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

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Species richness, abundance and biomass of fishes from seagrass and unvegetated habitats were analysed for sites in the Swan Bay region of Port Phillip Bay, and Corner Inlet, Victoria, Australia. Fish assemblages from intertidal Zostera muelleri sites showed greater similarity with a number of shallower unvegetated sites than to subtidal Heterozostera tasmanica. Similarly, subtidal, unvegetated sites in Swan Bay were more similar to subtidal Heterozostera sites than to other, generally shallower, unvegetated sites. Species richness was significantly higher in seagrass beds and showed an increase over the warmer months of the year. Differences in abundance and biomass of fishes between vegetated and unvegetated habitats depended on the location of sampling. In general, abundance and biomass of fish in the deeper subtidal seagrass were significantly higher than in unvegetated habitats of the same depth. In contrast, abundance and biomass were not significantly different between seagrass and unvegetated habitats of the intertidal and shallow subtidal zones when significant abundances of the long-finned goby, Favonigobius lateralis, occurred on unvegetated habitats. As a consequence, species richness would be expected to decrease with any seagrass loss; however, decreases in abundance and biomass of fishes would be greatest with loss of deeper subtidal Heterozostera habitats. Although juveniles of a number of commercially important species were associated with subtidal Heterozostera habitats, juveniles of other species of commercial significance were found on shallow, unvegetated habitats. Newly settled juveniles of the King George whiting, Sillaginodes punctata, were associated with unvegetated patches amongst subtidal seagrass in Swan Bay, in contrast with a direct association with seagrass reported from other areas.

Keywords: fish; sea grasses; community composition; commercial species; Australia south coast

Introduction

Seagrass habitats are widely regarded to be important habitats for fishes, including juveniles of commercially important species (Bell & Pollard, 1989). These habitats are thought to provide refuge from predation, and enhanced production of food either through increased production of epifauna on blades or indirectly through the generation of detritus. One method of assessing the importance of seagrass habitats to fishes is by comparison with other habitats, particularly unvegetated habitats (Ferrell & Bell, 1991).

In general, previous studies have suggested that seagrass habitats contain a greater diversity and abundance of fishes than adjacent unvegetated habitats (Bell & Pollard, 1989 and references therein; Sogard & Able, 1991; Connolly, 1994a). A number of studies have, however, produced results contradicting this paradigm. No significant difference in the number of fish species in seagrass and unvegetated sites was found in a turbid estuary in South Africa (Hanekom & Baird, 1984). Heck and Thoman (1984) found no significant difference in fish abundances between vegetated and unvegetated habitats in Chesapeake Bay. Humphries et al. (1992) found that the seagrass Ruppia supported higher abundances and biomass of juvenile fishes, but a lower species diversity than bare sand areas. Ferrell and Bell (1991) found that the seagrass Zostera contained similar numbers, but more species, of fishes compared with adjacent sand areas. Both habitats contained more species and individuals than distant sand habitats. An important finding was that patterns varied depending on the site sampled; many previous studies have concentrated on one site, or sites within a limited geographic area.

Apart from abundance and species richness, species composition in seagrass and unvegetated habitat also tends to show marked differences. In general, species composition in seagrass includes many small species with cryptic habits, together with juveniles of larger...
species and some large mobile fish; in contrast, species gaining protection through schooling or camouflage are most abundant on bare substrate (Bell & Pollard, 1989).

Understanding the patterns of distribution and abundance of fishes in seagrass and unvegetated habitat in Victoria is particularly pertinent given the major (70%) decline in *Heterozostera* beds in Westernport (Bulthuis et al., 1984; Shepherd et al., 1989), which is situated between Port Phillip Bay and Corner Inlet on the Victorian coast. This decline occurred between about 1970 and 1984, and was paralleled by a decline in the total commercial catch of finfish in the bay (MacDonald, 1992). Catches of some commercial species, such as King George whiting, *Sillaginodes punctata*, leatherjackets (Monacanthidae) and blue rock whiting, *Haletta semifasciata*, showed a dramatic decline (MacDonald, 1992). Other species, such as yellow-eye mullet, *Aldrichetta forsteri*, and Australian salmon, *Arrips* spp., showed no decline in catch following the loss of seagrass (MacDonald, 1992).

