a second reader. The index of APE calculated for repeat readings by the main reader was 0.70 % indicating a high consistency of similarity between readings. This is reflected in the distributions of differences revealing that around 84% of first and second readings were the same (Fig. 5.9.21). There was no indication of a skewed distribution that would result from consistently assigning higher or lower estimates on the second reading. The index of APE for estimates between the main and second reader was slightly higher at 2.35% reflecting less consistency between readers. Age estimates were the same 46% of the time, with clear evidence of the second reader overestimating age by one year in 42% of all fish.

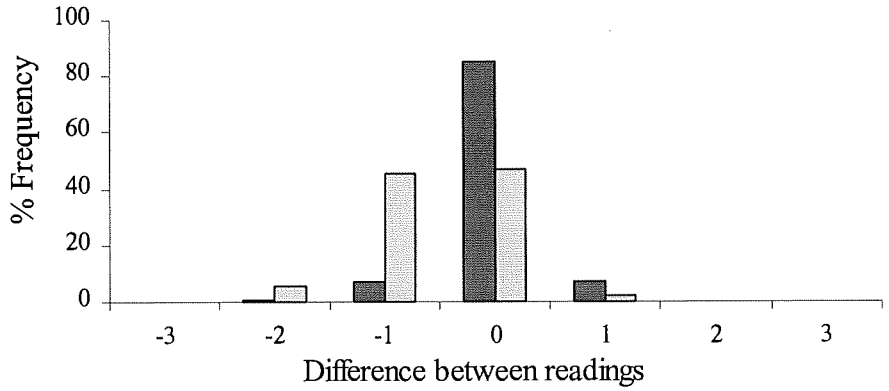



Fig. 5.9.21 Distribution of differences in estimated ages of *Platycephalus bassensis* for repeat readings by the same reader (dark bars) and a second reader (light bars).

*5.9.5.5 Growth*

Von Bertalanffy growth curves were fitted to male and female individual length-at-age data separately and combined (Fig. 5.9.22). Growth curves were found to be significantly different between males and females ($F=39.9$, df 3,591, $P<0.001$). The von Bertalanffy growth parameters are presented in Table 5.9.9. The respective asymptotic lengths ($L_\infty$) for males and females were 36.6 cm and 40.5 cm respectively. Mean lengths-at-age for males and females estimated separately, and combined are presented in Table 5.9.10. The mean length of females is consistently higher than that of males for all age-classes up to 16 years, the oldest age-class consisting of only one fish. There was a broad range of lengths within individual age-classes with a maximum of 7 age-classes present in a 1 cm size-class. Growth is rapid until around 3 years old and 22-25 cm and then slows appreciably. Maximum ages for males and females were 17 and 16 years old respectively.

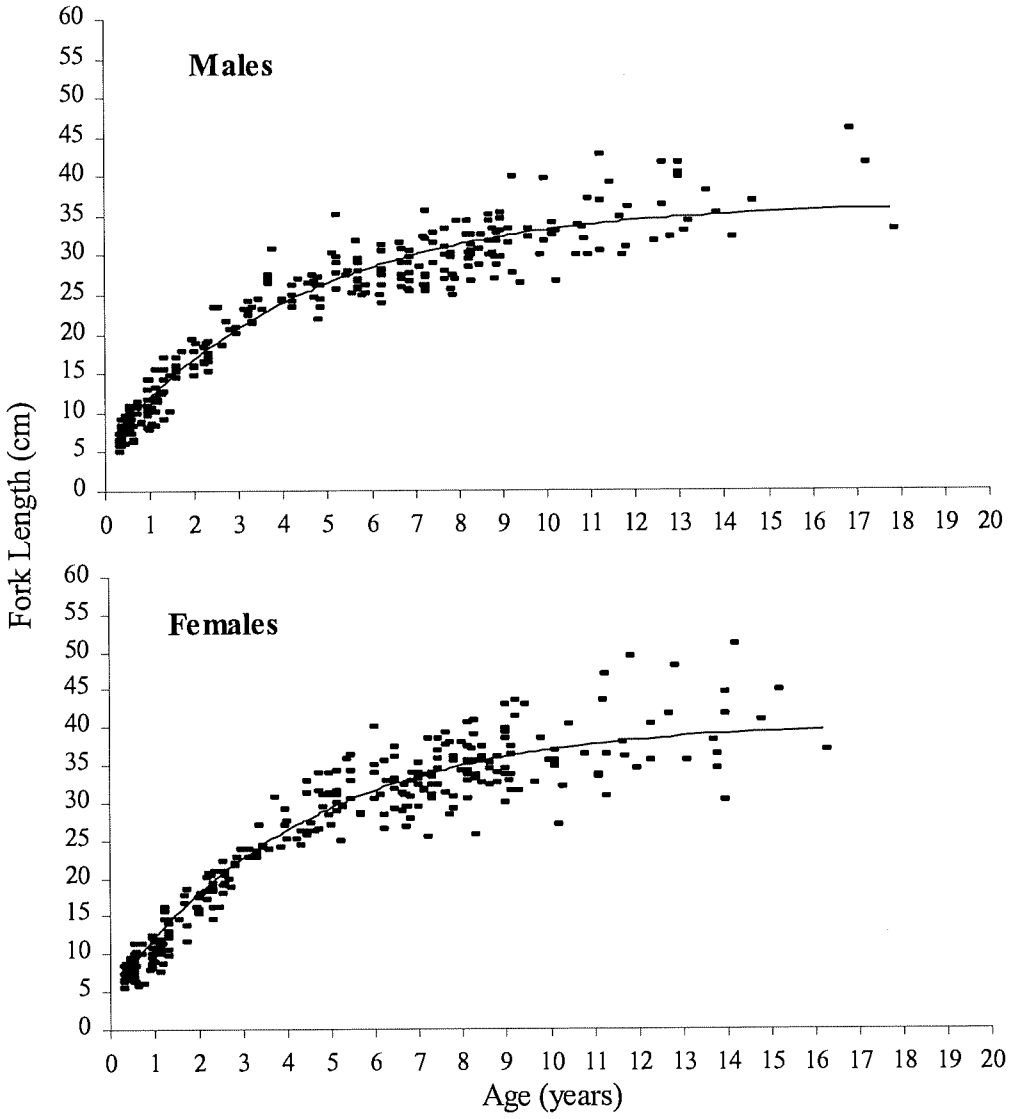



Fig. 5.9.22 Von Bertalanffy growth curves for male and female *Platycephalus bassensis* from southern and eastern Tasmania.

Table 5.9.9 Von Bertalanffy growth parameters derived from length at age data for *Platycephalus bassensis* from southern and eastern Tasmania.

| | | Von Bertalanffy growth parameters | | | | | |
|---|---|---|---|---|---|---|---|
| | n | $L_\infty$ | s.e. | K | s.e. | to | s.e. |
| All | 597 | 38.46 | 0.57 | 0.23 | 0.01 | -0.63 | 0.07 |
| Females | 307 | 40.45 | 0.78 | 0.23 | 0.01 | -0.52 | 0.08 |
| Males | 290 | 36.60 | 0.72 | 0.22 | 0.01 | -0.79 | 0.09 |

Table 5.9.10 Mean lengths at age for male and female *Platycephalus bassensis* separately and combined from southern and eastern Tasmania

| Age | Females - Males - Juveniles | | | Females - Juveniles | | | Males - Juveniles | | |
|---|---|---|---|---|---|---|---|---|---|
| | n | Mean | s.d. | n | Mean | s.d. | n | Mean | s.d. |
| 0 | 119 | 8.84 | 1.87 | 119 | 8.84 | 1.87 | 119 | 8.84 | 1.87 |
| 1 | 63 | 13.74 | 2.90 | 48 | 12.99 | 2.73 | 52 | 13.23 | 2.81 |
| 2 | 45 | 19.37 | 2.28 | 30 | 19.38 | 2.30 | 19 | 18.68 | 2.45 |
| 3 | 33 | 24.83 | 2.37 | 16 | 25.27 | 2.39 | 17 | 23.41 | 2.35 |
| 4 | 35 | 27.39 | 3.02 | 19 | 28.96 | 3.06 | 16 | 25.53 | 1.61 |
| 5 | 42 | 29.72 | 3.69 | 19 | 31.02 | 3.51 | 23 | 27.82 | 2.65 |
| 6 | 53 | 30.05 | 3.12 | 30 | 32.54 | 2.85 | 23 | 28.12 | 2.31 |
| 7 | 53 | 31.61 | 3.94 | 28 | 33.71 | 3.42 | 25 | 29.25 | 3.08 |
| 8 | 64 | 33.43 | 3.53 | 35 | 35.11 | 3.49 | 29 | 31.40 | 2.32 |
| 9 | 22 | 34.74 | 4.73 | 12 | 36.45 | 4.43 | 10 | 32.69 | 4.43 |
| 10 | 19 | 33.45 | 3.38 | 8 | 34.76 | 3.91 | 11 | 32.49 | 2.74 |
| 11 | 18 | 36.94 | 5.61 | 10 | 38.29 | 6.24 | 8 | 35.26 | 4.51 |
| 12 | 11 | 39.20 | 4.73 | 4 | 41.40 | 5.20 | 7 | 37.94 | 4.33 |
| 13 | 11 | 36.58 | 3.93 | 7 | 37.29 | 4.68 | 4 | 35.35 | 2.09 |
| 14 | 4 | 40.38 | 7.99 | 2 | 46.05 | 7.14 | 2 | 34.70 | 3.39 |
| 15 | 1 | 44.80 | | 1 | 44.80 | | 0 | | |
| 16 | 2 | 41.45 | 6.58 | 1 | 36.80 | | 1 | 46.10 | |
| 17 | 2 | 37.75 | 6.01 | 0 | | | 2 | 37.75 | 6.01 |

The relationship between length and weight was examined for males and females with the slopes of the regression of log weight against log length showing no significant difference (ANCOVA, F=2.919, df 1,1204, P>0.1). Given a common slope, there was no significant difference in the intercepts for the two sexes (ANCOVA, F=0.714, df 1,1205, P>0.1). Hence, both sexes and juveniles were combined to produce the relationship between fork length and weight shown in Fig. 5.9.22 and Table 5.9.11.

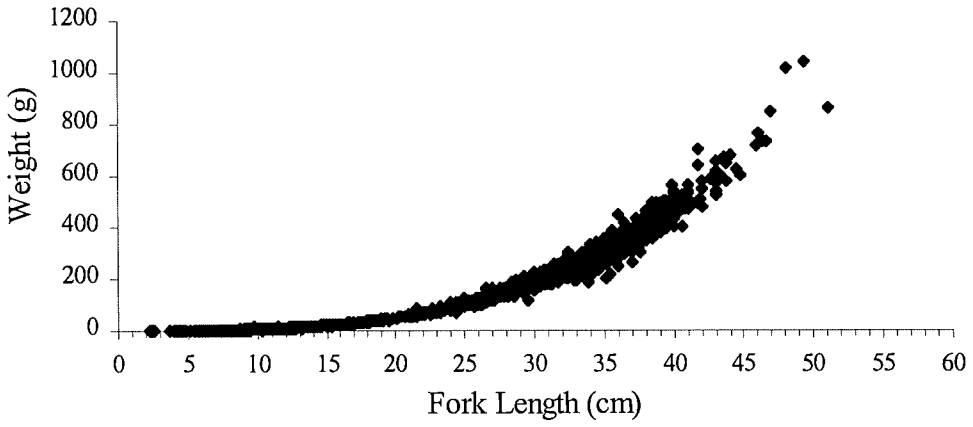



Fig. 5.9.22 Relationship of fork length against weight for *Platycephalus bassensis* from southern and eastern Tasmania.

The relationship between otolith weight and age was examined separately for male and female *P. bassensis* from all years (Fig. 5.9.23). An examination of the distribution of residuals from the linear regression shows that variance in otolith weight increased with age for both sexes, thereby violating the assumption of homogenous variances. The problem of heteroscedasticity was best solved by logarithmic transformation of otolith weight and age, with the residual plots showing no increase in variance with age. The regression of log otolith weight against log age were significantly different for males and females (ANCOVA, F 8.186, df 1,329, $P<0.01$). Hence, the relationship between otolith weight and age was calculated separately for males and females (Table 5.9.11).

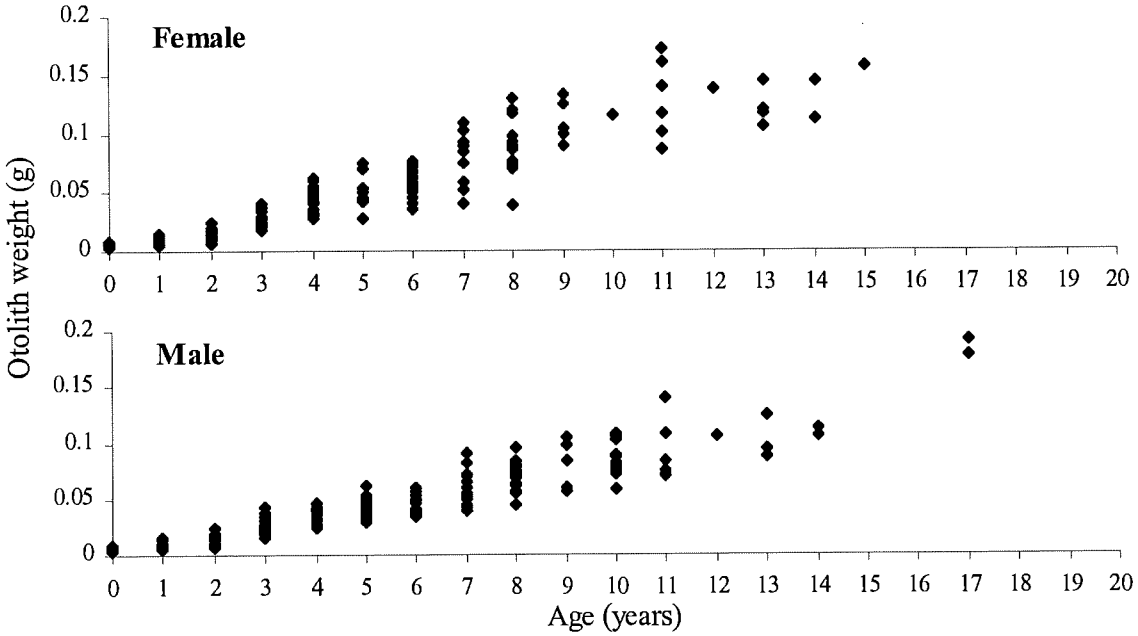



Fig. 5.9.23 Relationship of otolith weight against age for female and male *Platycephalus bassensis* from southern and eastern Tasmania.

Table 5.9.11 Length (FL)-weight (WT) and otolith weight (OT)-age regressions for *Platycephalus bassensis* from southern and eastern Tasmania.

| | | | Y = a+bX | | |
|---|---|---|---|---|---|
| Y | X | n | a | b | $r^2$ |
| $\log_{10}$WT | $\log_{10}$FL | 1743 | -2.479 | 3.207 | 0.99 |
| Female $\log_{10}$OT | $\log_{10}$AGE | 170 | -5.367 | 2.754 | 0.95 |
| Male $\log_{10}$OT | $\log_{10}$AGE | 203 | -5.505 | 2.874 | 0.94 |

*5.9.5.6 Age composition*

The age composition of the inshore *P. bassensis* population was estimated from the 1995 samples, with the number of fish aged proportional to the number in each 2 cm size-class

from the size composition of the total population in that year. A maximum of 16 age-classes of females and 15 age-classes of males occurred in the samples, dominated by 2 to 10 year old fish, which made up 88% of the sample (Fig. 5.9.24). There was no significant difference in the age composition of males and females (KS test, P>0.5). The high proportion of 8 year olds in the sampled population, representing the 1986 year-class, indicates that strong recruitment occurred in that year. As increased selectivity of *P. bassensis* occurs at around 28 cm, or 4-5 years old, the strong year-class does not represent selectivity factors. There is also evidence of the 1989 year-class (5 year olds) being weak.

The distribution of 0+ fish and unsexable 1+ fish appear to be restricted to inshore waters and are not included in this analysis, primarily as selectivity of 1+ and 2+ males and females in gillnets results in undersampling of those age-classes. While the inclusion of beam trawl samples would tend reduce this bias, the catchability of these age-classes by beam trawl is uncertain.

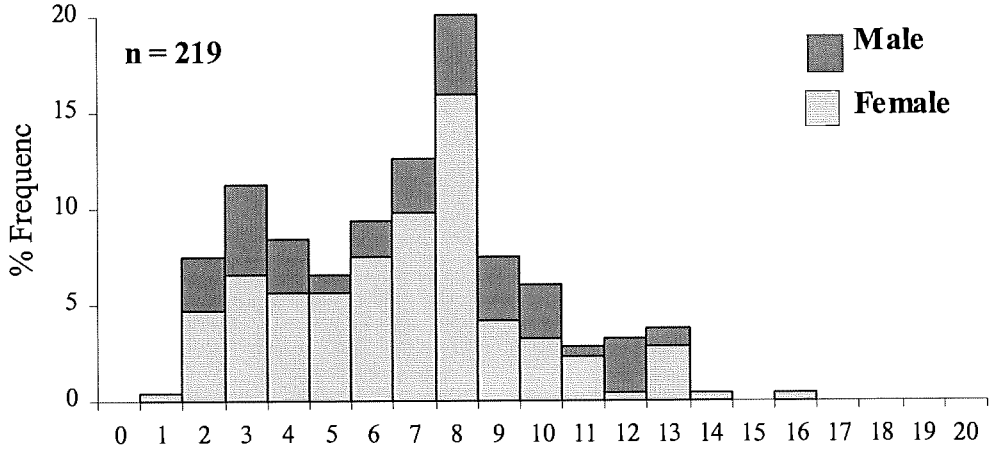



Fig. 5.9.24 Age composition of male and female *Platycephalus bassensis* from inshore waters of southern and eastern Tasmania in 1995. n is sample size.

## 5.9.6 Discussion

### *5.9.6.1 Size at maturity*

In the present study, the size at 50% maturity for male and female *Platycephalus bassensis* was 21.0 and 23.5 cm, respectively, representing fish in the 2+ age-class. While this is consistent with the size at maturity of *P. bassensis* in Port Phillip Bay, Victoria of around 20-21 cm (Brown 1978), this length was determined by comparing GSI and fork length, and hence represents the lower size limit. The difference between the smallest and largest *P. bassensis* to reach maturity (males 19.0-24.5 cm and females 20.0-29.5 cm), indicates a broad range of sizes and ages at which sexual maturity may occur. Such variations in body size at first sexual maturity are common to fishes (Nikolskii 1969), and may be related to

variations in size at age resulting from differences in juvenile growth rate or extended spawning seasons.

*5.9.6.2 Temporal and spatial patterns of spawning*

The presence of ripe, running ripe and spent *P. bassensis* (≥ stage 5) from October to March clearly demonstrates that in southern and eastern Tasmania, spawning occurs over an extended period lasting up to six months. However, the increased number of fish with resting stage gonads from January to March indicates that the bulk of spawning occurs between October and December, with a lower level of spawning activity in the latter half of the spawning period. The high GSI's in those months also suggests that spawning peaks between October and December.

Spawning commenced soon after water temperature rose in October, and may be linked to the timing of the spring bloom in productivity in these waters which begins around mid-September (Harris *et al.* 1987). However, given that the duration of the spring bloom in the shelf waters of southern and eastern Tasmania can vary by as much as three months from year to year (Harris *et al.* 1991), the extended spawning period may also be a strategy to maximise the number of larvae encountering suitable feeding conditions.

In contrast, despite monthly sampling through the year the trend in GSI's in *P. bassensis* in Port Phillip Bay indicated spawning occurred between August and October (Brown 1978). While some female fish with ripe ovaries were caught as late as December, Brown (1978) suggested that these fish did not spawn, but reabsorbed their gonads. However, the timing and duration may vary from year to year as no platycephalid larvae were caught during August or September during ichthyoplankton surveys of Port Phillip Bay (Jenkins 1986). While identification was only to family level, the fact the *P. bassensis* is the most abundant and earliest spawning flathead in Port Phillip Bay (Brown 1978), suggests that the 'type 1' larvae in Jenkins (1986) was most likely that of *P. bassensis*. The 'type 1' larvae were caught continuously from October through to April with abundances peaking in November and December. This spawning period is consistent to that found in the present study for *P. bassensis* in southern and eastern Tasmania.

The presence of running ripe *P. bassensis* in all areas sampled strongly indicates that spawning is widespread throughout inshore waters of Tasmania. This is supported by the presence of small preflexion larvae in Norfolk Bay at stations furthest from the entrance to the bay. By spawning throughout coastal and estuarine waters close to settlement habitats, and by having larvae concentrated in mid-water, *P. bassensis* have developed spawning and early life-history strategies that minimise the advective loss of larvae offshore that could occur during periods of increased westerly winds which results in a flow of surface waters off the shelf. In the present study, there was also no evidence of large scale replacement of waters in Norfolk Bay that would transport larvae away from suitable settlement areas.

Variations in larval transport have been identified as a significant source of mortality and can play a major role in determining recruitment success (Nelson *et al.* 1977, Bailey 1981). This is particularly significant in species whose larvae are distributed inshore and whose nursery areas are also inshore. While such offshore advective losses of *P. bassensis* larvae spawned in coastal embayments (eg. Norfolk Bay) and estuaries (eg. Georges Bay) are unlikely, the significance of shelf spawning to overall egg production is yet to be determined.

*5.9.6.3 Larval development*

Development of larvae of *P. bassensis* is similar to that described for other platycephalid larvae off southern Australia, *P. speculator* (Hyndes *et al.* 1992a), and *P. fuscus* (Neira and Miskiewicz 1998) They are characterised by a large and wide head with extensive spination, moderate to large, fan shaped pectoral fins and 26-28 myomeres. However, *P. bassensis* larvae are distinguished from both *P. fuscus* and *P. speculator* by the larger size at both notochord flexion (6.0->8.4 mm) and pelvic (5.9-7.4 mm) and dorsal fin (6.2->8.4 mm) formation. In addition, the trunk and tail was only lightly pigmented in *P. bassensis* larvae, which contrasts the moderate to heavy pigment in larval *P. fuscus* and *P. speculator* (Neira and Miskiewicz 1998).