The aim of the present study was to compare abundance, species richness and biomass of fishes between seagrass and adjacent unvegetated habitats in southern Australia. The null hypothesis was that no differences in these parameters would be observed between these habitats. Sampling was conducted at sites in the Swan Bay region of Port Phillip Bay, and Corner Inlet, Victoria, Australia. Seagrass sites consisted of either *Zostera muelleri*, which generally occurs intertidally, or *Heterozostera tasmanica*, which generally occurs subtidally. By examining variability for a number of sites and sampling periods, more confidence could be placed in the generality of any patterns observed with studies covering a restricted number of sites and sampling periods. Such sampling allowed inferences to be made about the possible effects of seagrass habitat loss on fish communities. For example, will species richness, abundance and biomass of fish decline in areas of seagrass loss? The relationship between recruitment of some important commercial species and seagrass habitat was examined in detail.

**Methods**

**Study areas**

Swan Bay (Figure 1) is a small bay linked to Port Phillip Bay in the region of Port Phillip Heads. Swan Bay is shallow (<3 m), relatively protected from the prevailing westerly winds, and tidal currents are weak. The central region of Swan Bay tends to unvegetated sediment with large patches of drift algae, much of
which is unattached but still growing. Seagrass on the adjacent coast of Port Phillip Bay consists of narrow (20 m wide) bands of *Heterozostera* running parallel to the shoreline. This coast is more exposed to wave action and strong tidal currents associated with Port Phillip Heads. Cover of macro-algae is very low compared with Swan Bay. Sediments in Swan Bay are fine with a relatively high organic content; in contrast, sediments on the adjacent coast of Port Phillip Bay are coarser sands with lower organic content (Shaw & Jenkins, 1992). The tidal range in the area is less than 1 m.

Within the Swan Bay region, three sites (Queenscliff, Tin Can and North Jetty) were within the bay itself, whilst St Leonards was on the adjacent coast of Port Phillip Bay (Figure 1). Seagrass was *H. tasmanica* with the exception of Queenscliff which was *Z. muelleri*. Unvegetated habitats at North Jetty and Tin Can were in the form of patches amongst seagrass and associated macroalgae.

Corner Inlet (Figure 1) is a large, enclosed bay with extensive shallow (<2 m) areas dissected by deeper channels. The tidal range is large (>2 m) and, therefore, currents in channels are relatively strong. There are large subtidal beds of eelgrass, *H. tasmanica*, and tapeweed, *Posidonia australis*, together with smaller areas of intertidal *Z. muelleri*. Channels are generally unvegetated.

Three sites were sampled in Corner Inlet (Figure 1). Seagrass at Bennison and Toora Channel was *H. tasmanica* whilst seagrass at Canoe Point was *Z. muelleri*. Both seagrass and unvegetated habitats were sampled at these sites.

Approximate depths of sites and habitats relative to mean low water spring (MLWS) are presented in Table 1. Three categories of habitat depth were defined arbitrarily: above MLWS to the level of MLWS (intertidal); depths down to 0.3 m below MLWS (shallow subtidal); and deeper than 0.3 m below MLWS (deeper subtidal) (Table 1). Samples in unvegetated habitat were within 20 m of adjacent seagrass habitat.

Table 1. Depths, relative to mean low water spring, of sites sampled

<table>
<thead>
<tr>
<th>Bay</th>
<th>Site</th>
<th>Port Phillip</th>
<th>Swan Bay</th>
<th>Corner Inlet</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>St Leonards</td>
<td>Queenscliff</td>
<td>Toora Ch.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>North Jetty</td>
<td>Tin Can</td>
<td>Canoe Pt.</td>
</tr>
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<td></td>
<td></td>
<td></td>
<td>Bennison</td>
</tr>
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<td>H</td>
<td>UV</td>
<td>Z</td>
<td>UV</td>
</tr>
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<td>−0.3</td>
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<tr>
<td>Category</td>
<td>DST</td>
<td>SST</td>
<td>INT</td>
<td>INT</td>
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</table>

H, *Heterozostera*; Z, *Zostera*; UV, unvegetated; INT, intertidal; SST, shallow subtidal; DST, deeper subtidal.

Field sampling

Sampling was conducted approximately monthly in Swan Bay and bimonthly in Corner Inlet for 1 year. Previous studies of fish in seagrass habitats in Victoria have concentrated on intertidal to shallow subtidal sites. For this study, it was considered desirable to sample deeper subtidal sites as well. For this purpose, a modified beach seine was constructed which could be deployed from a boat in water to a depth of approximately 2 m. The net was 10 m in length, had a 3 m drop, and a mesh of approximately 1 mm². This mesh size allowed the collection of juvenile fishes immediately after settlement in seagrass beds. The net was heavily weighted with both leads and chain to increase efficiency of sampling in seagrass habitat. Notwithstanding this, variation in capture efficiency with habitat complexity, and also fish behaviour (i.e. schooling vs. cryptic), would be expected. Trials with a net of similar construction over artificial seagrass showed a capture efficiency of between 20 and 83% depending on taxon (Sutherland, 1994). Capture efficiency of a goby species over unvegetated habitat was found to be approximately 70% (Henry & Jenkins, 1995), with some individuals probably burrowing to escape capture. Pilot sampling indicated that a 15-m haul gave an adequate sample size, but at the same time was small enough in sample area to allow specific microhabitats to be targeted. From the boat, ropes were hauled through detachable weights which stopped the net from rising from the bottom until completely retrieved. Three haphazardly placed replicate hauls of the seine were conducted in each habitat. Fishes were anaesthetized in benzocaine and preserved in 95% ethanol. Sampling was only conducted in daylight.

Laboratory methods

In the laboratory, fishes were identified to the lowest taxonomic level possible, and larger species were measured for standard length (SL, tip of snout to
Table 2. Mean number of individuals per haul and percentage contribution of each species of fish collected from seagrass and unvegetated habitats at sites in Swan Bay and at St Leonards

<table>
<thead>
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<th>Tin Can</th>
<th>North Jetty</th>
<th>St Leonards</th>
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</table>
posterior of caudal peduncle). Approximate biomass were estimated by first using previously determined regressions of standard length against length to caudal fork (LCF) to estimate the latter, and then using regressions formulated by G. Edgar (unpubl. data) to convert total length to wet weight. For small species with high abundances, such as gobiids and atherinids, the mean length of a small subsample was used to estimate the total biomass for the sample. In the case of pipefish, a single average length estimate was used to convert numbers to biomass for each species.

Data analysis

Cluster analysis was used to compare fish communities amongst site/habitat combinations. Similarities were estimated using the Bray-Curtis coefficient on log(x+1) transformed data. Agglomeration was achieved using group average clustering. Analysis was carried out using the Primer computer package from Plymouth Marine Laboratory, U.K.

Analysis of variance was conducted to assess spatial and temporal variations in the species richness, abundances and biomass of fishes in seagrass and unvegetated habitats. The samples were interspersed amongst habitat patches as much as the natural distribution of habitat would allow; however, ideal interspersion was not always possible, and results must be interpreted in this light. Sites in Corner Inlet and sites in Swan Bay were nested within the two bays, and the bimonthly samples from Corner Inlet were analysed with the Swan Bay samples from the corresponding months. Sites were treated as a random factor, whilst habitats and times were fixed. Box and residual plots were used to examine data for departures from assumptions of analysis of variance. It was necessary to log(x+1) transform the abundance and biomass data to stabilize variances; however, species number data did not require transformation. Variance components were approximated using the method described by Sokal and Rohlf (1981). Planned comparisons were used to compare seagrass and unvegetated habitats in individual combinations of the date and site (Sokal & Rohlf, 1981). As the number of comparisons exceeded the degrees of freedom for the appropriate (third-order) interaction, it was necessary to adjust the significance level (Sokal & Rohlf, 1981). This was achieved by dividing $\alpha=0.05$ by the number of comparisons (15) in excess of the degrees of freedom for the interaction, resulting in a critical value of 0.0033. Analyses were conducted for total fishes and for 'benthic' fishes, total fishes less the pelagics schooling atherinids and clupeoids. Analyses were carried out on the 'Systat' computer program (Wilkinson et al., 1992).

**Results**

**Fish community composition**

**Swan Bay.** The community composition in seagrass habitat at North Jetty and Tin Can was dominated by the spotted pipefish, *Stigmatopora argus* (Table 2). The pattern of dominance was very similar at both sites, with other moderately abundant species including silver fish, *Leptatherina presbytoides*, Port Phillip pipefish, *Vanacampus phillipi*, common weedfish, *Heteroclinus perspicillatus*, sixspine leatherjacket, *Meuschenia*
freycineti, bridled leatherjacket, Acanthaluteres spilomelanurus, little rock whiting, Neodax balteatus, and blue rock whiting, Haletta semifasciata. Community composition was also similar between the unvegetated habitats at these sites (Table 2), with dominants including spotted pipefish (probably associated with small seagrass isolates within the area), King George whiting, Sillaginodes punctata, and girdled goby, Nesogobius sp. 1. These unvegetated habitats differed in the high abundances of L. presbyteroides at North Jetty, and Castelnau’s goby, Nesogobius pulchellus, at Tin Can.