Small platycephalid larvae can be confused with scorpaenids and triglids that also have early developing fan-shaped pectoral fins and extensive spination. However, triglid larvae have more prominent posttemporal spines, a duck-bill shaped snout, 27-37 myomeres and lower two or three pectoral fin rays elongate and detached from the rest of the fin in larger larvae (Jordan *et al.* 1998). Small scorpaenid larvae have a rounder head without a flattened, elongate snout, while larger larvae are easily distinguished by morphology, fin meristics and the presence of a single dorsal fin (Neira and Furlani 1998).

*5.9.6.4 Recruitment*

The size-class of juvenile *P. bassensis* sampled between March and November 1996 represents the 0+ cohort resulting from spawning that commenced the previous October. There was a broad range of lengths in every month indicating that settlement occurred over an extended period reflecting the extended spawning period in *P. bassensis*. Newly settled *P. bassensis* were first caught in January, although the presence of fish up to 7 cm in that month suggests that initial settlement occurs some time earlier. The smallest new recruit was 2.1 cm, suggesting settlement to benthic habitats occurs close to this size. This is larger than the size at settlement of approximately 1.3 cm in *P. speculator* (Hyndes *et al.* 1992a), and may reflect selectivity of the beam trawl. However, the fact that the cod-end mesh size used by Hyndes *et al.* (1992a) was 9.5 mm, compared to 7.0 mm in the present study indicates that selectivity alone probably does not account for the lack of *P. bassensis* < 2.1 cm. The lack of smaller recruits may also reflect the fact that initial settlement does not occur into subtidal unvegetated habitats.

Length-frequency distributions suggest the presence of two cohorts of 0+ *P. bassensis* in most months. The existence of multiple 0+ cohorts may reflect periodicity in the temporal pattern of spawning (Szedlmayer *et al.* 1990, Jordan 1994b), variability in larval supply (Jenkins and Black 1994), and larval duration (Cowen 1991, Jenkins and May 1994), or a combination of factors. While the monthly distribution of GSI's show no indication of distinct peaks in spawning, such monthly sampling may miss finer temporal patterns. It is clear that further work is needed to resolve the otolith microstructure of *P. bassensis* before the influence of temporal patterns of spawning and variations in larval growth rates and duration on the recruitment processes in this species can be evaluated.

*5.9.6.5 Temporal and spatial patterns of abundance*

Distinct seasonal variations in abundance of *P. bassensis* were apparent in all inshore regions of southern and eastern Tasmania. In Norfolk Bay, abundances were consistently highest in spring and summer and lowest in winter, while in Georges Bay and Prosser Bay abundances peaked in spring and were lowest in winter. The low abundances inshore in winter contrast the high abundance on the shelf at that time (Jordan 1997), suggesting that throughout southern and eastern Tasmania mature *P. bassensis* move from inshore waters onto the shelf at the end of autumn. The shift in distribution appears to be unrelated to the seasonal decrease in water temperature, which varies little between inshore and shelf regions. However, prey abundance in inshore soft-sediment habitats have been shown to be at their lowest during winter (Edgar and Shaw 1995a), suggesting the movement may be in response to decreased food availability inshore. However, the lack of data on seasonal variations in prey abundance on the shelf precludes an assessment of its influence in determining seasonal patterns of *P. bassensis* distribution.

In the present study, adult *P. bassensis* were found to be common in both unvegetated and *Heterozostera* habitats, with relative abundances changing through time reflected by the significant habitat and time interaction in all three areas. Distinct habitat differences were apparent only in spring, with abundances higher in *Heterozostera* compared to unvegetated habitats in Norfolk Bay and Georges Bay, whereas in Prosser Bay abundances were higher in unvegetated habitats. These differences are possibly related to the differences in the habitat characteristics of the inshore regions, with beds of *Heterozostera* in Prosser Bay only small and sparse, which is related both to the high degree of exposure of the bay and the significant loss of *Heterozostera* beds that has occurred over the past 20-30 years (Rees 1993). Such losses have not been apparent in Georges Bay and Norfolk Bay (Rees 1993). This contrasts other studies where adult *P. bassensis* have shown no strong association with seagrass habitats (Last 1983, Edgar and Shaw 1995a).

While little habitat preference was identified for adult *P. bassensis*, significant habitat differences were found for juveniles, with abundances consistently higher in unvegetated compared to *Heterozostera* habitats. The low abundance of juveniles in the shallow nearshore beach habitat suggests that the unvegetated subtidal zone is a more significant

nursery area for the species. This is supported by the low abundance of *P. bassensis* in shallow beach habitats during extensive beach seine surveys of Tasmania (Last 1983). A number of studies have identified unvegetated habitats as a nursery areas for platycephalids (Bell *et al.* 1984, Hyndes *et al.* 1992a, Edgar and Shaw 1995a, Ayvazian and Hyndes 1995, Jenkins *et al.* 1996). The significance of unvegetated habitats as a nursery area for temperate Australia platycephalids is also supported by the lack of juveniles in vegetated habitats, despite extensive surveys of both seagrass (Burchmore *et al.* 1984, Ferrell *et al.* 1993, Bell and Westoby 1986b) and reef-algal beds (Jenkins *et al.* 1996). The preference for unvegetated habitats by juvenile *P. bassensis* in the present study is consistent with results from studies in Western Port, Victoria (Edgar and Shaw 1995a) and Tasmania (Last 1983). It is likely that *P. bassensis* use unvegetated habitats as a nursery area, as camouflage allows them some protection from predators. In addition, while benthic invertebrate production is generally higher in seagrass beds (Edgar 1990, Edgar *et al.* 1994), enhanced food production in unvegetated habitats can occur through regular phytoplankton blooms (McLachlan *et al.* 1981) and the presence of detached macrophytes (Robertson and Lenanton 1984, Shaw and Jenkins 1992). Similar patterns of recruitment to unvegetated habitats is common in families such as Pleuronectidae (flounders) that are also protected by camouflage (Crawford 1984, Connolly 1994, Jenkins *et al.* 1996).

Initial settlement and growth of *P. bassensis* was found to occur exclusively in subtidal unvegetated habitats, with some post-settlement movement into beds of *Heterozostera* occurring around 7 cm (approximately 3-5 months old). However, there was no indication of a ontogenetic habitat shift in juvenile *P. bassensis*, as only small numbers of 0+ and 1+ fish were found in *Heterozostera* beds. The present study supports the findings of Edgar and Shaw (1995a), who found no indication of a change in habitat preference with growth in juvenile *P. bassensis* in Western Port, Victoria. Size dependant shifts in habitat, however, are common to many species that initially recruit to both seagrass beds (Robertson 1977, Middleton *et al.* 1984, Love *et al.* 1991, Worthington *et al.* 1992) and sandy beaches (Bennett 1989, Hyndes *et al.* 1996). While a complete shift in habitat is not apparent in *P. bassensis*, close to maturity they increasingly utilise both *Heterozostera* beds and inner- and mid-shelf waters (Jordan 1997).

#### *5.9.6.6 Age validation*

For validation to be considered complete annual periodicity of increment formation must be established on otoliths with differing numbers of increments (Beamish and McFarlane 1983). Trends in marginal increments have been commonly been used to establish that increments are formed annually (Beckman *et al.* 1989, Massey and Horn 1990, Hyndes and Potter 1996). In the present study, annual trends in marginal increments of sagittae of *P. bassensis* were consistent regardless of the number of opaque zones, confirming that one increment is formed each year. The consistent decrease in marginal increments in December indicates that translucent material starts to form in early summer. The trend in

marginal increments in *P. bassensis* are consistent with that observed in sagittae of *P. speculator* where one increment was formed each year, irrespective of the number of increments (Hyndes *et al.* 1992b). However, increment formation in *P. speculator* is complete in early spring, some months earlier than *P. bassensis*. The difference in timing may be related to the later increase in water temperatures in southern and eastern Tasmania. There was no indication of variability in the timing of annuli formation with increasing age.

*5.9.6.7 Growth*

The monthly progression of the 0+ cohort indicates rapid growth during summer and autumn (Jan-May) when water temperatures are at a maximum. Growth then slows appreciably during winter and spring (June-Nov) to reach approximately 7-13 cm after one year. The absence of a distinct opaque zone in the otoliths of this cohort is consistent with the conclusion that this represents the 0+ age-class. A single opaque zone became discernible in otoliths of this cohort by December, now the 1+ age-class. This cohort progressed to a mean length of around 17 cm by the following December when otoliths possessed two opaque zones, therefore representing 2+ fish. These mean lengths-at-age for 1+ fish are consistent with that of *P. bassensis* from Port Phillip Bay as defined from modal progressions, but are considerably larger than the 15 cm defined for 2+ fish (Brown 1978). The smaller size of 2+ fish in Port Phillip Bay can be attributed to the fact that sampling of this age-class was restricted to winter before the period or faster growth in late spring and early summer. Such growth rates are considerably lower and less variable than those of the corresponding age for *P. speculator* which reach 19-31 cm after one year and 21-40 cm after two years (Hyndes *et al.* 1992b). The inclusion of considerable numbers of juveniles the values for $t_0$ for males and females were -0.52 and -0.79 respectively, indicating that the von Bertalanffy growth curve is a reasonable representation of growth of juvenile *P. bassensis*.

Growth of male and female *P. bassensis* is relatively rapid for the first 3 years, slowing appreciably at around 22-25 cm which is consistent with the size at maturity. After 3-4 years there was an increasing variation in size-at-age with fish at the minimum legal length of 30 cm ranging from 4 to 11 years old. Females are larger than males at corresponding ages, with growth curves diverging with increasing age and maturity. As there was little difference in the age composition between sexes, the larger female size can be attributed to significantly higher growth rates and not greater longevity. A larger female size appears to be a life-history strategy to increase reproductive potential through increased fecundity in larger fish.

The maximum ages of 17 years for males and 16 years for females found in the present study is significantly higher than that of 7 years for males and 9 years for females reported for *P. bassensis* from Victorian waters (Brown 1978). The lower maximum ages may reflect either spatial variations in the age structure or underestimates of age in older fish

due to the use of whole otoliths by Brown (1978). Firstly, the presence of older fish in inshore waters in the present study indicates that such age-classes are not restricted to shelf waters, and are likely to occur in Port Phillip Bay. Secondly, a comparison of whole and sectioned sagittae in *P. speculator* found whole otoliths underestimated age by as much as six years in old fish (Hyndes *et al.* 1992b). This is consistent with previous studies where ages estimates were lower from whole otoliths compared to those sectioned or broken and burnt (Beamish 1979, Campana 1984, Collins *et al.* 1988). The consistent marginal increment trend in fish aged 9-16 also supports the maximum ages found in the present study. The use of whole otoliths and smaller representation of juveniles in the study of Brown (1978) also resulted in considerable differences in the von Bertalanffy growth parameters compared to those in the present study.

*5.9.6.8 Age Composition*

A maximum of 16 age-classes of *P. bassensis* were present in inshore waters of southern and eastern Tasmania, dominated by 2 to 10 year old fish, which made up around 88% of the population. There was clear evidence of variable recruitment in the population of *P. bassensis,* with the 1986 year-class dominant. However, the relative abundance of the youngest age-classes will be influenced by the decreased catchability of small fish to the particular sampling gear. The data strongly suggests that there were few 5+ fish present in 1995, representing the 1989 year-class. While there was evidence of selectivity in the gillnets of the fish <28 cm, the low abundance of the 1989 year-class is unlikely to be biased given the mean length-at-age of 5+ male and female *P. bassensis* is 28 cm and 32 cm, respectively. This is further supported by the higher proportion of younger age-classes in the gillnet samples in that year despite their lower catchability.

## 5.10 Southern sea garfish (*Hyporhamphus melanochir*)

### 5.10.1 Introduction

Southern sea garfish (*Hyporhamphus melanochir*) inhabit coastal waters and estuaries of southern Australia from Eden (N.S.W.) to Perth (W.A) (Gomon *et al.* 1994). The species supports valuable commercial and recreational fisheries throughout southern Australia. In Tasmania, the commercial catch rose to around 50-60 tonnes p.a during the 1980's and has traditionally been taken by beach seine, although alternative methods have recently been used such as lampara/purse seine, pushnets and dipnets. Landings in 1997 were around 87 tonnes with 70% of the catch taken by beach seine (Lyle 1998). Catches are concentrated along the south-east and north coasts, particularly around Flinders Island. Despite the importance of this species to Tasmania's scalefish fishery, biological studies have been mainly limited to South Australian waters (Ling 1958, Jones 1990, Klumpp and Nichols 1983). However, aspects of reproduction, diet, morphometrics and age and growth for *H. melanochir* from eastern Tasmania have recently been examined by St. Hill (1996).

Given the lack of information on the reproductive biology, early life-history, size composition and age and growth of *H. melanochir* from Tasmanian waters, the aim of this chapter is to (1) examine spatial and temporal patterns of spawning, (2) describe the egg development, (3) examine the size composition of commercial landings, and (4) determine the age and describe the growth of *H. melanochir* from sectioned sagittal otoliths.

### 5.10.2 Methods

#### *5.10.2.1 Survey area and sampling regime*

*Hyporhamphus melanochir* were sampled from the commercial fishery in north and east coast regions of Tasmania to obtain information on the size composition of landings (Fig. 5.10.1). Fish were sampled monthly, depending on the availability of fish, from the north coast between May and October 1995 and the east coast between May 1996 and February 1997. Fish from the north coast were taken primarily by beach seine while those from the east coast were caught using dipnets. A random sample of a minimum of 200 individuals were measured for fork length (FL) to the nearest half centimetre. Up to 50 fish from each sampling period from the east coast were also frozen and processed later for biologicals in the laboratory.

In addition, research sampling of juvenile and adult *H. melanochir* was conducted in the northern part of Great Oyster Bay adjacent to Swansea monthly from September 1996 to January 1997 (Fig. 5.10.2). Fish were sampled with surface set 30 m long multi-panel gillnets comprising three randomly placed 10 m panels of increasing gillmesh size (28, 36 and 48 mm). Two multi-panel gillnets were set from 1 to 4 hrs at several sampling sites in depths of 3-6 m. The gillnets were buoyed at both ends and anchored at one end with a 1.5 kilogram lead weight. Juvenile and adult *H. melanochir* were also caught at night by dipnet adjacent to the gillnet sampling sites. Running ripe male and female *H. melanochir*

were obtained for stripping between mid October and mid January. As *H. melanochir* tended to school by sex, it was often necessary to sample fish from several sites within the northern part of Great Oyster Bay within a night to capture both sexes.

In the field, eggs and sperm from running ripe *H. melanochir* were stripped into 2 litre aerated glass jars. Most jars contained an artificial substrate for eggs to attach to, while a jar with no substrate was included to assess the importance of attachment to egg development. The substrate provided was 1.5 cm wide strips of onion bag mesh which were weighted with small stainless steel shackles. All jars were held in a water filled eski, to stabilise temperature. Eggs were returned to the laboratory for rearing experiments.

In order to further examine the distribution of spawning, *H. melanochir* eggs were sampled from the north western part of Great Oyster Bay through a depth stratified survey (Fig. 5.10.2). The sampling area consists of a unvegetated sand embayment with varying amounts of filamentous drift algae. Sampling was conducted monthly from September 1996 and ceased when no further eggs were caught in February 1997. The survey area was stratified into 3 depth strata (2-5, 5-8 and 8-11 m). Eggs were sampled in each strata with a beam trawl with an opening of 2.0 x 0.9 m. Full details of gear design is presented in Chapter 4. In each stratum, three non-overlapping randomly placed one minute trawls were conducted at a tow speed of 2 knots. The volume of algae in the net (litres. $tow^{-1}$) was estimated and the entire contents retained for later sorting of *H. melanochir* eggs. In addition, beam trawl samples from *Heterozostera* sites in Lime Bay and Sommers Bay within Norfolk Bay (see Fig. 4.2), were examined for *H. melanochir* eggs from samples taken in October and December 1996. All sampling was conducted during daylight hours. Beam trawl catch rates were calculated as the number of eggs per tow.

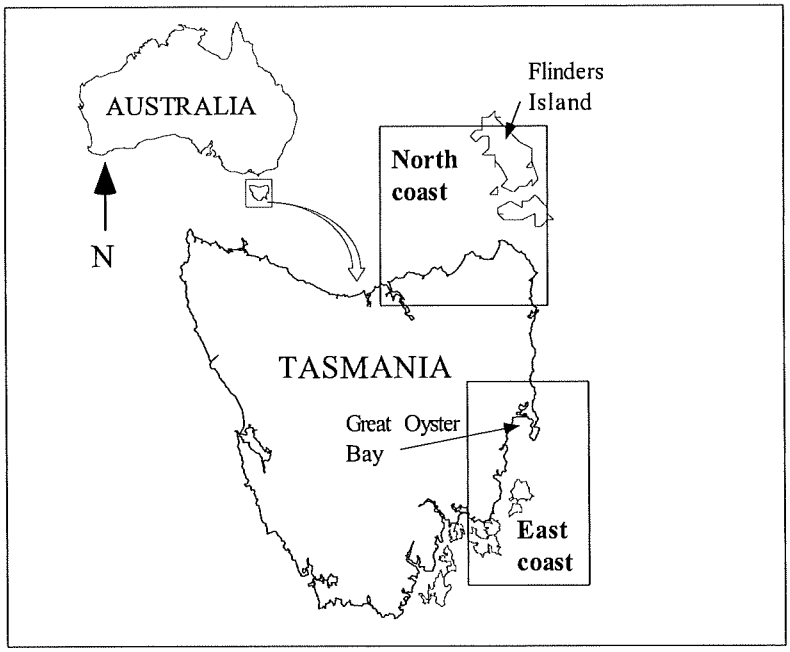



Fig. 5.10.1 North and east coast sampling regions for *Hyporhamphus melanochir* from the commercial fishery in Tasmania.

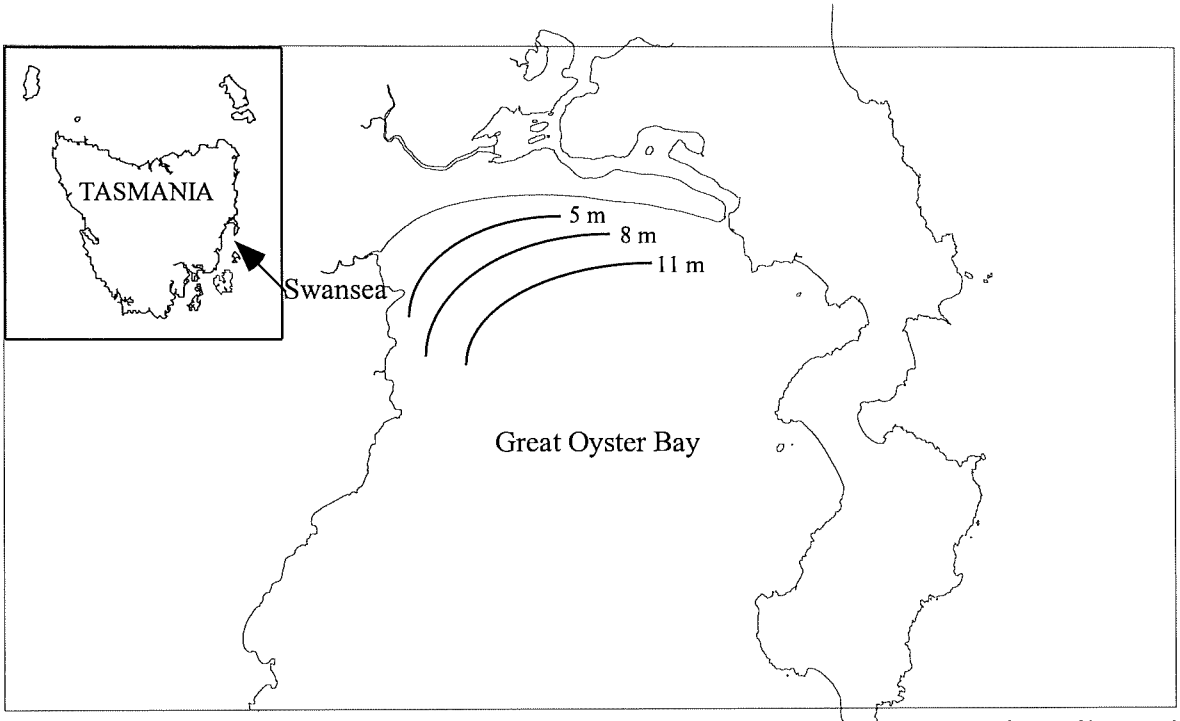



Fig. 5.10.2 Sampling area and depth strata for *Hyporhamphus melanochir* adults, juveniles and eggs in Great Oyster Bay, eastern Tasmania.

*5.10.2.2 Laboratory analysis*

All juvenile and adult *H. melanochir* from research sampling and a random subsample of 50 adults from each commercial sample were retained and processed for biologicals including fork length (FL) (to the nearest millimetre), total weight (to the nearest gram), sex, gonad stage and gonad weight (to the nearest gram). Gonads were staged macroscopically (see Table 4.5), and sagittal otoliths were removed from all fish.

*H. melanochir* eggs were reared through to hatching in the same 2 litre containers they were fertilised in, with regular 50% water changes made. Temperature ranged from about 14.0 to 16.5° C, which is similar to bottom temperatures of that in shallow bays on the east coast of Tasmania during early summer. A subsample of eggs were removed and preserved in 10% formaldehyde at 12 hrs, then every 24 hrs until 5.5 days and then every 48 hrs until hatching.

Wild caught *H. melanochir* eggs from beam trawl sampling were staged in the laboratory based on development stages defined from rearing experiments. Backcalculated spawning dates of eggs that survived until the time of sampling were obtained by subtracting the estimated age in days from the calendar date of capture.

All unspecified body lengths of postflexion larvae refer to standard length (SL) (i.e. tip of the upper snout to the posterior region of the hypural plate). All other measurements are the same as that detailed for *P. bassensis* in section 5.9.2.2.

Otolith preparation, ageing procedures, age validation and growth analysis of juvenile and adult *H. melanochir* is that same as that detailed for *P. bassensis* in section 5.9.1.2. In brief, age was estimated from transversely sectioned sagittae by counting the presumed annual increments (opaque or dark zones) from the primordium to the edge of the otolith section on the ventral sector of the proximal side.

#### *5.10.2.3 Statistical analysis*

The abundance of *H. melanochir* eggs across depth strata was analysed by one-way analysis of variance (ANOVA). Data were tested for conformity to assumptions of ANOVA with the *f-max* test for heteroscedascity and by observing residual plots and log transformed where necessary. Differences between group means were tested with Ryans Q-test.

An absolute age was assigned to *H. melanochir* using a birth date of 1 December, which corresponds to the mid-point of the spawning season. Von Bertalanffy growth curves were fitted to the individual length-at-age data for males and females combined by direct non-linear least-squares estimation.

Using the estimated ages, mean lengths at age were calculated for juvenile, male and female *H. melanochir* combined, and the age composition of commercial landings from the east coast region estimated by applying the age length key to the length-frequency data as follows:

$A_t = S_x(L_x P_{tx})$ where:

$A_t$ = the estimated number of fish of age $t$ in the length-frequency sample
$L_x$ = the number of fish of length $x$ in the length-frequency sample
$P_{tx}$ = the proportion of aged fish of length $x$ which were aged $t$

### 5.10.3 Spawning and early life-history

#### *5.10.3.1 Gonadal development*

Trends in mean gonadosomatic (GSI) for male and female *H. melanochir* were analysed monthly from east coast regions from February 1996 to February 1997. Monthly mean GSI's showed the same overall trend for both males and females (Fig. 5.10.3). Mean GSI's rose from a low in May to a peak in December before declining through the next few months. There was no indication of two distinct spawning periods although the monthly sampling is likely to be too coarse to detect shorter periodicity in spawning activity. Gonads reached a maximum of 5.2% and 15.8% of total body weight for males and females, respectively, during the spawning season.

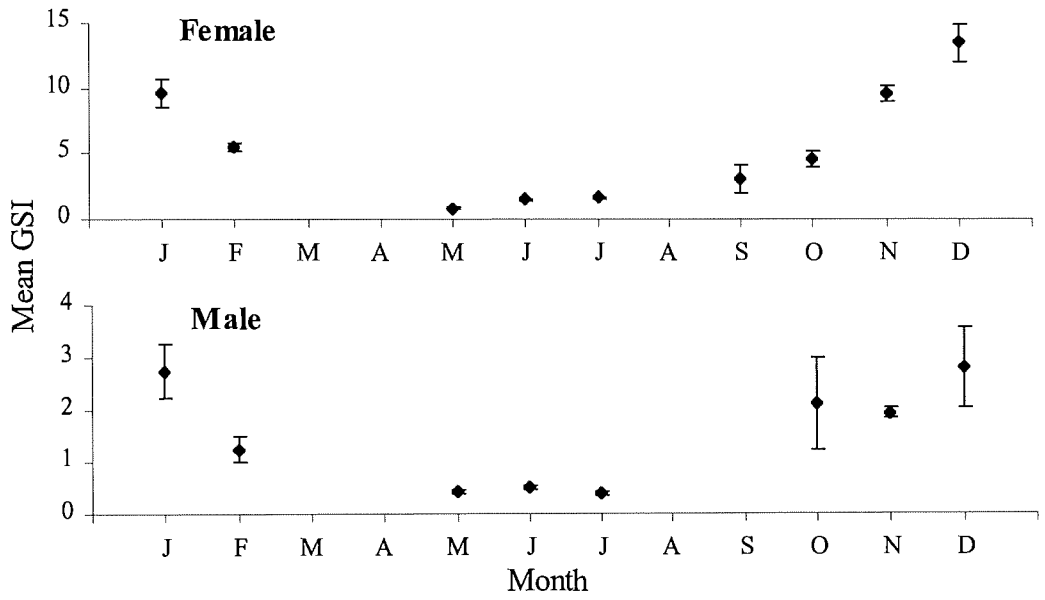



Fig. 5.10.3 Mean gonadosomatic indices (GSI) for male and female *Hyporhamphus melanochir* from the east coast of Tasmania. Error bars are standard error.

The temporal patterns of spawning from the GSI's is also reflected in the monthly trend in gonad stages, with most females in the resting phase (stage 2) from May to July, and ripe, running ripe and spent fish (≥ stage 5) from October to February (Fig. 5.10.4). No spent (stage 7) males were caught during the study. The decrease in GSI's through the spawning period reflects the increasing proportion of recovering (stage 2) fish. This suggests that an increasing proportion of the population completes spawning between December and February. The lack of samples in March precludes an assessment of the full duration of the spawning period.

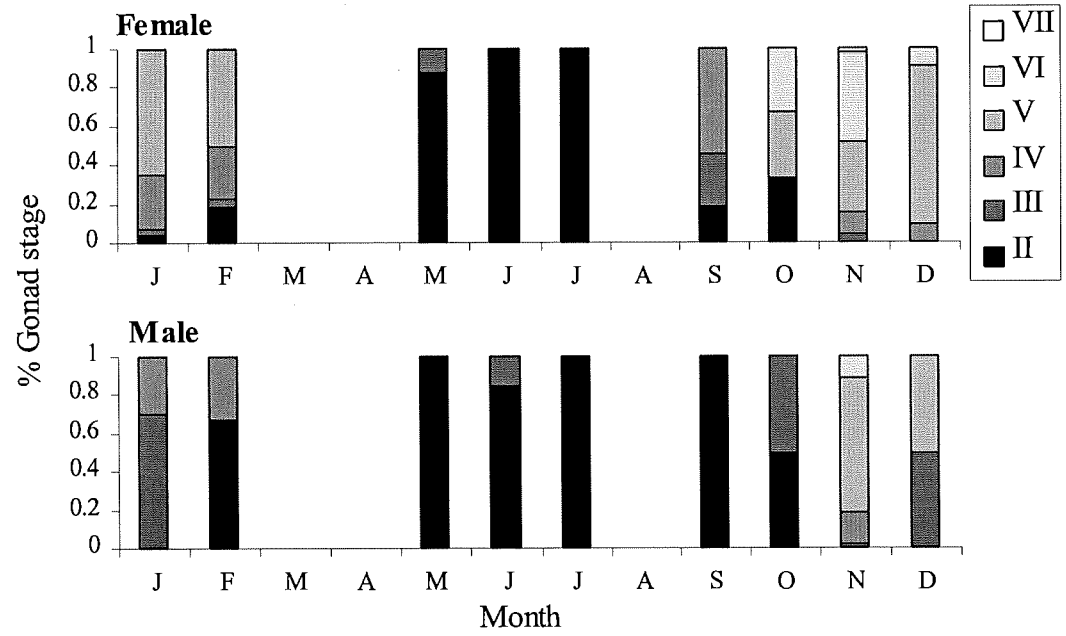



Fig 5.10.4 Monthly percentage of gonad stages for female and male *Hyporhamphus melanochir* from the east coast of Tasmania.

*5.10.3.2 Spawning distribution*

A total of 618 *H. melanochir* eggs were caught from 14 one-minute beam trawl tows across the three depth strata in Great Oyster Bay. Eggs were invariably found singly, with their choronic filaments heavily entangled in filamentous drift algae. Around 10% of eggs could not be staged due to cloudiness of the eggs. Eggs had a tendency to become cloudy if not removed from the algae within 12 hours of sampling. Around 4% of eggs were empty shells, presumably due to mortality or hatching.

Most eggs were caught in the two to five metre depth stratum (Fig. 5.10.5, Table 5.10.1), with abundances from this shallow stratum significantly higher than those from either of the deeper strata (Table 5.10.2). The observed distribution of eggs was unrelated to the volume of algae on which the eggs were attached, which was shown to increase with depth (Fig. 5.10.5). Another possible explanation for the distributions would be a higher rate of egg predation in deeper water. If this were occurring, the age distribution of eggs would vary across depth, with fewer late stage eggs being caught where predation was heavy. However, there was no difference in the age composition across the strata.

Despite sampling across depths from 2 to 8 m in *Heterozostera* sites in Norfolk Bay during the peak of the spawning period (Oct-Dec), no eggs were caught in beam trawl samples from this area. While *H. melanochir* are present in the bay, and are fished commercially during the summer period, the absence of eggs in samples suggests that spawning areas are either restricted to particular areas within the bay, or *Heterozostera* beds are not an important spawning habitat. As shallow unvegetated habitats similar to those in Great Oyster Bay were not sampled in Norfolk Bay, the significance of this habitat across a broader spatial scale cannot be assessed.

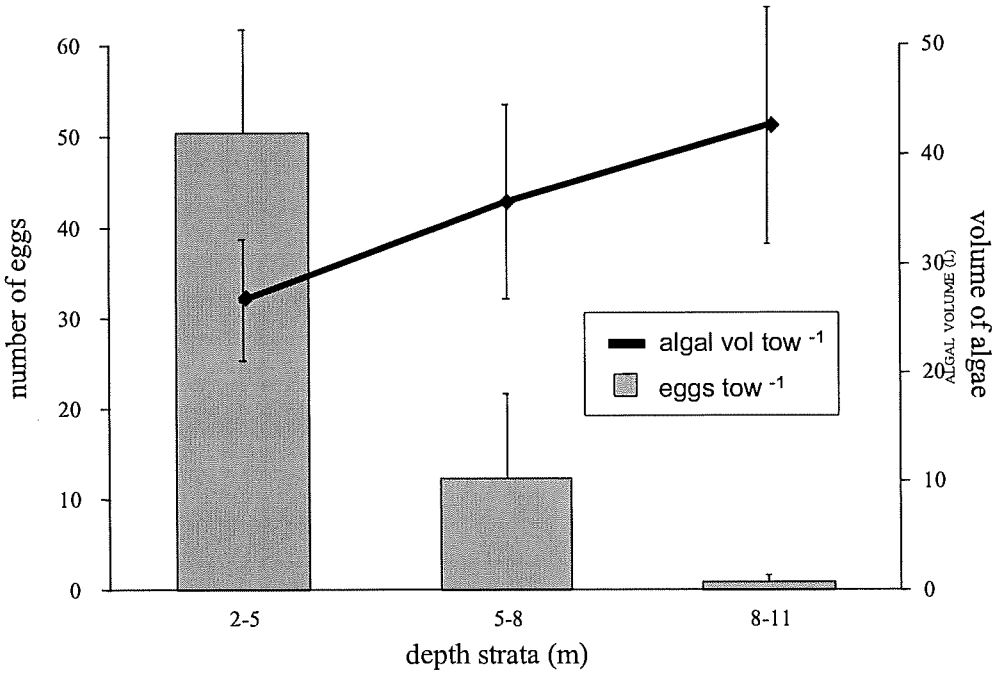



Fig. 5.10.5 Mean abundance (N. tow$^{-1}$) of *Hyporhamphus melanochir* eggs by depth strata in Great Oyster Bay. Bold line indicates mean algal volume (litres.tow$^{-1}$). Error bars are s.e.

Table 5.10.1 One-way ANOVA for abundance of *Hyporhamphus melanochir* eggs by depth strata.

| | Log egg abundance | | | |
|---|---|---|---|---|
| Factor | DF | MS | F | Prob. |
| Depth strata | 2 | 25.7 | 19.34 | <0.001 |
| Error | 21 | 1.3 | | |

Table 5.10.2 Ryans Q-test for *Hyporhamphus melanochir* egg abundance across depth strata from ANOVA in Table 5.10.1. ($\alpha$=0.05). Bold underlining indicates no significant difference.

| Depth strata | 2 - 5 m | 5 - 8 m | 8 - 11m |
|---|---|---|---|
| Mean egg abundance | 3.7 | 1.3 | 0.2 |

Wild caught *H. melanochir* eggs were assigned an age in days based on staging criteria developed from laboratory reared eggs (section 5.10.2.3). The earliest calculated spawning date was October 30 with the latest date being January 7 (Fig. 5.10.6). The absence of eggs spawned during mid December more likely reflects the extended period between sampling than a complete cessation of spawning. This is supported by the fact that both male and female GSI's were highest during this period. It was not possible to confirm the exact commencement of the spawning season from egg distributions, as spawning had begun prior to the first sampling period in early November 1996. While no eggs were taken in the last sample in February 1997, suggesting spawning had ceased, the presence of ripe fish in that month indicates that some spawning may continue into late summer. The abundance of eggs from individual spawning dates varied considerably during the sampling period, however, no distinct periodicity was apparent.

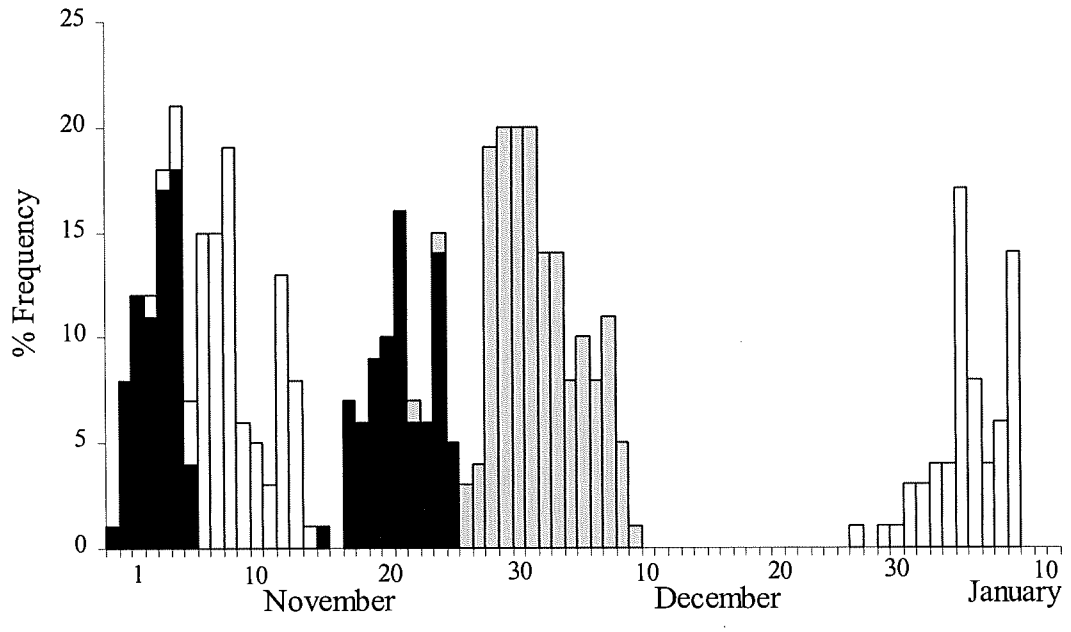



Fig. 5.10.6 Backcalculated spawning dates of *Hyporhamphus melanochir* eggs collected between November and January 1996. Horizontal bar indicates the period when no sampling was conducted. Different shadings represents eggs from different sampling periods.

Of particular note is the low abundance of late stage eggs, with the oldest egg estimated to be 19 days post-spawning, and most eggs being less than 14 days post-spawning. This suggests either a high rate of egg mortality or transport of eggs from the area. Possible causes of mortality are predation, or movement of eggs onto the beach with the drift algae. Following strong winds from the south or south-east, large quantities of drift algae are washed onto the beaches along the northern end of Great Oyster Bay. Examination of this algae revealed similar abundances of eggs as that found in algae from beam trawl tows. However, attempts to correlate egg mortality with periodicity in winds from the south or south-east were unsuccessful. Strong northerly winds may result in export of eggs from the spawning areas due to drift algae being carried off-shore although limited sampling in deeper water failed to locate eggs in depths >11 m.

High rates of predation by fish seems unlikely, as the eggs are clear, and difficult to detect visually, and become well entangled in the drift algae. It is possible that some eggs are consumed incidentally by herbivorous fish. Predation by small invertebrates living within the drift algae bed is more a likely cause of some egg mortality.

*5.10.3.3 Egg morphology and development*

During the stripping process, *H. melanochir* eggs were negatively buoyant, sinking immediately to the bottom of the jars or until they came in contact with the artificial substrate. In addition, there was no tendency for eggs to adhere to each other and they dispersed immediately after entering the jars. This may explain why eggs were found singly within the drift algae.

Egg densities for rearing were 50 to 80 eggs per 2 litre jar. Egg mortality during the rearing process was negligible for jars with artificial substrate, but were high where no substrate was provided. In jars with no substrate, eggs tended to aggregate, and no eggs survived through to hatch. Rate of development prior to mortality in these eggs was also considerably slower than those from treatments with artificial substrate. All staging and description was done from eggs from treatments with substrate.

*Hyporhamphus melanochir* eggs are round and 2.84-3.01 mm in diameter (mean=2.93 mm, n=20) (Fig. 5.10.7a-h). The chorion is smooth, unpigmented and covered with approximately 100 (mean=98, n=20) long (mean 8.0mm, n=100) hairs. Eggs were successfully reared through to hatching with the following descriptions representing eight development at a range of time intervals post-fertilization.

Stage 1 (0-12 hrs post-fertilization, Fig. 5.10.7a). The cytoplasm starts to accumulate at the vegetal pole and has divided to around the 8 cell stage. The yolk-mass remains unsegmented with oil droplets remaining clumped.

Stage 2 (12-36 hrs, Fig 5.10.7b). During this blastula stage the blastoderm forms a well defined cap with individual blastomeres still visible.

Stage 3 (36-84 hrs, Fig. 5.10.7c). The blastoderm covers around one third of the yolk surface area with the periblast clearly visible between the blastomeres and the yolk mass.

Stage 4 (84-132 hrs, Fig. 5.10.7d). During this stage the blastoderm increases in size before the margins develop into the germ ring. The embryo forms along the embryonic shield, the blastopore closes and notochord and optic vesicles become visible. The oil droplets move to a position in line with the anterior end of the embryo. A depression forms in the yolk mass around the head of the embryo.

Stage 5 (132-228 hrs, Fig. 5.10.7e). Circulation commences and tail becomes fully differentiated from yolk surface. In late stage 5 eggs pectoral fin buds form, myomeres become visible and the tail increases in length to around one third total length.

Stage 6 (228-324 hrs, Fig. 5.10.7f). Flexion commences in early stage 6 eggs with the caudal fin membrane clearly visible. By late stage 6 larvae had completed flexion with incipient rays evident on caudal and pectoral fins. The eyes are well formed and the perivitelline space becomes larger.

Stage 7 (324-516 hrs, Fig. 5.10.7g). Body depth of larvae becomes increases, with initial ossification of caudal rays commencing. Mouth well formed. Pectoral fins rays remain incipient. Volume of yolk mass decreases.

Stage 8 (516-720 hrs, Fig. 5.10.7h). Myomeres well formed and head and body depth increases. No pigmentation present.

*5.10.3.4 Larval development*

*Hyporhamphus melanochir* larvae hatched 28-30 days after fertilisation at 7.8 - 8.5 mm (mean = 8.2 mm, n = 20). The body of newly hatched yolk-sac larvae is very elongate (Fig. 5.10.8). The head is round and small and the eyes are large and round. The mouth is large, reaching to approximately the centre of the eye in newly hatched larvae. Spination is absent from all larval stages examined. Larvae have 61 myomeres. Pigmentation in yolk-sac larvae is heavy and concentrated along the dorsal surface of the trunk and tail. The caudal fin is well developed at hatching with incipient rays present in the pectoral fins..

Larvae are characterised by heavy pigmentation, a long gut, lack of head spines and large size at hatching. Newly hatched larvae are competent swimmers, have a minimal yolk reserve, with yolk absorption complete within 24 hours of hatching.

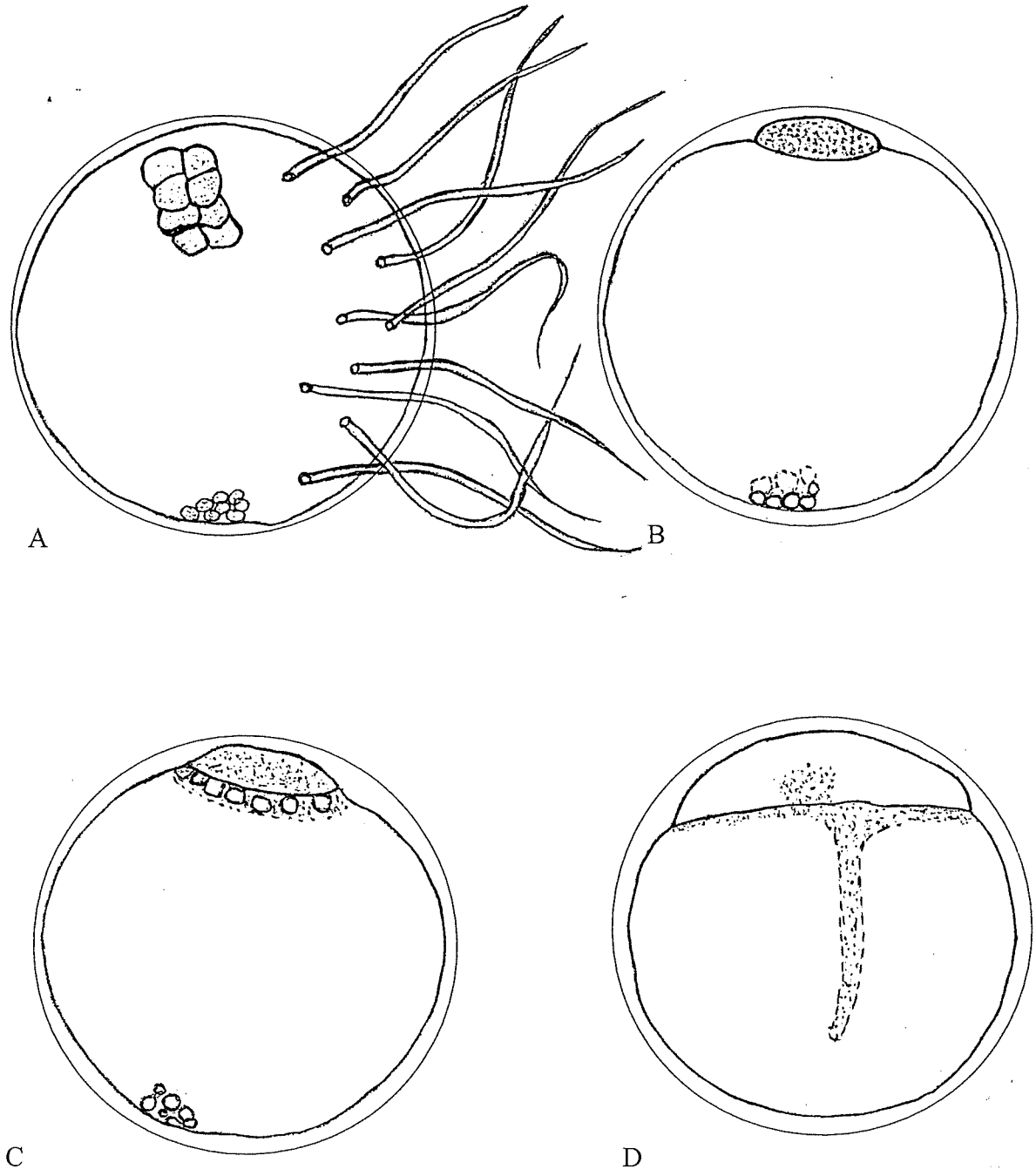



Fig. 5.10.7 a-d. Egg developmental stages of *Hyporhamphus melanochir*. Choronic filaments represented in Fig. A, but not included in other stages.