Dominance patterns in Heterozostera habitat at St Leonards (Table 2) were markedly different to the same habitat in Swan Bay. Although occurring in much lower abundances than the spotted pipefish in Swan Bay, the dominant species was the wide-bodied pipefish, Stigmatopora nigra. Other relatively abundant species included clingfish, Parviceps sp., and blue sprat, Spratelloides robustus. Like Swan Bay, H. perspicillatus was a relatively abundant species, but in contrast to Swan Bay, its congener, the Adelaide weedfish, Heteroclinus adelaidae, was almost as abundant. Dominant species on unvegetated habitat at St Leonards were juveniles of the yellow-eye mullet, A. forsteri, the greenback flounder, Rhombosolea tapirina, and the long-snouted flounder, Ammotretis rostratus.

Abundance patterns at Queenscliff were also distinct from other sites (Table 2). Hardheads (atherinids) were dominant in seagrass; however, unlike North Jetty and Tin Can, the small-mouthed hardhead, Atherinosoma microstoma, occurred in greater numbers than L. presbyteroides. The dominant pipefish in this habitat was the hairy pipefish, Urocampus carinirostris. Also abundant, relative to other habitats, was the bridled goby, Arenigobius frenatus. Unvegetated habitat at Queenscliff was dominated by the long-finned goby, Favonigobius lateralis. Also common were juvenile S. punctata and Rhombosolea tapirina.

Corner Inlet. Favonigobius lateralis was a dominant species on unvegetated habitats in Corner Inlet (Table 3), co-dominating with A. microstoma at Bennison and L. presbyteroides at Toora Channel. Hardheads were relatively uncommon at Canoe Point, where important species included S. nigra, A. frenatus, blue-spotted gobies, Pseudogobius olorum, and juvenile trevally, Pseudocaranx dentex.

Abundant species in the intertidal Zostera site at Canoe Point included P. olorum, U. carinirostris, S. nigra and A. frenatus (Table 3). Also important were small juveniles of the luderick, Girella tricuspidata, in December and the sandy sprat, Hyperlophus viitatus, in February. The shallow, subtidal Heterozostera habitat at Bennison also had significant abundances of P. olorum and U. carinirostris; however, in this case, the soldierfish, Gymnapistes marmoratus, A. spilomelanurus and F. lateralis were also important (Table 3). The deeper Heterozostera site at Toora Channel was dominated by A. spilomelanurus, S. nigra and S. argus, Wood’s siphonfish, Siphamia cephalotes, and N. balteatus.

Cluster analysis

Cluster analyses were initially conducted at the level of individual replicates at sites analysed separately for each month. Although there was some variation between months, groupings tended to be related more to habitats than locations, and, in general, Zostera sites and most unvegetated sites grouped together whilst Heterozostera sites and the deeper unvegetated sites of Swan Bay grouped together. These results can be summarized by cluster analysis of the data pooled across replicates and times (Figure 2).

Variation in species richness, abundance and biomass in Swan Bay and Corner Inlet

Between Swan Bay and Corner Inlet, there was no significant difference in species numbers (Table 4). All interactions involving sites were significant, indicating that species number varied depending on the
particular combination of habitat, date and site examined (Table 4). The largest variance component, however, was for the habitat main effect (Figure 3), which was related to the fact that in the majority of habitat comparisons for each combination of site and date, species number in seagrass was higher. Moreover, all significant differences occurred for comparisons where species number was higher in seagrass (Figure 3). The date main effect made a relatively important contribution to the variance, probably related to an increase in the number of species over the warmer months of the year (Figure 3).

A similar pattern of significant interactions associated with sites occurred for total abundances (Table 4). The most important variance component was for the interaction between site and habitat (Table 4). This resulted from the fact that abundances in seagrass were always higher, and often significantly higher, at Tin Can, North Jetty, Toora Channel and Canoe Point; but at Queenscliff and Bennison, abundances were generally not significantly different, and on the two occasions when significant differences did occur at Bennison, abundances were higher on unvegetated habitat (Figure 4). The main factors causing the observed difference when comparing Bennison and Queenscliff to other sites was a decreased abundance of fishes in seagrass, and relatively high abundances of $F. \text{lateralis}$ on unvegetated areas (Table 3). Results for total abundance, less atherinids and clupeoids, were essentially the same as total abundance, and therefore will not be reported here.