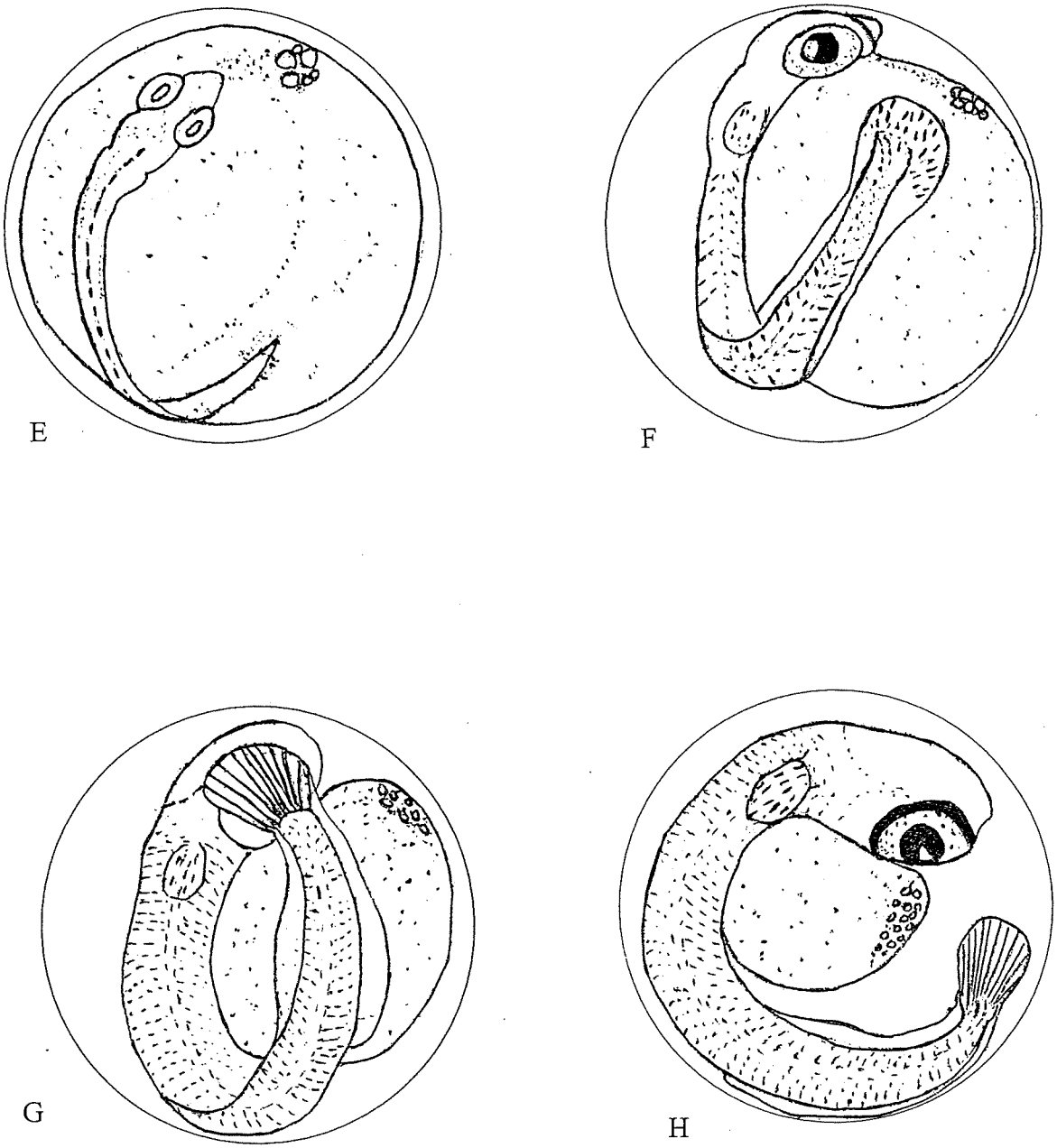



Fig. 5.10.7 e-h. Egg developmental stages of *Hyporhamphus melanochir*. Choronic filaments not represented.

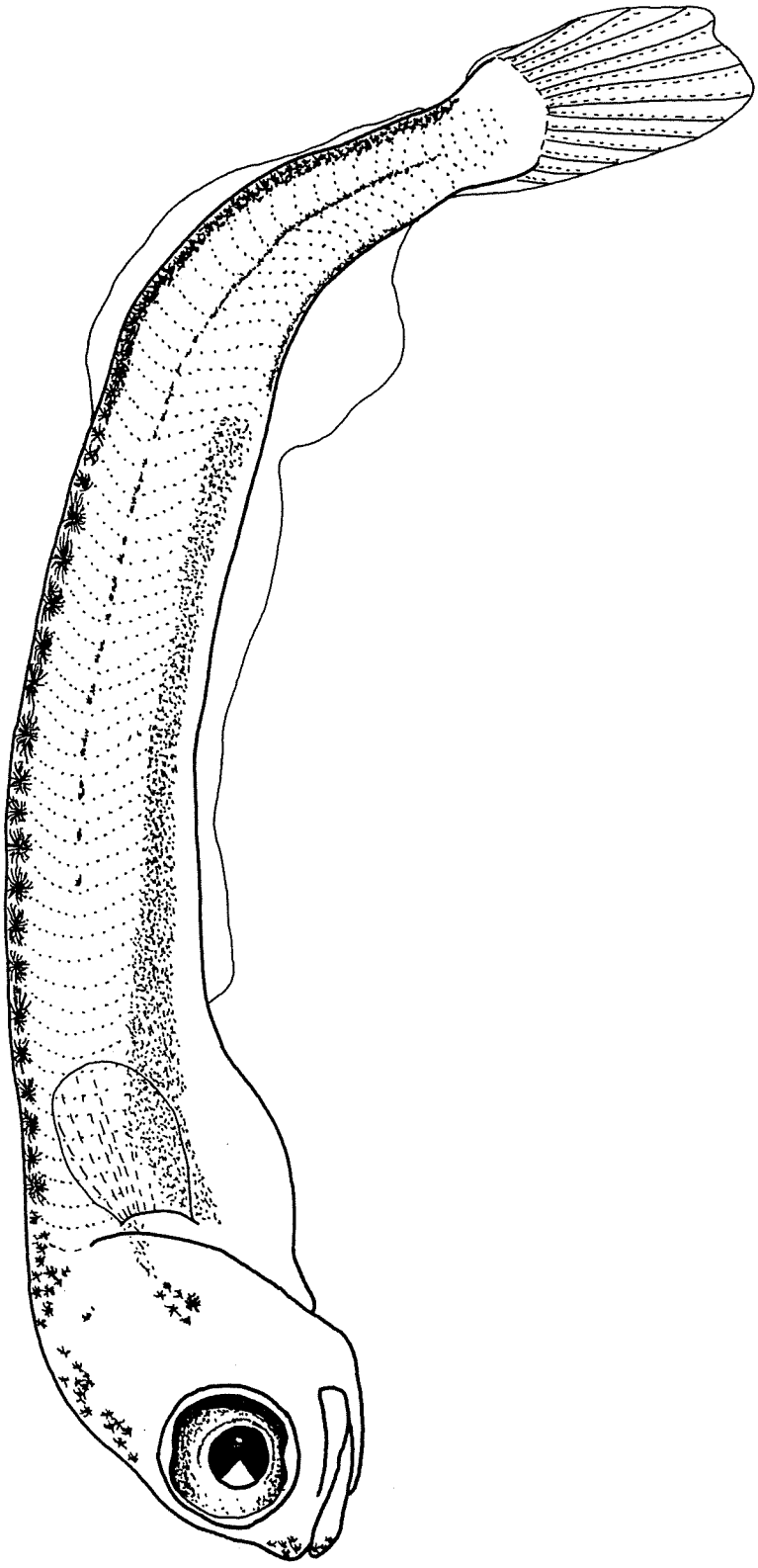


Fig. 5.10.8 Recently hatched *Hyporhamphus melanochir* larvae. Length 8.9 mm.

### 5.10.4 Size compositions

*H. melanochir* sampled from the commercial fishery on the east coast region of Tasmania ranged from 19.0 to 42.0 cm with the distribution consisting of a single mode skewed to the left, with a mean of 30.7 cm (Fig. 5.10.9). The size range of fish was narrow with over 90% of fish between 25 and 35 cm and few fish >35 cm.

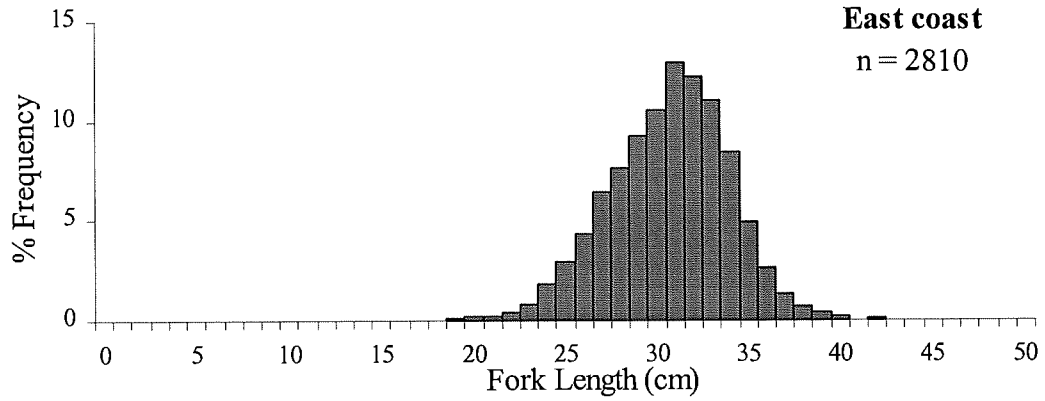



Fig. 5.10.9 Size-frequency distribution of *Hyporhamphus melanochir* from the commercial fishery on the east coast of Tasmania. n is sample size.

*H. melanochir* sampled from the commercial fishery on the north coast region of Tasmania ranged from 17.0 to 45.7 cm, with the distribution consisting of a single mode with a mean of 32.3 cm (Fig. 5.10.10). The size range of fish was, however, considerably broader than that from the east coast, with only 71% of fish between 25 and 35 cm, and 26% of fish >35 cm.

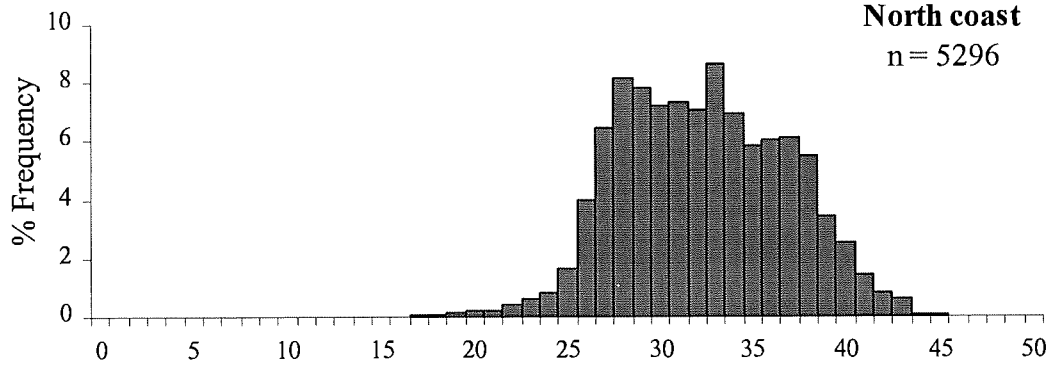



Fig. 5.10.10 Size-frequency distribution of *Hyporhamphus melanochir* from the commercial fishery on the north coast of Tasmania. n is sample size.

Monthly length-frequency distributions of *H. melanochir* from the east coast reveal considerable variability in the size composition of commercial catches throughout the year, with mean length ranging from 28.2 cm in May to 32.6 cm in December (Fig. 5.10.11). The overall size range, however, was generally more consistent. The presence of a bimodal distribution in summer and autumn indicates the movement of a cohort at around

25 cm into the catches in January, which merges with the main cohort by December. The age class of these fish is examined later in the age and growth section.

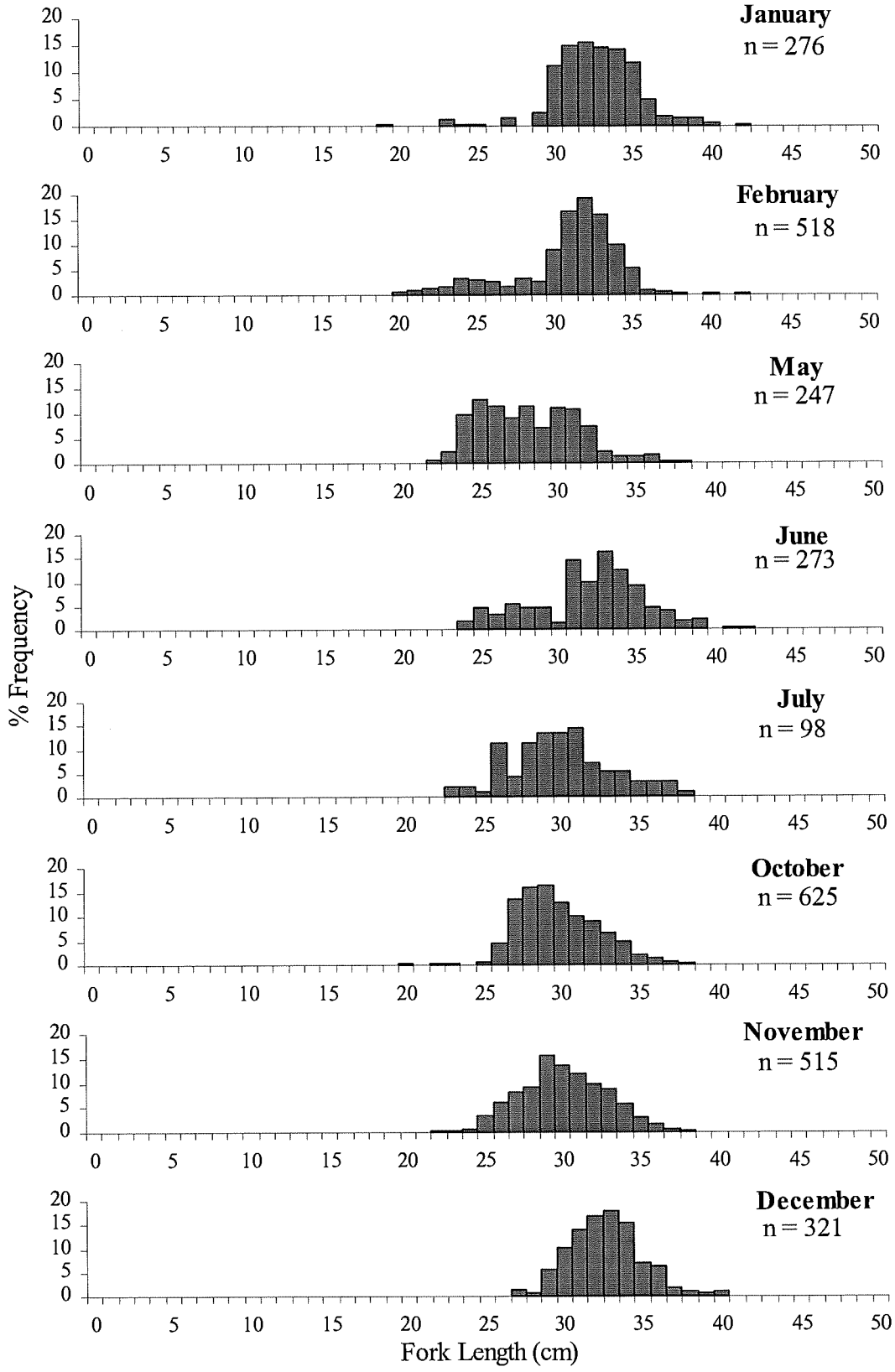



Fig. 5.10.11 Monthly size-frequency distribution of *Hyporhamphus melanochir* from the commercial fishery on the east coast of Tasmania. n is sample size.

The size range of fish on the north coast was large and generally consistent in most months, although few fish >35 cm were present in June (Fig. 5.10.12). There is evidence of a discrete cohort of small fish (around 20-25 cm) entering the catches in May before merging with the main cohort in latter months.

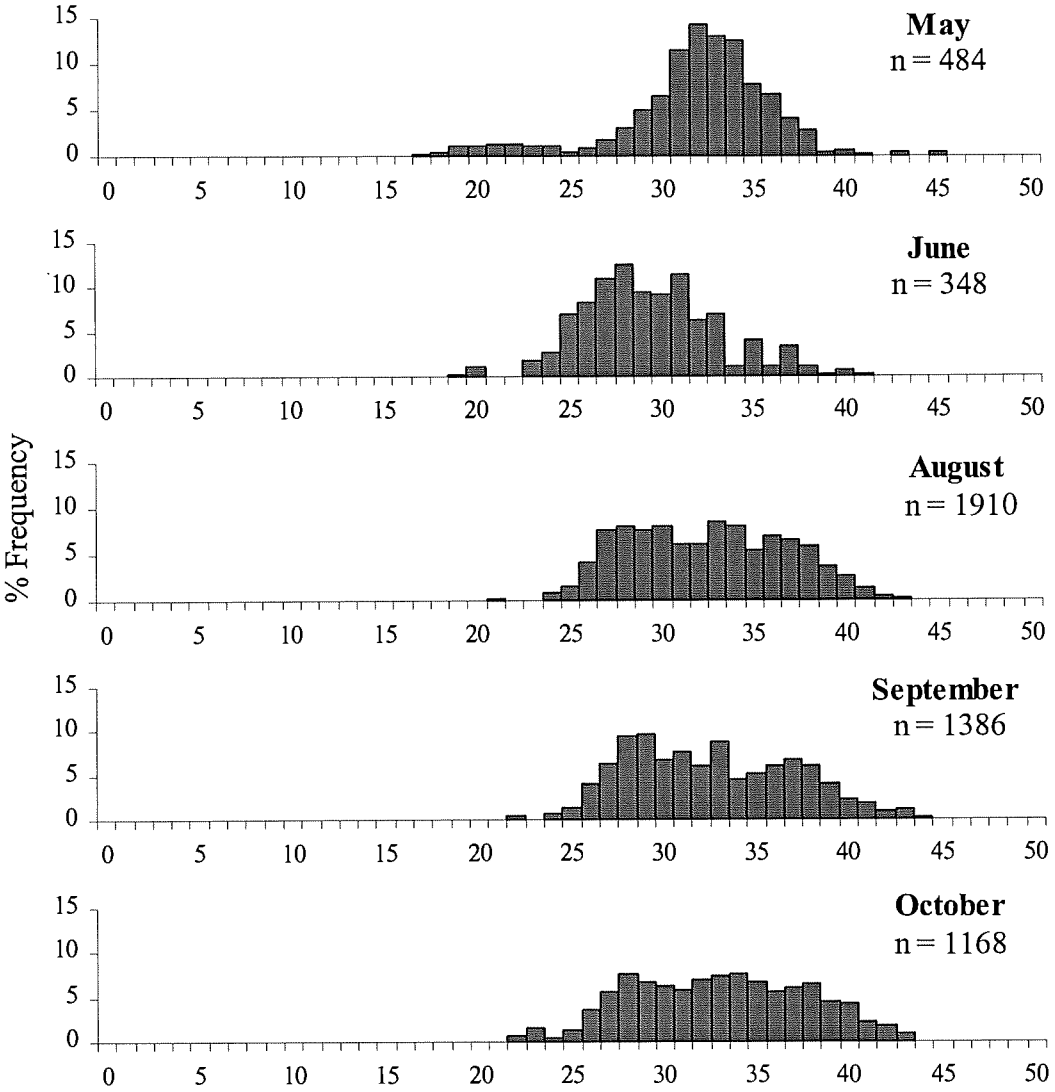



Fig. 5.10.12 Monthly size-frequency distribution of *Hyporhamphus melanochir* from the commercial fishery on the north coast of Tasmania. n is sample size.

Research sampling of *H. melanochir* was conducted on the east coast of Tasmania with surface gill-nets, dip nets and beach seine. Fish ranged from 4.2 to 37.5 cm, with the distribution consisting of three distinct modes at around 7, 15-17 and 29 cm (Fig. 5.10.13).

Temporal trends in size-compositions of *H. melanochir* from research sampling show modal progression of distinct size classes (Fig. 5.10.14). Fish first appeared at 4-9 cm in July, progressing to around 13 cm by the following January, reaching a mean length of around 18 cm by the following November. The age-class of these fish is examined in the section on age and growth. The absence of mature fish in most months reflects the lack of gillnet sampling in those months rather than the absence of fish from the sampling areas.

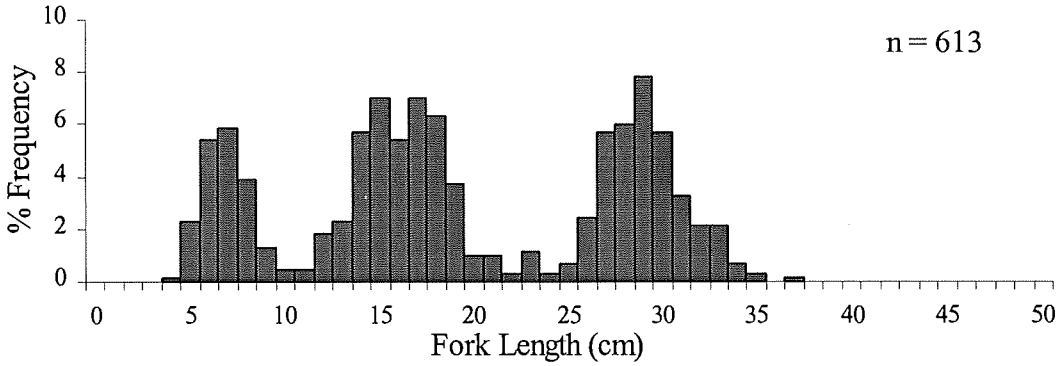



Fig. 5.10.13 Size-frequency distribution of *Hyporhamphus melanochir* from research sampling on the east coast of Tasmania. n is sample size.

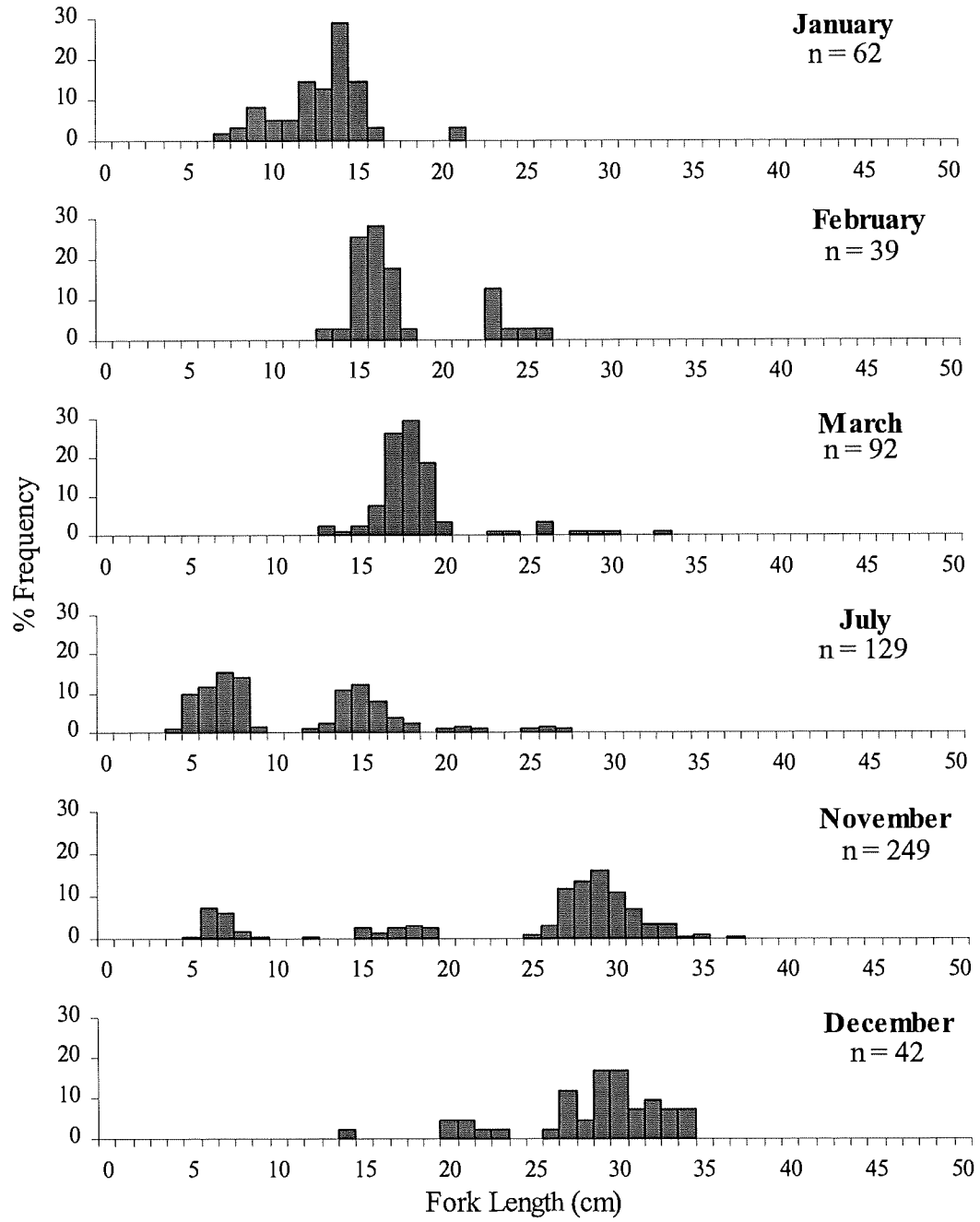



Fig. 5.10.14 Monthly size-frequency distribution of *Hyporhamphus melanochir* from research sampling on the east coast of Tasmania. n is sample size.

## 5.10.5 Age, growth and age composition

### *5.10.5.1 Otolith structure and interpretation*

Sagittal sections of *H. melanochir* showed distinctive alternating opaque and translucent zones seen under transmitted light (Fig. 5.10.15). A total of 10.8% of otoliths were rejected due to the poor quality of sections. The increment banding pattern remained relatively easy to read in old fish despite the narrowing of translucent zones. The primordial area of all otoliths consisted of an opaque region with no obvious increment structure. Immediately adjacent to this was a broad opaque zone with a mean radius (±s.d.) of 442 ± 82 μm that occurred in 82.2% of all sagittae examined (Fig. 5.10.15). This zone was characterised by being fainter than adjacent opaque zones. The structure of the otolith differed outside this zone with all sagittae of sufficient radius having a consistent narrow opaque zone with a mean radius of 1200 ±87 μm. Beyond this, there were distinctive opaque zones that decreased in width towards the margin.

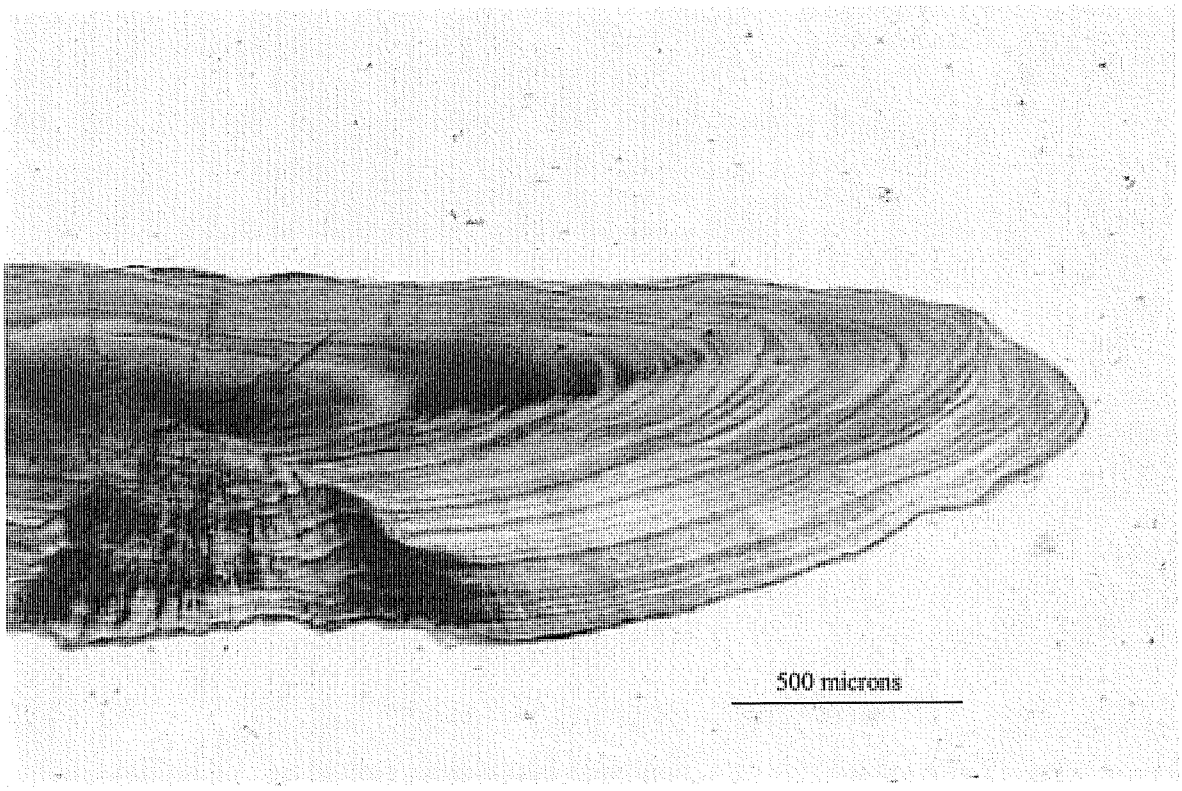



Fig. 5.10.15 Transverse sagittal otolith scans of *Hyporhamphus melanochir*

Given the variability in the structure of the opaque zones in the region of the primordium, the definition of the first annual increment was based on the relationship between the otolith radius and mean length of the 0+ cohort. Monthly progressions of juvenile size-compositions show *H. melanochir* first appeared in samples in July at around 4-9 cm from spawning that peaked the previous spring and summer (Fig. 5.10.14). This cohort had progressed to around 13.2 cm by the following January, and given the birth date of 1 December which corresponds to the mid-point of the spawning season, is now defined as the 1+ age-class. This cohort had reached a mean length of around 18 cm by November. The otolith radius of the 0+ cohort increased from a mean of 416 μm in July to 488 μm in November, a radius slightly larger than the first opaque zone (442 μm) suggesting the first opaque zone was laid down in spring.

By January, fish now defined as the 1+ cohort had a mean radius of 570 μm. The otolith radius increased in this cohort to 1107 μm by the following November. This is consistent with the radius of the first distinct narrow opaque zone seen in all otoliths, and hence defined as the second annual increment.

*5.10.5.2 Growth*

Von Bertalanffy growth curves were fitted to combined male and female individual length-at-age data as sample sizes were limited for both sexes (Fig. 5.10.16). The respective asymptotic lengths ($L_\infty$) for males and females combined was 34.3 cm (Table 5.10.3). Mean lengths-at-age for males and females combined are presented in Table 5.10.4. There was a broad range of lengths within individual age-classes with a maximum of 6 age-classes present in a 1 cm size-class. Growth is rapid until around 3 years old and 25 cm and then slows appreciably. Maximum ages for males and females from research and commercial sampling on the east coast of Tasmania were 7 and 8 years old respectively.

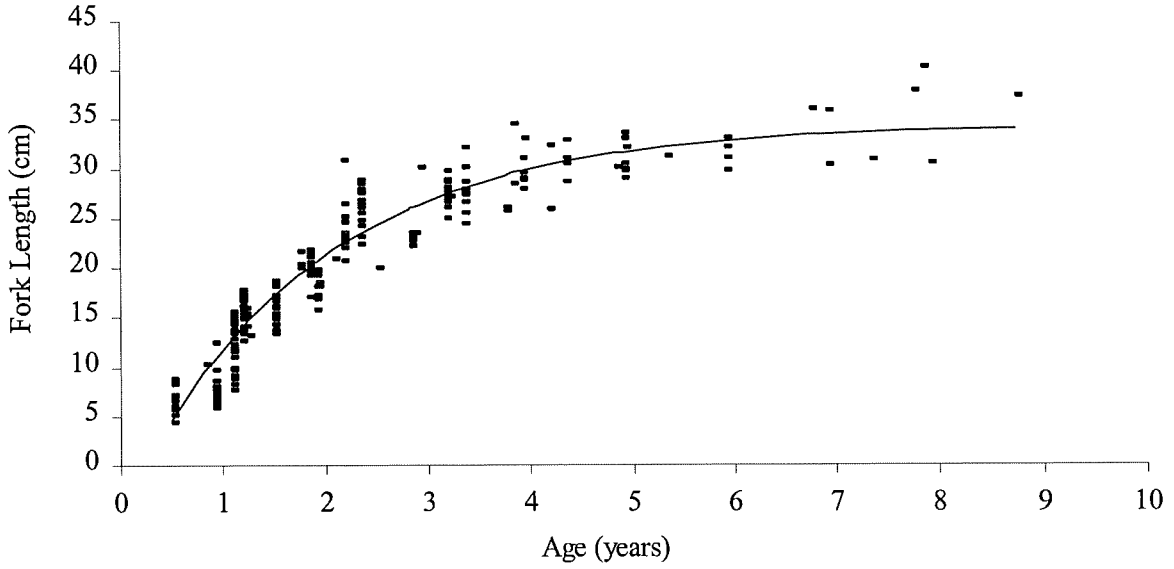



Fig. 5.10.16 Von Bertalanffy growth curves for male, female and juvenile *Hyporhamphus melanochir* combined from eastern Tasmania.

Table 5.10.3 Von Bertalanffy growth parameters derived from length at age data for *Hyporhamphus melanochir* from eastern Tasmania.

| | | Von Bertalanffy growth parameters | | | | | |
|---|---|---|---|---|---|---|---|
| | n | $L_\infty$ | s.e. | K | s.e. | to | s.e. |
| Males/Females /Juveniles | 227 | 34.3 | 0.69 | 0.54 | 0.03 | 0.23 | 0.04 |

Table 5.10.4 Mean lengths at age for age-classes of male and female *Hyporhamphus melanochir* from eastern Tasmania. n is sample size.

| Age | Females - Males - Juveniles | | |
|---|---|---|---|
| | n | Mean | s.d. |
| 0 | 27 | 74.6 | 1.64 |
| 1 | 94 | 157.4 | 3.26 |
| 2 | 39 | 248.5 | 2.65 |
| 3 | 36 | 281.3 | 2.17 |
| 4 | 14 | 306.6 | 2.07 |
| 5 | 9 | 317.0 | 1.13 |
| 6 | 3 | 340.4 | 3.26 |
| 7 | 5 | 340.8 | 4.61 |
| 8 | 1 | 372.0 | - |

*5.10.5.3 Age composition*

The age composition of *H. melanochir* from the commercial fishery on the east coast of Tasmania between 1995 and 1997 was estimated separately, with the number of fish aged proportional to the number in each 2 cm size-class from the size composition of the total population sample (Fig. 5.10.17). A maximum of 8 age-classes occurred in the catches, dominated by 4 and 5 year old fish, which together made up 43% of the sampled population.

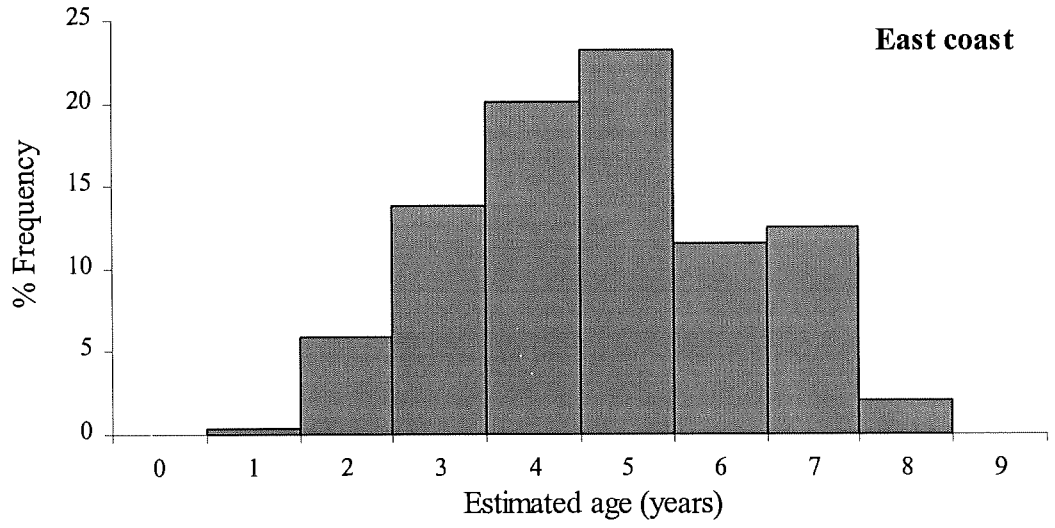



Fig. 5.10.17 Estimated age composition of *Hyporhamphus melanochir* from the commercial fishery in east coast region of Tasmania.

## 5.10.6 Discussion

### *5.10.6.1 Spawning and early life-history*

The presence of ripe, running ripe and spent *H. melanochir* (≥ stage 5) from October to February clearly demonstrates that spawning occurs over an extended period of at least five months in eastern Tasmania. However, the increased proportion of fish with resting stage gonads by February suggests that the bulk of spawning occurs between October and December, with a lower level of spawning activity in the latter half of the spawning period. The pattern of GSI's also suggests that spawning peaks between October and December. While the earliest back-calculated spawning date for *H. melanochir* was in late October, the presence of running ripe fish in early October indicates some spawning occurred at that time. This is consistent with the October peak in female *H. melanochir* GSI's previously documented for eastern Tasmania (St Hill 1996).

The timing of spawning in *H. melanochir* may be linked to the timing of the spring bloom in productivity in these waters which begins around late September (Harris *et al.* 1987). However, given that the duration of the spring bloom in the shelf waters of southern and eastern Tasmania can vary by as much as three months from year to year (Harris *et al.* 1991), the extended spawning period may also be a strategy to maximise the number of larvae encountering suitable feeding conditions. The extended spawning period is also related to the fact that *H. melanochir* are serial spawners, with asynchronous oocyte development occurring simultaneously in reproductively active ovaries (St Hill 1996).

The fact that most eggs examined were caught in depths of two to five metres, and the lack of age difference between eggs from different strata that would be expected with a higher rate of egg predation in deeper water, suggests that spawning is concentrated in shallow water. This is also consistent with the distribution of fish in spawning condition which were also concentrated close to shore. After fertilisation, *H. melanochir* eggs became negatively buoyant suggesting that they sink immediately to the bottom and become attached to the drift algae by their choronic filaments. There was no evidence that *H. melanochir* eggs are attached in clusters on seagrass blades as there was no tendency for eggs to adhere to each other in the rearing jars, and were found singly within the drift algae. In addition, as few seagrass beds are present in shallow water in Great Oyster Bay, such habitat appears to be of little importance as a spawning habitat in this area. The lack of eggs in *Heterozostera* beds in Norfolk Bay also indicates that this habitat may be of little importance in areas of substantial seagrass habitat, although the extent of spawning in this bay is still unclear. However, seagrass habitats may be of greater significance around areas such as Flinders Island (see Chapter 5.7), where the majority of shallow water habitat consists of seagrass beds and spawning is known to occur (A. Jordan unpubl. data). Further sampling in these areas will need to be conducted before the full extent of the spawning habitat requirements can be assessed.

*5.10.6.2 Age, growth and age composition*

The monthly progression of the 0+ cohort indicates rapid growth during summer when water temperatures are at a maximum. Growth then slows appreciably during winter and spring to reach approximately 7 cm after one year. The absence of a distinct opaque zone in the otoliths of this cohort is consistent with the conclusion that this represents the 0+ age-class. A single opaque zone became discernible in otoliths of this cohort by December at an age of one year. This cohort progressed to a mean length of around 17 cm by the following November when otoliths possessed two opaque zones, therefore representing 2+ fish.

Growth of male and female *H. melanochir* is relatively rapid for approximately the first 3 years, slowing appreciably at around 25 cm. After 3-4 years there was an increasing variation in size-at-age with fish at a length of 30 cm ranging from 2 to 8 years old.

The maximum age of 8 years for female *H. melanochir* found in the present study is consistent with that reported for South Australian waters (Jones 1990). However, males reached 8 in eastern Tasmania compared to 10 years in South Australia. The lower maximum ages for males may reflect either spatial variations in the age structure or the small sample size of males. In addition, the sample aged from eastern Tasmania contained considerably fewer fish >35 cm than that represented in the size composition of fish from the north coast, or South Australian waters (Jones 1990). While annuli in *H. melanochir* from South Australian waters have been validated through the use of marginal increments (Ling 1958), there is a need to conduct such analysis from fish in Tasmania in order to confirm growth and age estimates.

A maximum of 9 age-classes of *H. melanochir* were represented in samples from commercial dipnet fishery in eastern Tasmania, dominated by 4 and 5 year old fish which made up around 43% of the population. Evidence suggests that larger fish are underrepresented in dipnet landings, with larger fish possibly remaining in deeper water outside the depth range of the gear. Further sampling will be required by beach seine on the east coast before the size composition of the entire population can be assessed. Despite this, the present data shows no evidence of variable recruitment in the population of *H. melanochir* with no particular year-class dominant.

The lack of aged samples from the north coast of Tasmania precludes an assessment of the extent of spatial variations in growth and age composition. Such analysis will need to be conducted before population models can be advanced for this species across the entire distribution of the commercial fishery.

## 5.11 Minor commercial species

### 5.11.1 Introduction

A range of commercial species, including Eastern Australian salmon, flounder, mullet and jackass morwong inhabit coastal soft-sediment habitats throughout Tasmania, and are a significant component of both commercial and recreational fisheries (Lyle 1998, Lyle and Smith 1998). Combined commercial landings of these species throughout Tasmania were around 825 tonnes in 1996/97 (Lyle 1998). Details of the biology, population parameters and commercial catch history for these species is summarised in Lyle (1994).

While detailed biological studies have been conducted on some species, such as jackass morwong (see Lyle 1995), aspects of recruitment and habitat associations are poorly studied in others. The aim of this chapter is to summarise the size composition information collected in the sampling programs detailed in the previous chapters in order to examine the significance of inshore soft-sediment habitats around Tasmania for life-history stages of the above species.

### 5.11.2 Yellow-eye mullet (*Aldrichetta forsteri*)

#### *5.11.2.1 Size compositions*

Research sampling of *Aldrichetta forsteri* was conducted on the east coast of Tasmania with beam trawl, gillnets and beach seine. Fish ranged from 3.0 to 38.5 cm with the distribution consisting of two distinct modes at 9-10 and 30-31 cm (Fig. 5.11.1). This distribution reflects small fish caught by beach seine and larger fish caught by gillnet.

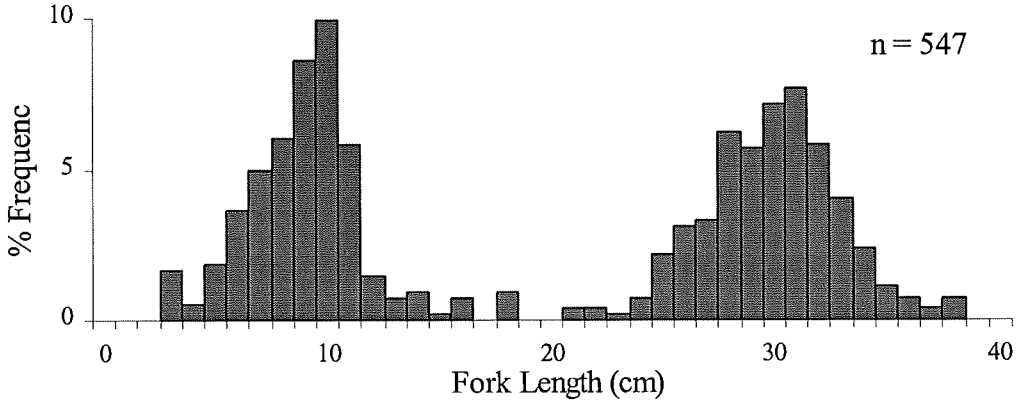



Fig. 5.11.1 Length-frequency distribution of *Aldrichetta forsteri* from southern, eastern and northern Tasmania. n is sample size.

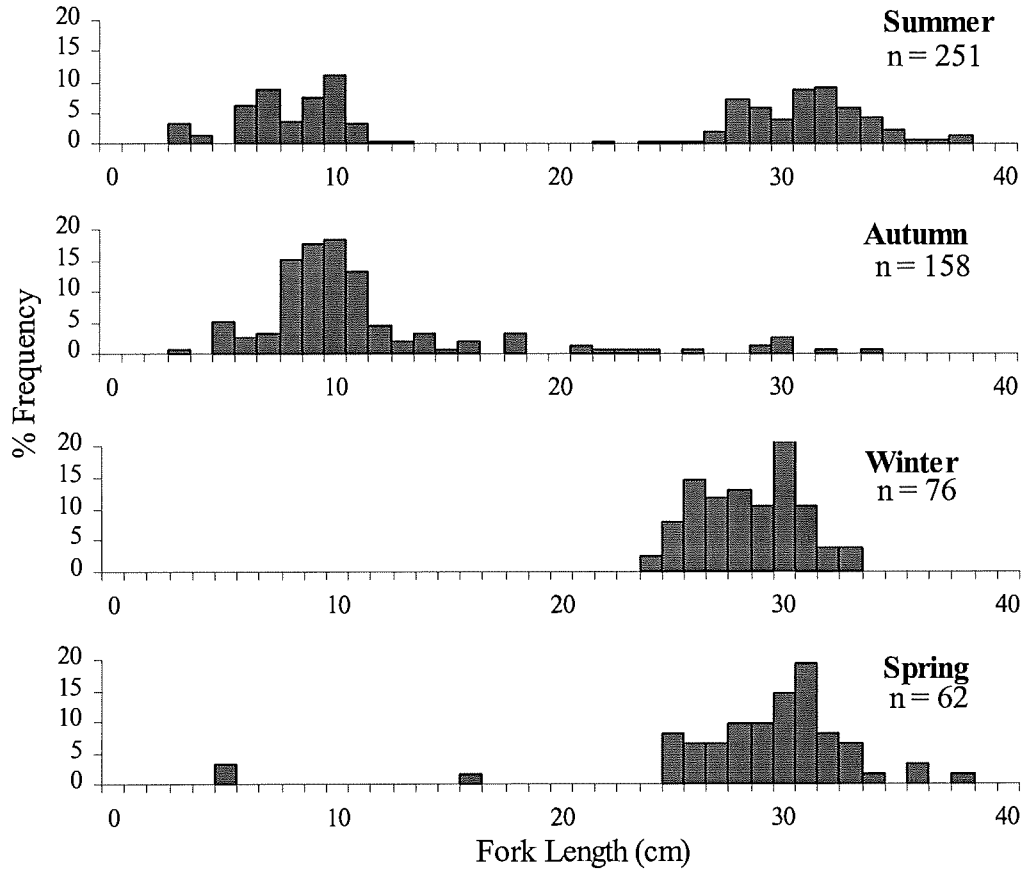



Fig. 5.11.2 Seasonal length-frequency distributions of *Aldrichetta forsteri* from southern, eastern and northern Tasmania. n is sample size.

The seasonal length-frequencies show the recruitment of small (~3 cm) juveniles in summer and autumn, although there was a broad range of sizes in the smaller mode in those months (Fig. 5.11.2). There was little seasonal change in the size composition of larger fish, although few fish in this mode was caught in autumn. The absence of small fish in winter reflects the lack of beach seining in that season.

### *5.11.2.2 Reproduction*

The seasonal distribution of gonad stages shows the presence of resting phase and early developing fish (stage 2 and 3) during winter and spring and ripe, running ripe and spent fish ($\geq$ stage 5) during summer. While spawning occurred during summer, the full duration of the spawning season cannot be determined due to the absence of mature fish in the autumn samples (Fig. 5.11.3).

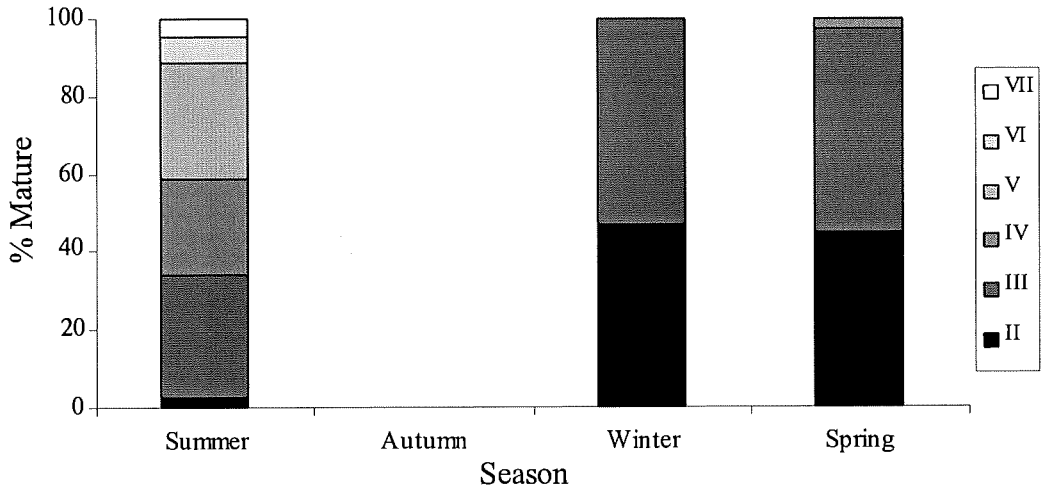



Fig. 5.11.3 Seasonal gonad stage percentages for female *Aldrichetta forsteri* from southern, eastern and northern Tasmania.

## 5.11.3 Eastern Australian salmon (*Arripis trutta)*

### *5.11.3.1 Size compositions*

Research sampling of *Arripis trutta* was conducted on the east coast of Tasmania with gillnets and beach seine. Fish ranged from 3.8 to 45.0 cm with the distribution consisting of several modes, at around 7, 13, 26 and 38 cm (Fig. 5.11.4). This distribution reflects small fish caught by beach seine and larger fish caught by gillnet.

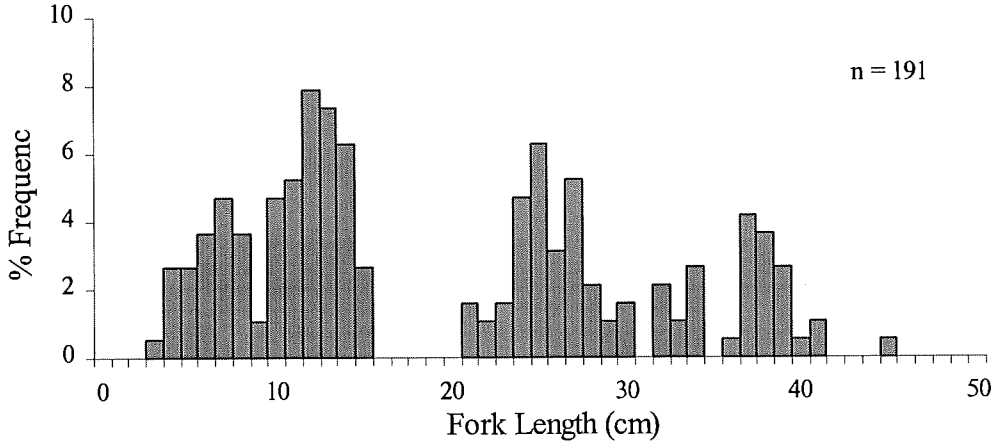



Fig. 5.11.4 Length-frequency distributions of *Arripis trutta* from research sampling in southern and eastern Tasmania. n is sample size.

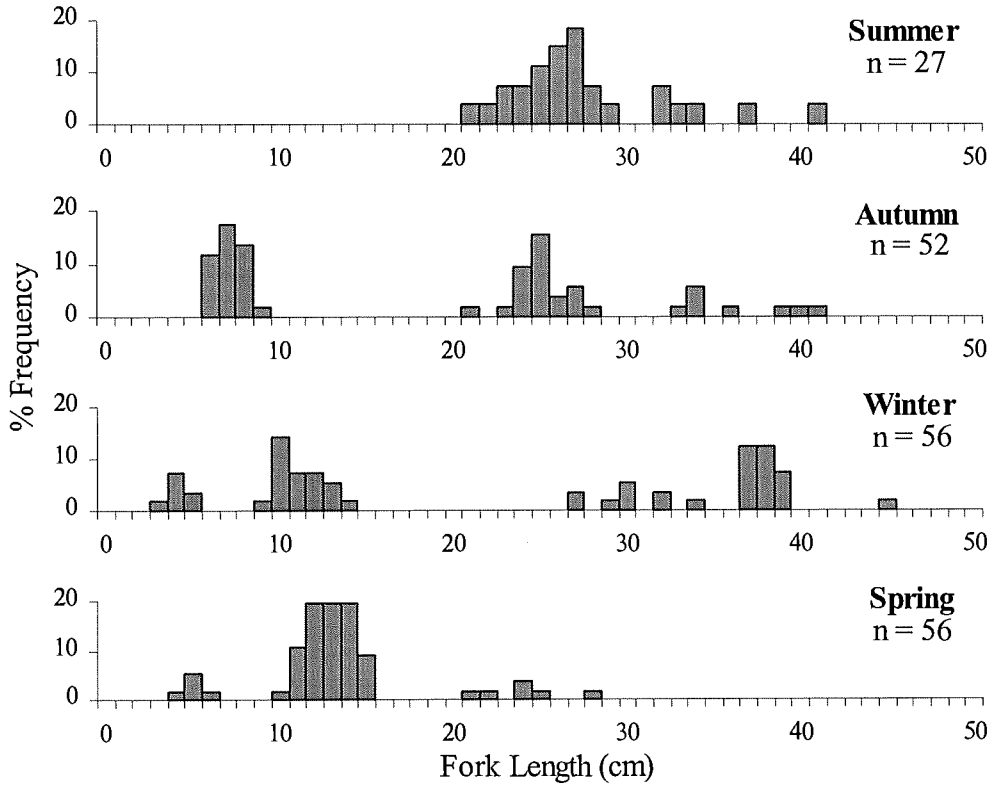



Fig. 5.11.5 Seasonal length-frequency distributions of *Arripis trutta* from research sampling in southern and eastern Tasmania. n is sample size.

The seasonal length-frequencies show the recruitment of small (4-6 cm) 0+ juveniles in winter, and the progression of 1+ fish from a mean length of 7.6 cm in autumn to 13.0 cm in spring (Fig. 5.11.5). There was little seasonal change in the size composition of larger fish, although few fish in this mode was caught in autumn.

### 5.11.4 Greenback flounder (*Rhombosolea tapirina)*

#### *5.11.4.1 Size compositions*

Research sampling of *Rhombosolea tapirina* sampled from the east coast of Tasmania ranged from 2.0 to 32.5 cm, with the distribution dominated by a mode at around 6 cm and a smaller mode at 20 cm (Fig. 5.11.6).

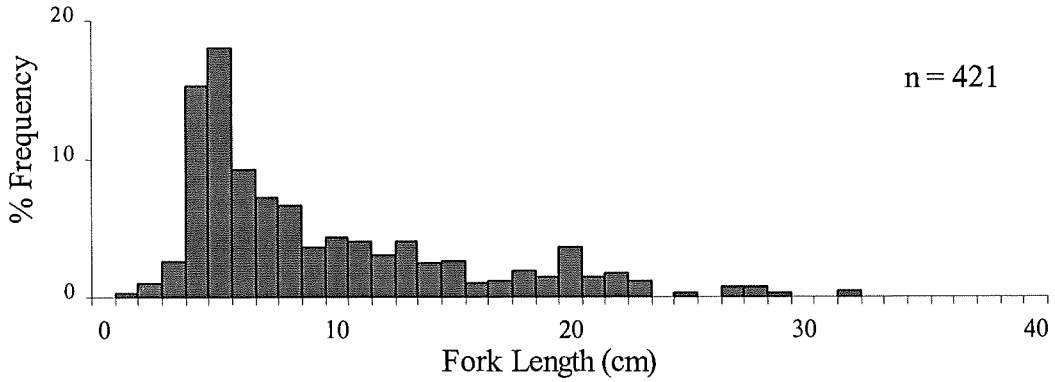



Fig. 5.11.6 Length-frequency distributions of *Rhombosolea tapirina* from research sampling in southern and eastern Tasmania. n is sample size.

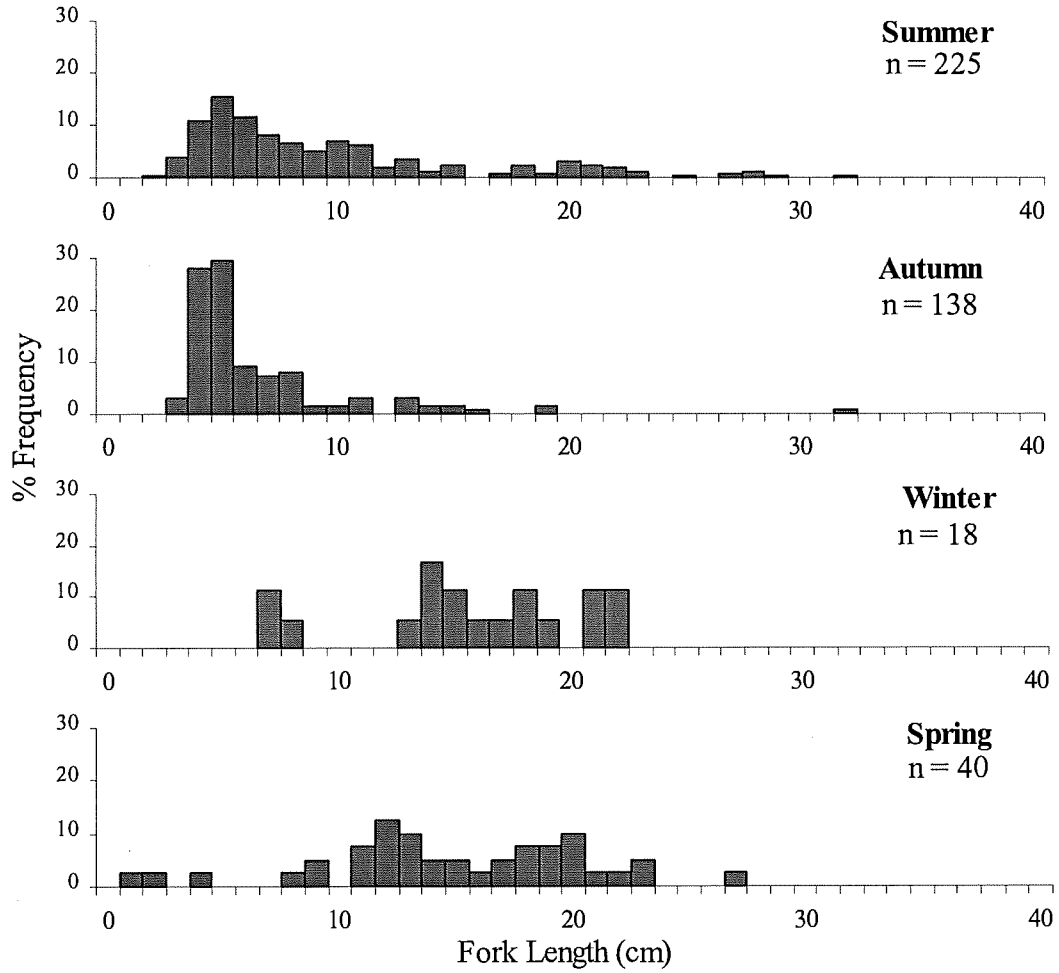



Fig. 5.11.7 Seasonal length-frequency distributions of *Rhombosolea tapirina* from research sampling in southern and eastern Tasmania. n is sample size.

The seasonal length-frequencies show the recruitment of small (2-3 cm) juveniles in spring and summer (Fig. 5.11.7). The broad range of lengths of 0+ fish reflects the extended period of spawning and recruitment. While there is no clear distinction of size classes in the population, particularly in summer when a broad range of sizes were caught, three distinct modes at 3, 12 and 20 cm are present in spring.

## 5.11.5 Jackass morwong (*Nemadactylus macropterus*)

### *5.11.5.1 Size compositions*

Very few *N. macropterus* were caught in inshore waters, despite extensive sampling of vegetated and unvegetated habitats with several gear types in three areas along the south and east coast (see sampling regime for *Platycephalus bassensis* in Chapter 5.9). The length-frequency distributions of *N. macropterus* from inshore regions of south-eastern Tasmania ranged from 7.4 to 17.5 cm, with the distribution dominated by two modes at around 7-8 cm and 15-16 cm, representing the 0+ and 1+ age-classes, respectively (Fig. 5.11.8). Both-age classes were caught in unvegetated soft-mud habitats in depths of 3 to 12 m.

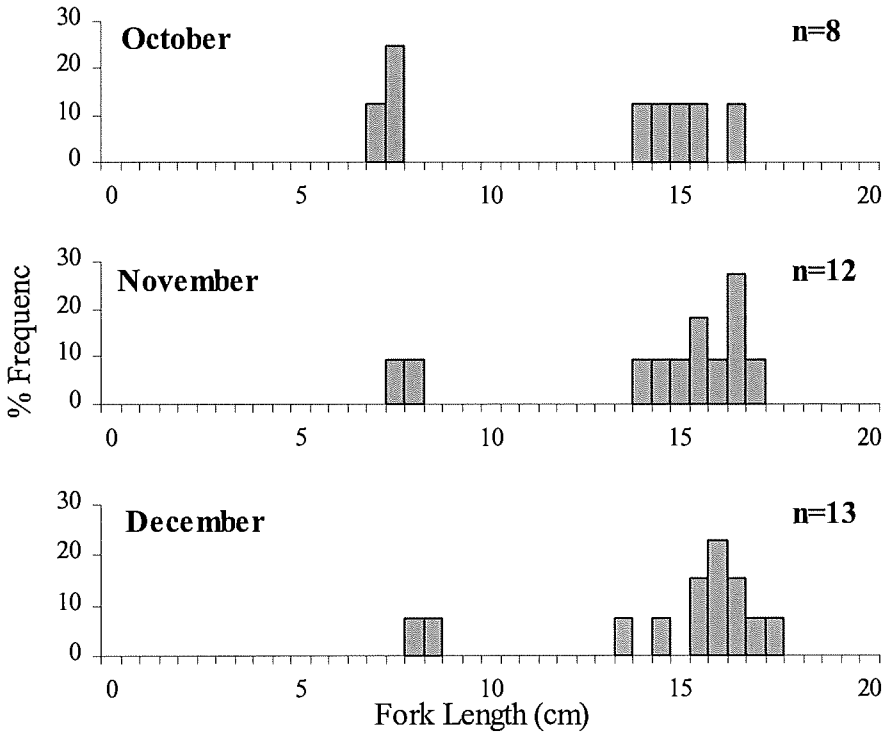



Fig. 5.11.8 Monthly length-frequency distribution of *Nemadactylus macropterus* collected in beam trawl sampling in south-east Tasmania between October and December 1995. n is sample size.

### 5.11.6 Discussion

The size composition of *Aldrichetta forsteri, Arripis trutta* and *Rhombosolea tapirina* in the present study reflects the selectivity of sampling gears, with juveniles caught by beach seine and adult fish by gillnet. In respect to *A. forsteri*, while small fish were dominated by a single mode, the large size range of juveniles suggests an overlap between age classes resulting from an extended spawning period. Given that spawning occurs at least during summer in Tasmanian waters, the 6-13 cm fish present in that season are most likely 1+ fish. This is consistent with *A. forsteri* in Port Phillip Bay where a distinct mode of 5-10 cm fish were present during summer (Jenkins *et al.* 1997). The presence of spawning fish in summer is consistent with that previously found for *A. forsteri* throughout south-eastern Australia (Thomson 1957).

High abundances of juvenile *A. forsteri* were caught in beach seine surveys of shallow beach habitats (see Chapter 5.5), with fish occurring at around 82% of all sites. In contrast, few juveniles were caught in beam trawl sampling of deeper subtidal habitats, possibly a result of gear avoidance, position in the water column or concentration in shallow water. Juvenile *A. forsteri* showed a preference for shallow intertidal habitats in Port Phillip Bay (Jenkins *et al.* 1996), which suggests that the lack of fish in beam trawl sampling may not have resulted entirely from net avoidance. Adult *A. forsteri* were common in both *Heterozostera* and unvegetated habitats in Georges Bay and Prosser Bay (see Chapter 5.3),

a pattern which is consistent with that found for the species in Western Port, Victoria (Edgar and Shaw 1995a).

Juvenile *Arripis trutta* first appear in Tasmanian waters during winter with a clear progression of this cohort during most seasons. High abundances of juvenile *A. trutta* were caught in beach seine surveys of shallow beach habitats, with fish occurring at 70% of all sites. The lack of juvenile *A. trutta* in beam trawl sampling of deeper subtidal habitats suggests that juveniles prefer shallow water as a nursery habitat. The fact that juveniles were characteristic of high energy beach sites supports this conclusion. Juvenile *A. trutta* showed a similar preference for shallow intertidal habitats in Port Phillip Bay (Jenkins *et al.* 1996), and therefore the lack of fish in beam trawl sampling may not be solely due to net avoidance. Adult *A. trutta* were caught over both *Heterozostera* and unvegetated habitats and were most common in Georges Bay and Prosser Bay.

The size composition of *Rhombosolea tapirina* in the present study shows small fish were dominated by a single mode, with the large size range of juveniles indicating an overlap between age classes resulting from an extended spawning period. *R. tapirina* is a serial spawner with spawning occurring at least from May to December in south-east Tasmania with a peak from late winter to early summer (Crawford 1984). Spawning appears to be concentrated in deeper coastal waters with larvae recruiting to shallow (~1 m deep) sheltered sandflats at around 1 cm over an period of around 6 months from early winter to mid summer (Crawford 1984). This is consistent with *R. tapirina* in Port Phillip Bay where recruitment was highest in shallow unvegetated habitats between August and January (Jenkins *et al.* 1997). The 3-10 cm fish present in summer represents 0+ fish, with some indication of a seasonal movement of this size class to around 11-12 cm in spring. This is consistent with the estimated mean length of 10.5 cm for 1+ fish (Kurth 1957). The second mode of around 20 cm in spring, therefore, most likely represents 2+ fish.

The significance of shallow beach habitats as a nursery area for *R. tapirina* is supported by the high abundance of this species in the beach seine survey detailed in Chapter 5.5. Juvenile *R. tapirina* were caught at 96% of all sites sampled throughout south-eastern Tasmania, and was a dominant part of the shallow beach community in all areas. While highest abundances were caught at unvegetated sites, juveniles were also present at all sites with beds of *Heterozostera* present. However, their preference for unvegetated habitats is likely to be related to their use of camouflage for protection for predators.

While only few juvenile *N. macropterus* were caught in the present study, the vulnerability of *N. macropterus* to beam trawls appears to be low during the day, and in fact, all catches were made in targeted sampling at night conducted over the settlement period. Fish in inshore waters have been observed schooling midwater during the day over subtidal reefs adjacent to unvegetated habitats (Last 1983). However, as gill-net sampling with 64 mm mesh in all routine sampling areas failed to capture larger (1+ and 2+) juveniles, the shallow inshore waters appear to less important as a nursery area than those of the inner-

and mid-shelf regions in south-eastern Tasmania (Jordan 1997). The full extent of distribution of 0+ fish can only be assessed after further targeted sampling in inshore waters.

## 6. Benefits

The major benefit of this study has been to provide information on both the habitat associations and biological parameters of economically important finfish species associated with coastal soft-sediment habitats throughout Tasmania. The use of multiple gear types with a large range of mesh sizes has further clarified the importance of each habitat by each life-history stage, particularly for sand flathead and southern sea garfish. The biological parameters defined for southern sea garfish and sand flathead will be incorporated into the further development of stock assessment models for these species.

The sampling of both unvegetated and seagrass habitats at a range of spatial and temporal scales has resulted in a greater understanding of the extent of natural fluctuations of fish assemblages in both habitat types at a scale relevant to management of such habitats throughout the State. Such information is particularly important in understanding the links between fisheries and habitats in coastal waters of Tasmania, particularly those that differ from elsewhere in southern Australia. The greater significance of unvegetated habitats compared to seagrass as a nursery area for economically important finfish species in Tasmania highlights the benefit of such regional studies. In addition, the study has resulted in a better understanding of the spatial scale that representative soft-sediment habitats should be included into the proposed system of marine reserves.

The limited mapping of seagrass habitats along the north coast identified around 530 $km^2$ of previously undocumented seagrass beds. While this clearly improves our understanding of the extent of seagrass beds throughout the State, it has highlighted the lack of information available on the distribution of such habitats and reflects the lack of habitat mapping at an appropriate spatial scale for effective management and monitoring.

The focus on habitat related issues in this project has also provided further stimulus for the development of habitat management guidelines which aim to improve the conservation and management of for estuarine and coastal habitats in Tasmania. A further objective will be to integrate habitat considerations and conservation into fishery management plans.

## 7. Further Development

While considerable advances were made in the present project in defining the life-history ecology and population parameters of several inshore demersal finfish species in Tasmania, there is a need for further studies in some key areas. Firstly, further research sampling of southern sea garfish will be required on the east coast of Tasmania before the size composition of the entire population can be assessed. In addition, further ageing studies are required in order to validate annuli, and examine spatial variations in growth and age

composition. Such information is necessary before an age and spatially-structured stock assessment model can be developed across the entire distribution of the commercial sea garfish fishery in Tasmania. Further examination of the spatial patterns of spawning in areas of extensive seagrass beds, such as Flinders Island, are also needed before the full extent of the spawning habitat requirements can be assessed.

Despite the detailed examination of the spawning and recruitment processes of sand flathead in the present study, there is a need for further work on the temporal pattern of spawning and factors leading to larval and post-settlement mortality. Before such work can be advanced, there is a need to resolve the otolith microstructure of sand flathead. Such studies may also assist in a better understanding of the factors resulting in year-class variability in the species.

This project has provided considerable information on the significance of coastal soft-sediment areas as fish habitats around Tasmania. The challenge now is to integrate this information in order to develop appropriate management measures that aim to minimise impacts on these habitats. In a recent survey of Tasmania's coastal zone planners, managers and local councils, it was clearly identified that at present, coastal managers are struggling to access the knowledge base required to manage Tasmania's coastal zone in a sustainable manner (Mount and Williamson 1996). What has impeded managers in the past has been the lack of scientific information on which to base sustainable management decisions in the coastal zone.

However, while this project provides considerable information on habitats and communities, the data are not in a form readily useable by managers, planners, local councils, government authorities and the wider community. Therefore, further work needs to be undertaken to synthesise this information with other research projects relevant to estuarine, coastal and marine habitat management into a single set of management guidelines. It is envisaged that such guidelines will cover such activities as reclamation and dredging, extractive operations, waterfront developments, marinas, catchment management, recreational facilities and point and diffuse source pollutants. The need to develop such guidelines has found substantial support amongst coastal managers and planners across Tasmania (Mount and Williamson 1996), and will assist in assessing proposals to ensure that they are sensitive to the estuarine, coastal and marine environments. It will also develop an awareness amongst the wider community of the sources of fisheries habitat problems and the actions required to remedy them. There is also a need to further disseminate information on habitat issues to the community through Coastcare and Fishcare activities.

The development of habitat guidelines are integral to fulfilling the objectives of sustainability detailed in Tasmania's *State Coastal Policy* and the *Living Marine Resources Management Act 1995*. An additional outcome will be to integrate this planning with

Catchment Management Plans to allow the cumulative impacts of activities within a catchment to be assessed. A further priority will be to further integrate habitat considerations and conservation into fishery management plans.

An additional key area of further development is the inclusion of coastal information gained in this project into both the Tasmanian Oil Spill Response Atlas and the Tasmanian node of the Australian Coastal Atlas. A 1996 review of the data requirements of the Atlas revealed considerable deficiencies in the area of coastal environments (eg. sandy beaches, intertidal areas) and habitats (eg. seagrass, kelp). Such data are now available but are yet to be centralised and put in a form readily available for inclusion in these coastal atlases, which are designed to support management with local up-to-date information. While there is a real need to incorporate the information from this project into both Atlases, there is currently insufficient resources to undertake such work.

Mapping of key coastal habitat around the state is also seen as an important area for further development in order to address the lack of information at the appropriate spatial scale for effective management. The lack of adequate coastal planning information has led to delays in the development of activities such as aquaculture and poor planning decisions that have resulted in environmental degradation, such as the loss of seagrass habitats. Tasmanian coastal zone planners and resource managers have identified habitat maps as the most important set of information required to support decisions leading to sustainable management and use of the coastal zone.

An additional outcome of a coastal habitat mapping project would be as an integral part of identifying representative habitats to be included in a system of marine protected areas around Tasmania. This is particularly important for soft-sediment areas as there is a real need to adequately represent these habitat types and their faunal assemblages in a marine reserve system. While a regional classification of Tasmania's coastal waters has recently been completed, the objective of incorporating representative habitats in each bioregion is still limited by the lack of knowledge of the distribution of such habitats, particularly seagrass beds. Detailed habitat maps at the appropriate scale would provide such information. They would also assist the planning for future marine farming areas by clearly identifying areas of vulnerable habitats.

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# 8. References

**Adams, S. M.** (1976). The ecology of eelgrass *Zostera marina* (L.) fish communities. I. Structural analysis. *Journal of Experimental Marine Biology and Ecology* **22**, 269-291.

**Ayvazian, S. G. and Hyndes, G. A.** (1995). Surf zone fish assemblages in south-western Australia: do adjacent nearshore habitats and the warm Leeuwin Current influence the characteristics of the fish fauna? *Marine Biology* **122**, 527-536.

**Bailey, K. M.** (1981). Larval transport and recruitment of Pacific Hake, *Merluccius productus*. *Marine Ecology Progress Series* **6**, 1-9.

**Beamish, R. J.** (1979). Differences in the age of Pacific hake (*Merluccius productus*) using whole and sectioned otoliths. *Journal of the Fisheries Research Board of Canada* **36**, 141-151.

**Beamish, R. J. and McFarlane, G. A.** (1983). The forgotten requirement of age validation in fisheries biology. *Transactions of the American Fisheries Society* **112**, 735-743.

**Beamish, R. J. and Fournier, D. A.** (1981). A method of comparing the precision of a set of age determinations. *Journal of the Fisheries Research Board of Canada* **36**, 1395-1400.

**Beckman, D. W., Wilson, C. A. and Stanley, A. L.** (1989). Age and growth of Red Drum, *Sciaenops ocellatus*, from offshore waters of the northern Gulf of Mexico. *Fishery Bulletin (US)* **87**, 17-28.

**Bell, J. D. and Pollard, D. A.** (1989). Ecology of fish assemblages and fisheries associated with seagrasses. In: *Biology of seagrasses: a treatise on the biology of seagrasses with special reference to the Australian region*. pp. 565-609. A. W. D. Larkum, A. J. McCoomb and S. Shepherd (Eds.). Elsevier: Amsterdam.

**Bell, J. D. and Westoby, M.** (1986a). Importance of local changes in leaf height and density to fish and decapods associated with seagrasses. *Journal of Experimental Marine Biology and Ecology* **104**, 249-274.

**Bell, J. D. and Westoby, M.** (1986b). Variation in seagrass height and density over a wide spatial scale: effects on common fish and decapods. *Journal of Experimental Marine Biology and Ecology* **104**, 275-295.

**Bell, J. D., Pollard, D. A., Burchmore, J. J., Pease, B. C. and Middleton, M. J.** (1984). Structure of a fish community in a temperate tidal mangrove creek in Botany Bay, New South Wales. *Australian Journal of Marine and Freshwater Research* **35**, 33-46.

**Bell, J. D., Ferrell, D. J., Mc Neill, S. E. and Worthington, D. G.** (1992). Variation in assemblages of fish associated with deep and shallow margins of the seagrass *Posidonia australis*. *Marine Biology* **114**, 667-676.

**Bell, J. D., Steffe, A. S. and Westoby, M.** (1988). Location of seagrass beds in estuaries: effects on associated fish and decapods. *Journal of Experimental Marine Biology and Ecology* **122**, 127-146.

**Bennett, B. A.** (1989). The fish community of a moderately exposed beach on the Southwestern Cape of South Africa and an assessment of this habitat as a nursery for juvenile fish. *Estuarine, Coastal and Shelf Science* **28**, 293-305.

**Blaber, S. J. M.** (1980). Fish of the Trinity Inlet system of north Queensland with notes on the ecology of fish faunas of tropical Indo-Pacific estuaries. *Australian Journal of Marine and Freshwater Research* **31**, 137-146.

**Blaber, S. J. M. and Whitfield, A. K.** (1977). The feeding ecology of juvenile mullet (Mugilidae) in south-eastern African estuaries. *Biological Journal of the Linnaean Society* **9**, 227-284.

**Blaber, S. J. M. and Blaber, T. G.** (1980). Factors affecting the distribution of juvenile estuarine and inshore fish. *Journal of Fish Biology* **17**, 143-162.

**Blackburn, M. and Gartner, P. E.** (1954). Populations of barracouta, *Thysites atun* (Euphrasen), in Australian waters. *Australian Journal of Marine and Freshwater Research* **5**, 411-68.

**Bray, J. R. and Curtis, J. T.** (1957). An ordination of the upland forest communities of southern Wisconsin. *Ecological Monographs* **27**, 325-349.

**Brown, I. W.** (1978). Ecology of three sympatric flatheads (Platycephalidae) in Port Phillip Bay, Victoria. Ph.D. Thesis, Monash University, Victoria

**Bulthuis, D. A. and Woelkerling, W. J.** (1983). Seasonal variation in standing crop, density and leaf growth rate of the seagrass *Heterozostera tasmanica*, in Western Port and Port Phillip Bay, Victoria, Australia. *Aquatic Botany* **16**, 111-136.

**Burchmore, J. J., Bell, J. D. and Pollard, D. A.** (1984). Community structure and trophic relationships of the fish fauna of an estuarine *Posidonia australis* seagrass habitat in Port Hacking, New South Wales. *Aquatic Botany* **18**, 71-87.

**Campana, S. E.** (1984). Comparison of age determination methods for the starry flounder. *Transactions of the American Fisheries Society* **113**, 365-369.

**Clarke, K. R.** (1993). Non-parametric multivariate analyses of change in community structure. *Australian Journal of Ecology* **18**, 117-43.

**Clarke, K. R. and Green, R. H.** (1988). Statistical design and analyses for a "biological effects" study. *Marine Ecology Progress Series* **46**, 213-226.

**Clarke, K. R. and Warwick, R. M.** (1994). Changes in marine communities: an approach to statistical analysis and interpretation. Bourne Press Ltd, Bournemouth.

**Clarke, B. M.** (1997). Variation in surf-zone fish community structure across a wave-exposure gradient. *Estuarine, Coastal and Shelf Science* **44**, 659-674.

**Collins, M. R., Schmidt, D. J., Waltz, C. W. and Pickney J. L.** (1988). Age and growth of the king mackerel, *Scomberomorus cavalla*, from the Atlantic coast of the United States. *Fishery Bulletin (US)* **87**, 49-61.

**Connolly, R. M.** (1994). A comparison of fish assemblages from seagrass and unvegetated areas of a southern Australian estuary. *Australian Journal of Marine and Freshwater Research* **45**, 1033-1044.

**Cowen, R. K.** (1985). Large scale pattern of recruitment by the labrid *Semicosstphus pulcher*: causes and implications. *Journal of Marine Research* **43**, 719-742.

**Cowen, R. K.** (1991). Variation in the planktonic larval duration of the temperate wrasse *Semicossyphus pulcher*. *Marine Ecology Progress Series* **69**, 9-15.

**Crawford, C. M.** (1984). An ecological study of Tasmanian flounder. Ph.D. Thesis, University of Tasmania, Hobart.

**Davenport, S. and McLoughlin, R.** (1993). Preliminary assessment of the distribution and potential impact of the introduced starfish *Asterias amurensis* in Tasmanian waters. Status report to Fisheries Research and Development Corporation. June 1993. 37 pp.

**Day, R. W. and Quinn, G. P.** (1989). Comparison of treatments after an analysis of variance in ecology. *Ecological monographs* **59**, 433-463.

**De Ben, W. A., Clothier, W. D., Ditsworth, G. R. and Baumgartner, D. J.** (1990). Spatio-temporal fluctuations in the distribution and abundance of demersal fish and epibenthic crustaceans in Yaquina Bay, Oregon. *Estuaries* **13**, 469-478.

**Edgar, G. J.** (1990). The influence of plant structure on the species richness, biomass and secondary production of macrofaunal assemblages associated with Western Australian seagrass beds. *Journal of Experimental Marine Biology and Ecology* **137**, 215-40.

**Edgar, G. J. and Shaw, C.** (1995a). The production and trophic ecology of shallow-water fish assemblages in southern Australia. I. Species richness, size structure and production of fishes in Western Port, Victoria. *Journal of Experimental Marine Biology and Ecology* **194**, 53-81.

**Edgar, G. J. and Shaw, C.** (1995b). The production and trophic ecology of shallow-water fish assemblages in southern Australia. II. Diets of fishes and trophic relationships between fishes and benthos in Western Port, Victoria. *Journal of Experimental Marine Biology and Ecology* **194**, 83-106.

**Edgar, G. J. and Shaw, C.** (1995c). The production and trophic ecology of shallow-water fish assemblages in southern Australia. III. General relationships between sediments, seagrasses, invertebrates and fishes. *Journal of Experimental Marine Biology and Ecology* **194**, 107-131.

**Edgar, G. J., Shaw, C., Watson, G. F. and Hammond, L. S.** (1994). Comparison of species richness, size structure and production of benthos in vegetated and unvegetated habitats in Western Port, Victoria. *Journal of Experimental Marine Biology and Ecology* **176**, 201-26.

**Ferrell, D. J. and Bell, J. D.** (1991). Differences among the assemblages of fish associated with *Zostera capricorni* and bare sand over a large spatial scale. *Marine Ecology Progress Series* **72**, 15-24.

**Ferrell, D. J., Mc Neill, S. E., Worthington, D. G. and Bell, J. D.** (1993). Temporal and spatial variation in the abundance of fish associated with the seagrass *Posidonia australis* in south-eastern Australia. *Australian Journal of Marine and Freshwater Research* **44**, 881-899.

**Gomon, M. F., Glover, J. C. M. and Kuiter, R. H.** (1994). The fishes of Australia's south coast. State Print, Adelaide.

**Gray, C. A. and Bell, J. D.** (1986). Consequences of two common techniques for sampling vagile macrofauna associated with the seagrass *Zostera capricorni*. *Marine Ecology Progress Series* **28**, 43-48.

**Gray, C. A., McDonall, V. C. and Reid, D. D.** (1990). By-catch from prawn trawling in the Hawkesbury River, New South Wales: Species composition, distribution and abundance. *Australian Journal of Marine and Freshwater Research* **41**, 13-26.

**Gray, C. A., McElligott, D. J. and Chick, R. C.** (1996). Intra- and inter-estuary differences in assemblages of fishes associated with shallow seagrass and bare sand. *Marine and Freshwater Research* **47**, 723-735.

**Hamdorf, I. and Kirkman, H.** (1995). Status of Australian Seagrass. Issues Paper -March 1995. Fisheries Pollution and Marine Environment Committee Issues Paper, 32.

**Harris, G. P., Nilsson, C., Clementson, L. A. and Thomas, D.** (1987). The water masses of the east coast of Tasmania: seasonal and interannual variability and the influence on phytoplankton biomass and productivity. *Australian Journal of Marine and Freshwater Research* **38**, 569-90.

**Harris, G. P., Griffiths, F. B., Clementson, L. A., Lyne, V. and Van der Doe, H.** (1991). Seasonal and interannual variability in physical processes, nutrient cycling and the structure of the food chain in Tasmanian shelf waters. *Journal of Plankton Research* **13** Suppl., 109-31.

**Heck, K. L. Jr. and Orth, R. J.** (1980). Seagrass habitats: the roles of habitat complexity, competition and predation in structuring associated fish and motile macroinvertebrate assemblages. pp. 449-64. In: *Estuarine Perspectives*. V. S. Kennedy (Ed). Academic Press, New York.

**Heck, K. L. Jr. Able, K. W., Fahay, M. P. and Roman, C. T.** (1989). Fishes and decapod crustaceans of Cape Cod eelgrass meadows: species composition, seasonal abundance patterns and comparison with unvegetated substrates. *Estuaries* **12**, 59-65.

**Humphries, P., Potter, I. C. and Loneragan, N. R.** (1992). The fish community in the shallows of a temperate Australian estuary: relationships with the aquatic macrophyte *Ruppia megacarpa* and environmental variables. *Estuarine, Coastal and Shelf Science* **34**, 325-346.

**Hyndes, G. A. and Potter, I. C.** (1996). Comparisons between the age structures, growth and reproductive biology of two co-occurring sillaginids, *Sillago robusta* and *S. bassensis*, in temperate coastal waters of Australia. *Journal of Fish Biology* **49**, 14-32.

**Hyndes, G. A., Neira, F. J. and Potter, I. C.** (1992a). Reproductive biology and early life-history of the marine teleost *Platycephalus speculator* Klunzinger (Platycephalidae) in a temperate Australian estuary. *Journal of Fish Biology* **40**, 859-874.

**Hyndes, G. A., Loneragan, N. R. and Potter, I. C.** (1992b). Influencing of sectioning otoliths on marginal increment trends and age and growth estimates for the flathead *Platycephalus speculator. Fishery Bulletin (US)* **90**, 276-284.

**Hyndes, G. A., Potter, I. C. and Lenanton, R. C. J.** (1996). Habitat partitioning by whiting species (Sillaginidae) in coastal waters. *Environmental Biology of Fish* **45**, 21-40.

**Jenkins, G. P.** (1986). Composition, seasonality and distribution of ichthyoplankton in Port Phillip Bay, Victoria. *Australian Journal of Marine and Freshwater Research* **37**, 507-520.

**Jenkins, G. P. and Black, K. P.** (1994). Temporal variability in settlement of a coastal fish, the King George whiting, *Sillaginodes punctata*, is determined by low-frequency hydrodynamics. *Limnology and Oceanography* **39**, 1744-1754.

**Jenkins, G. P. and May, H. M. A.** (1994). Variation in settlement and larval duration of King George whiting, *Sillaginodes punctata* (Sillaginidae), in Swan Bay, Victoria, Australia. *Bulletin of Marine Science* **54**, 281-296.

**Jenkins, G. P. and Sutherland, C. R.** (1997). The influence of habitat structure on nearshore fish assemblages in a southern Australian embayment: colonisation and turnover rate if fishes associated with artificial macrophyte beds of varying physical structure. *Journal of Experimental Marine Biology and Ecology* **218**, 103-125.

**Jenkins, G. P., Watson, G. F., Hammond, L. S., Black, K. P., Wheatley, M. J. and Shaw, C.** (1996). Importance of shallow water, reef-algal habitats as nursery areas for commercial fish from southeastern Australia. Final Report FRDC Project 92/44, 166 pp.

**Jenkins, G. P., May, H. A., Wheatley, M. J. and Holloway, M. G.** (1997). Comparison of fish assemblages associated with seagrass and adjacent unvegetated habitats of Port Phillip Bay and Corner Inlet, Victoria, Australia, with emphasis on commercial species. *Estuarine and Coastal Shelf Science* **44**, 569-588.

**Jones, G. K.** (1990). Growth and mortality in a lightly fishes population of garfish (*Hyporhamphus melanochir*), in Baird Bay, South Australia. *Transactions of the Royal Society of South Australia* **144**, 37-41.

**Jordan, A. R.** (1994a). Flatheads (Family: Platycephalidae). In: *Species Status Reports: Key scalefish species*. Department of Primary Industry and Fisheries, Marine Resources Division Internal Report No. 14.

**Jordan, A. R.** (1994b). Age, growth and back-calculated birthdate distributions of larval jack mackerel, *Trachurus declivis* (Pisces: Carangidae), from eastern Tasmanian coastal waters. *Australian Journal of Marine and Freshwater Research* **45**, 19-33.

**Jordan, A. R.** (1995). Interim report: Effect of the Iron Baron spill - Program 6B. Assessment of seagrass beds in the Tamar River and selected sites between Low Head and Five Mile Bluff. *Taroona Marine Research Laboratories Advice Note December 95*.

**Jordan A. R.** (1997). Demersal trawl surveys of the continental shelf of southern and eastern Tasmania 1993-95. *Technical Report of the Marine Laboratories Marine Research Division Tasmania*. **50**, 148 pp.

**Jordan, A. R., Richards, W. and Neira, F. J.** (1998). Family Triglidae. In: *The larvae of temperate Australian fishes: a laboratory guide for larval fish identification*. pp. 150-162. Neira, F. J., Miskiewicz, A. G. and Trnski, T. (eds). University of Western Australian Press, Nedlands WA.

**Kailola, P. J., Williams, M. J., Stewart, P. C., Reichelt, R. E., McNee, A. and Grieve, C.** (1993). *Australian Fisheries Resources*. Bureau of Resources Sciences and Fisheries Research and Development Corporation, Canberra. 422 pp.

**Kirkman, H. and Reid, D. D.** (1979). A study of the seagrass *Posidonia australis* in the carbon budget of an estuary. *Aquatic Botany* **7**, 173-183.

**Klumpp, D. W. and Nichols, P. D.** (1983). Nutrition of the southern sea garfish (*Hyporhamphus melanochir)*: gut passage rate and daily consumption of two food types and assimilation of seagrass components. *Marine Ecology Progress Series* **12**, 207-216.

**Kurth, D.** (1957). An investigation of the greenback flounder, *Rhombosolea tapirina* (Gunther). Ph.D. Thesis, University of Tasmania, Hobart.

**Larkum, A. W. D., Collett, L. C. and Williams, R. J.** (1984). The standing crop, growth and shoot production of *Zostera capricorni* Ashers. in Botany Bay, New South Wales, Australia. *Aquatic Botany* **19**, 307-327.

**Last, P.** (1983). Aspects of the ecology and zoogeography of fishes from soft-bottom habitats of the Tasmanian shore zone. Ph.D. Thesis, University of Tasmania, Hobart, Australia.

**Leis, J. M. and Rennis, D. S.** (1983). The larvae of Indo-Pacific coral reef fishes. University of New South Wales Press, Sydney, 269 p.

**Leis, J. M. and Trnski, T.** (1989). The larvae of Indo-Pacific shorefishes. University of New South Wales Press, Sydney, 371 pp.

**Levin, P. S.** (1993). Habitat structure, conspecific presence and spatial variation in recruitment of a temperate reef fish. *Oecologia* **94**, 176-185.

**Lincoln Smith, M. P., Bell, J. D. and Hair, C. A.** (1991). Spatial variation in abundance of recently settled rocky reef fish in southeastern Australia: implications for detecting change. *Marine Ecology Progress Series* **77**, 95-103.

**Ling, J. K.** (1958). The sea garfish, *Reporhamphus melanochir* (Cuvier & Valenciennes) (Hemiramphidae), in South Australia: breeding, age determination and growth rate. *Australian Journal of Marine and Freshwater Research* **9**, 60-111.

**Livingston, M. E. and Schofield, K. A.** (1995). Trawl survey of hoki and associated species on the Chatham Rise, November-December 1989. *New Zealand Fisheries Technical Report* No. 41, 31pp.

**Loneragan, N. R., Potter, I. C., Lenanton, R. C. J. and Caputi, N.** (1986). Spatial and seasonal differences in the fish fauna in the shallow of a large Australian estuary. *Marine Biology* **92**, 575-586.

**Love, M. S., Carr, M. H. and Haldorson, L. J.** (1991). The ecology of substrate associated juveniles of the genus *Sebastes*. *Environmental Biology of Fishes* **30**, 225-243.

**Lyle, J. M.** (1994). Fishery Status Report: Inshore trawl fishery. *Department of Primary Industry and Fisheries, Marine Resources Division Internal Report* No. 9, 75 pp.

**Lyle, J. M.** (1998). Overview of Tasmanian scalefish fishery based on 1995-1997 catch returns. *Stock Assessment Background Report* **98/02**, 33 pp.

**Lyle, J. M. and Ford, W. B.** (1993). Review of trawl research 1979-87, with summaries of biological information for the major trawl species. *Technical Report of the Marine Laboratories Marine Research Division Tasmania* **46**, 169 pp.

**Lyle, J. M. and Smith, J. T.** (1998). Pilot survey of licensed recreational sea fishing in Tasmania - 1995/96. *Technical Report of the Marine Laboratories Marine Research Division Tasmania* **51**, 55 pp.

**McLachlan, A., Erasmus, T., Dye, A. H., Wooldridge, T., van der Horst, G., Rossouw, G., Lasiak. T. A. and McGynne, L.** (1981). Sand beach energetics: an ecosystem approach towards a high energy interface. *Estuarine Coastal Shelf Science* **13**, 11-25.

**McNeill, S. E. and Bell, J. D.** (1992). Comparison of beam trawls for sampling macrofauna of *Posidonia* seagrass. *Estuaries* **15**, 360-367.

**Marshall, J. and Jordan, A. R.** (1992). A catalogue of ichthyoplankton from eastern Tasmanian waters. *Department of Sea Fisheries Tasmania Occasional Publication* **6**, 1-43.

**Martin, F. D. and Cooper, M.** (1981). A comparison of fish faunas found in pure stands of two tropical Atlantic seagrasses, *Thalassia testudinum* and *Syringodium filiforme*. *Northeast Gulf Science* **5**, 31-37.

**Martin, S. A. and Toothaker, L. E.** (1989). PERITZ: A FORTRAN program for performing multiple comparisons of means using the Peritz Q method. *Behaviour Research Methods, Instruments and Computers* **21**, 465-472.

**Massey, B. R. and Horn, P. L.** (1990). Growth and age structure of alfonsino (*Beryx splendens*) from the lower east coast, North Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research* **24**, 121-136.

**May, H. M. A. and Jenkins, G. P.** (1992). Patterns of settlement and growth of juvenile flounder, *Rhombosolea tapirina*, determined from otolith microstructure. *Marine Ecology Progress Series* **79**, 203-214.

**Middleton, M. J. , Bell, J. D., Burchmore, J. J., Pollard, D. A. and Pease, B. C.** (1984). Structural differences in the fish communities of *Zostera capricorni* and *Posidonia*

*australis* seagrass meadows in Botany Bay, New South Wales. *Aquatic Botany* **18**, 89-109.

**Milicich, M. J.** (1994). Dynamic coupling of reef fish replenishment and oceanographic processes. *Marine Ecology Progress Series* **110**, 135-144.

**Mills, D.** (1992). Habitat utilisation by fish in seagrass beds. BSc. Honours thesis, University of Tasmania. 70pp.

**Mount, R. E. and Williamson, S.** (1996). Needles in a haystack: the information requirements of Tasmania's coastal managers. Report to Ocean Rescue 2000, Project No. D705, Australian Nature Conservation Agency, Canberra, 39 pp.

**Neira, F. J. and Furlani, D.** (1998). Family Scorpaenidae. In: *The larvae of temperate Australian fishes: a laboratory guide for larval fish identification.* pp. 140-149. Neira, F. J., Miskiewicz, A. G. and Trnski, T. (Eds). University of Western Australian Press, Nedlands WA.

**Neira, F J. and Miskiewicz, T.** (1998). Family Platycephalidae. In: *The larvae of temperate Australian fishes: a laboratory guide for larval fish identification.* pp. 134-139. Neira, F. J., Miskiewicz, A. G. and Trnski, T. (Eds). University of Western Australian Press, Nedlands WA.

**Neira, F J., Coutin, P. C. and Elliot-Tippet, D.** (1997). Commercial catches of selected scalefish species from Victorian waters, 1978-1995. *Marine and Freshwater Resources Institute Report* No. 2, 64 pp.

**Nelson, W. R., Ingham, M. C. and Schaaf, W. E.** (1977). Larval transport and year-class strength of Atlantic menhaden, *Brevoortia tyrannus*. *Fishery Bulletin (US)* **75**, 23-41.

**Orth, R. J., Heck, K. L. and Van Montfrans, J.** (1984). Faunal communities in seagrass beds: a review of the influence of plant structure and prey characteristics on predator-prey relationships. *Estuaries* **7A**, 339-350.

**Potter, I. C. And Hyndes, G. A.** (1994). Composition of the fish fauna of a permanently open estuary in the southern coast of Australia, and comparisons with a nearby seasonally open estuary. *Marine Biology* **121**, 199-209.

**Potter, I. C., Beckley, L. E., Whitfield, A. K. and Lenanton, R. C. J.** (1990). Comparisons between the roles played by estuaries in the life cycles of fishes in temperate Western Australia and Southern Africa. In: *Alternative life history styles of fishes.* pp 143-178. Bruton, M. N (Ed.).

**Ratkowsky, D.** (1983). Nonlinear regression modelling. Marcel Dekker, Inc., New York, NY. 276 p.

**Rees, C.** (1993). Tasmanian seagrass communities. M.Env.Sc. Thesis. University of Tasmania Centre for Environmental Studies.

**Robertson, A. I.** (1977). Ecology of juvenile King George Whiting *Sillaginodes punctatus* (Cuvier and Valenciennes). *Australian Journal of Marine and Freshwater Research* **28**, 35-43.

**Robertson, A. I. and Lenanton, R. C. J.** (1984). Fish community structure and food chain dynamics in the surf-zone of sandy beaches: The role of detached macrophyte detritus. *Journal of Experimental Marine Biology and Ecology* **84**, 265-283.

**St. Hill, J. L.** (1996). Aspects of the biology of the southern sea garfish, *Hyporhamphus melanochir,* in Tasmania waters. BSc. Honours thesis, University of Tasmania. 70pp.

**Shaw, M. and Jenkins, G. P.** (1992). Spatial variation in feeding, prey distribution and food limitation of juvenile flounder, *Rhombosolea tapirina* Guenther. *Journal of Experimental Marine Biology and Ecology* **165**, 1-21.

**Shepherd, S. A. and Womersley, H. B. S.** (1981). The algal and seagrass ecology of Waterloo Bay, South Australia. *Aquatic Botany* **11**, 305-371.

**Shepherd, S. A. and Robertson, E. L.** (1989). Regional studies - seagrasses of South Australia, Western Victoria and Bass Strait. In: *Biology of seagrasses. a treatise on the biology of seagrasses with special reference to the Australian region.* pp. 211-225. A. W. D. Larkum, McComb, A. J. and Shepherd, S. A. (Eds). New York Amsterdam, Elsevier,

**Sogard, S. M. and Able, K. W.** (1991). A comparison of eelgrass, sea lettuce macroalgae, and marsh creeks as habitats for epibenthic fishes and decapods. *Estuarine, Coastal and Shelf Science* 33, 501-519.

**Sogard S. M., Powell, G. V. N. and Holmquist, J. G.** (1987). Epibenthic fish communities on Florida Bay banks: relations with physical parameters and seagrass cover. *Marine Ecology Progress Series* **40**, 25-39.

**Sogard S. M., Powell, G. V. N. and Holmquist, J. G.** (1989). Spatial distribution and trends in abundance of fishes residing in seagrass meadows on Florida Bay mudbanks. *Bulletin of Marine Science* **44**, 179-199.

**Sokal, R. R. and Rohlf, F. J.** (1981). Biometry. W. H. Freeman and Company, New York. 859 p.

**Stoner, A. W.** (1983). Distribution of fishes in seagrass meadows; role of macrophyte biomass and species composition. *Fishery Bulletin (US)* **81**, 837-846.

**Thomson, J. M.** (1957). The size at maturity and spawning times of some Western Australian estuarine fishes. *Western Australian Fisheries Department Fisheries Bulletin* **8**, 1-8.

**Walker, D. I. and McComb, A. J.** (1988). Seasonal variation in the productivity, biomass and nutrient status of *Amphibolis antarctica* (Labill.) Sonder ex Aschers. and *Posidonia australis* Hook f. in Shark Bay, Western Australia. *Aquatic Botany* **31**, 259-275.

**Warburton, K. and Blaber, S. J. M.** (1992) Patterns of recruitment and resource use in a shallow-water fish assemblage in Moreton Bay, Queensland. *Marine Ecology Progress Series* **90**, 113-126.

**Winer, B. J.** (1971). 'Statistical Principles in Experimental Design.' (NcGraw-Hill Kogakusha: Tokyo.)

**Weinstein, M. P.** (1979). Shallow marsh habitat as primary nurseries for fish and shellfish, Cape Fear River, North Carolina. *Fishery Bulletin (US)* **77**, 339-357.

**Weinstein, M. P. and Heck, Jr, K. L.** (1979). Ichthyofauna of seagrass meadows along the Caribbean coast of Panama and in the Gulf of Mexico: composition, structure and community ecology. *Marine Biology* **50**, 97-107.

**Wells, F. E. and Rose, R. A.** (1985). An analysis of benthic invertebrate communities in subtidal seagrass and sand habitats in Shark Bay, Western Australia. *Records of the Western Australian Museum* **12**, 47-56.

**Worthington, D. G., Ferrell, D. J., McNeill, S. E. and Bell, J. D.** (1992). Effects of the shoot density of seagrass on fish and decapods: Are correlations evident over larger spatial scales? *Marine Biology* **116**, 139-146.

**Young, P. C.** (1981). Temporal changes in the vagile epibenthic fauna of two seagrass meadows (*Zostera capricorni* and *Posidonia australis*). *Marine Ecology Progress Series* **5**, 91-102.

**Zar, J. H.** (1996). Biostatistical analysis. Prentice Hall Inc, Englewood Cliffs, N.J.

## Intellectual property

No processes have been developed during the course of this project with all results to be published in relevant scientific publications and other literature in the public domain.

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* FRDC funded positions

Appendix 1. List of common names, taxonomic names and CSIRO identification code for species caught in the study.

| Family | Common Name | Scientific name | CSIRO Code |
|---|---|---|---|
| Scyliorhinidae | Draughtboard Shark | *Cephaloscyllium laticeps* | 015001 |
| Triakidae | Gummy Shark | *Mustelus antarcticus* | 017001 |
| | School Shark | *Galeorhinus galeus* | 017008 |
| Squalidae | White Spotted Dogfish | *Squalus acanthias* | 020008 |
| Pristiophoridae | Southern Sawshark | *Pristiophorus nudipinnis* | 023001 |
| | Common Sawshark | *Pristiophorus cirratus* | 023002 |
| Rajidae | Thornback Skate | *Raja lemprieri* | 031007 |
| | Whitley's Skate | *Raja whitleyi* | 031006 |
| Urolophidae | Banded Stingaree | *Urolophus cruciatus* | 038002 |
| | Sparsely Spotted Stingaree | *Urolophus paucimaculatus* | 038004 |
| Anguillidae | Long-Finned Eel | *Anguilla reinhardtii* | 056002 |
| Galaxiidae | Common Jollytail | *Galaxias maculatus* | 102006 |
| | Spotted Mountain Galaxias | *Galaxias truttaceus* | 102010 |
| Gobiesocidae | Common Shore Eel | *Alabes dorsalis* | 206008 |
| Moridae | Eucla Cod | *Euclichthys polynemus* | 224001 |
| | Bearded Rock Cod | *Pseudophycis barbatus* | 224003 |
| | Red Cod | *Pseudophycis bachus* | 224006 |
| Ophidiidae | Rock Ling | *Genypterus tigerinus* | 228008 |
| Atherinidae | Small-Mouthed Hardyhead | *Atherinosoma microstoma* | 246001 |
| | Silverfish | *Leptatherina presbyteroides* | 246002 |
| | Pike-Headed Hardyhead | *Kestratherina esox* | 246003 |
| | Richardson's Hardyhead | *Atherinason hepsetoides* | 246004 |
| Zeidae | Silver Dory | *Cyttus australis* | 264002 |
| Syngnathidae | Common Seadragon | *Phyllopteryx taeniolatus* | 282002 |
| | Pot Bellied Seahorse | *Hippocampus abdominalis* | 282010 |
| | Brigg's Crested Pipefish | *Histiogamphelus briggsi* | 282011 |
| | Knife-Snout Pipefish | *Hypselognathus rostratus* | 282012 |
| | Brush-Tailed Pipefish | *Leptoichthys fistularius* | 282013 |
| | Half-Banded Pipefish | *Mitotichthys semistriatus* | 282015 |
| | Spotted Pipefish | *Stigmatopora argus* | 282017 |
| | Wide-Bodied Pipefish | *Stigmatopora nigra* | 282018 |
| | Mollison's Pipefish | *Mitotichthys mollisoni* | 282022 |
| | Port Phillip Pipefish | *Vanacampus phillipi* | 282023 |
| | Long-Snouted Pipefish | *Vanacampus poecilolaemus* | 282024 |
| | Short-Headed Seahorse | *Hippocampus breviceps* | 282026 |
| Scorpaenidae | Thetis Fish | *Neosebastes thetidis* | 287006 |
| | Red Rock Cod | *Helicolenus barathri* | 287008 |
| | Soldierfish | *Gymnapistes marmoratus* | 287018 |
| Triglidae | Spiny Gurnard | *Lepidotrigla papilio* | 288002 |
| | Round Snouted Gurnard | *Lepidotrigla mulhalli* | 288008 |
| Aploactinidae | Velvet Fish | *Aploactisoma milesii* | 290001 |
| Platycephalidae | Tiger Flathead | *Neoplatycephalus richardsoni* | 296001 |
| | Sand Flathead | *Platycephalus bassensis* | 296003 |
| | Rock Flathead | *Platycephalus laevigatus* | 296006 |
| | Yank Flathead | *Platycephalus speculator* | 296037 |
| Apogonidae | Wood's Siphon Fish | *Siphamia cephalotes* | 327032 |
| | Southern Cardinal Fish | *Vincentia conspersa* | 327033 |

| | | | |
|---|---|---|---|
| Sillaginidae | Eastern School Whiting | *Sillago flindersi* | 330014 |
| Carangidae | Silver Trevally | *Pseudocaranx dentex* | 337062 |
| Arripidae | Eastern Australian Salmon | *Arripis trutta* | 344002 |
| Sparidae | Black Bream | *Acanthopagrus butcheri* | 353003 |
| Mullidae | Southern Goatfish | *Upeneichthys vlamingii* | 355029 |
| Mugilidae | Yellow Eye Mullet | *Aldrichetta forsteri* | 381001 |
| Labridae | Castlenau's Wrasse | *Dotalabrus aurantiacus* | 384018 |
| | Rosy Wrasse | *Pseudolabrus psittaculus* | 384023 |
| Odacidae | Blue Rock Whiting | *Haletta semifasciata* | 385002 |
| | Slender Rock Whiting | *Siphognathus attenuatus* | 385004 |
| | Little Rock Whiting | *Neoodax balteatus* | 385005 |
| | Pigmy Rock Whiting | *Siphognathus beddomei* | 385006 |
| | Long Rayed Rock Whiting | *Siphonognathus radiatus* | 385007 |
| Leptoscopidae | Common Sandfish | *Lesueurina platycephala* | 398001 |
| | Pink Sandfish | *Crapatalus munroi* | 398002 |
| Bovichthyidae | Congolli | *Pseudaphritis urvilli* | 403003 |
| Blennidae | Blenny | *Parablennius tasmanianus* | 408002 |
| Clinidae | Crested Weedfish | *Cristiceps australis* | 416007 |
| | Adelaide Weedfish | *Heteroclinus adelaidae* | 416008 |
| | Longnose Weedfish | *Heteroclinus tristis* | 416009 |
| | Common Weedfish | *Heteroclinus perspicillatus* | 416013 |
| | The Girls' Weedfish | *Heteroclinus puellarum* | 416014 |
| Gobiidae | Tamar Goby | *Favonigobius tamarensis* | 428004 |
| | Orange-Spotted Goby | *Nesogobius hinisbyi* | 428006 |
| | Castelnau's Goby | *Nesogobius pulchellus* | 428007 |
| | Bridled Goby | *Arenigobius bifrenatus* | 428008 |
| | Blue-Spotted Goby | *Pseudogobius olorum* | 428009 |
| | Girdled Goby | *Nesogobius sp.1* | 428195 |
| | Twin-Barred Goby | *Nesogobius sp.3* | 428196 |
| | Opalescent Goby | *Nesogobius sp.6* | 428197 |
| | Lagoon Goby | *Tasmanogobius lasti* | 428262 |
| Bothidae | Small Toothed Flounder | *Pseudorhombus jenynsii* | 460002 |
| Pleuronectidae | Long Snouted Flounder | *Ammotretis rostratus* | 461001 |
| | Greenback Flounder | *Rhombosolea tapirina* | 461003 |
| | Spotted Flounder | *Ammotretis liturata* | 461004 |
| | Derwent Flounder | *Taratretis derwentensis* | 461011 |
| Monacanthidae | Toothbrush Leatherjacket | *Acanthaluteres vittiger* | 465002 |
| | Rough Leatherjacket | *Scobinichthys granulatus* | 465007 |
| | Pigmy Leatherjacket | *Brachaluteres jacksonianus* | 465025 |
| | Gunn's Leatherjacket | *Eubalichthys gunnii* | 465034 |
| | Six-Spined Leatherjacket | *Meuschenia freycineti* | 465036 |
| | Bridled Leatherjacket | *Acanthaluteres spilomelanurus* | 465043 |
| | Barred Toadfish | *Contusus richei* | 467001 |
| | Smooth Toadfish | *Tetractenos glaber* | 467003 |
| | Prickly Toadfish | *Contusus brevicaudas* | 467044 |
| Diodontidae | Globe Fish | *Diodon nichthemerus* | 469001 |