Variance in biomass showed a similar trend to abundances; the variance explained by the interaction between site and habitat was again most important, but the proportion was reduced relative to abundance, whilst the contribution of the interaction between site, habitat and date increased (Table 4). In general, there were fewer significant differences between habitats for individual site/date combinations, and in no case was biomass significantly higher in unvegetated habitat (Figure 5). Whilst seagrass habitat had significantly higher abundances on all dates at Canoe Point (Figure 4), differences in biomass were only significant in December (Figure 5), indicating that although fish were more numerous in seagrass at this site, they were smaller. Large biomass values for seagrass at Toora Channel in December and April (Figure 5) resulted from the collection of adult rock flathead, $P. \text{laevis}$.  

### Commercial species

This section concentrates primarily on results from the Swan Bay region of Port Phillip Bay, where more frequent sampling allowed analysis of seasonal trends in abundance and recruitment.

**King George whiting, S. punctata.** *Sillaginodes punctata* in Swan Bay showed a peak in unvegetated habitats over the late spring period at Tin Can and North Jetty, and a peak in February/March at Queenscliff (Figure 6). Comparatively small numbers were collected from St Leonards in the late spring, predominantly in seagrass (Table 2). The late-spring peaks were associated with the influx of post-larvae approximately 20 mm SL. Juveniles subsequently grew to a size of

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### Table 4. Analysis of variance of species number, total abundance and total biomass of fishes on seagrass and unvegetated habitats at sites in Swan Bay and Corner Inlet sampled bimonthly over 12 months

<table>
<thead>
<tr>
<th>Source</th>
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<th>Species number</th>
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<th>Total biomass</th>
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<td></td>
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<td>F</td>
<td>%</td>
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<td>10.63**</td>
<td>12.8</td>
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<tr>
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<td>20.65*</td>
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<td>1.21NS</td>
<td>0.7</td>
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<tr>
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<td>8.4</td>
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<td>9.69</td>
<td>4.34**</td>
<td>6.1</td>
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<tr>
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%, percentage of variance explained; NS, $P > 0.05$; *$P < 0.05$; **$P < 0.001$.  

Fish assemblages in Victorian seagrass beds 577
Figure 3. Mean species richness of fishes in bimonthly samples collected from seagrass and unvegetated habitats at sites in Swan Bay and Corner Inlet. Solid bars, seagrass; open bars, unvegetated. *Significant differences between habitats based on planned comparisons. Error bars are standard error.

Figure 4. Mean abundance of fishes in bimonthly samples collected from seagrass and unvegetated habitats at sites in Swan Bay and Corner Inlet. Solid bars, seagrass; open bars, unvegetated. *Significant differences between habitats based on planned comparisons. Error bars are standard error.
80–100 mm over the first year (Figure 6). Few *S. punctata* were collected in Corner Inlet (Table 3).

**Blue rock whiting**, *H. semifasciata*. *Halettasemifasciata* were abundant in the *Heterozostera* habitats of Swan Bay (Figure 7), but were absent from the Queenscliff and St Leonards sites (Table 2). Recruitment of this species occurred from January to March at a size of 10–15 mm, which is particularly reflected by the pattern of abundance at Tin Can (Figure 7). Like *S. punctata*, size at the end of the first year was approximately 80–100 mm. Larger individuals up to 280 mm were collected occasionally.

**Sixspine leatherjacket**, *M. freycineti*. *Meuschenia freycineti* showed a similar pattern to *H. semifasciata*, with a specific preference for *Heterozostera* habitat of Swan Bay (Figure 8). In this case, a few individuals were also collected at St Leonards (Table 2). Settlement occurred at a size of 10–15 mm over an extended period through spring and summer, that is reflected in the time course of abundance in Swan Bay (Figure 8). Larger individuals up to 250 mm were collected. The low numbers of rough leatherjacket, *Scobinichthys granulatus*, in Corner Inlet showed a similar preference for *Heterozostera* (Table 3).

**Yellow-eye mullet**, *A. forsteri*. *Aldrichetta forsteri* was collected in significant numbers only at the Queenscliff and St Leonards sites (Figure 9). At Queenscliff, the relative abundances in the two habitats varied between sampling dates, whilst at St Leonards, the majority of individuals were collected on sand (Figure 9). The smallest individuals, approximately 30–35 mm in length, were collected from February to May (Figure 9). The largest individuals collected were approximately 110 mm SL.

**Greenback flounder**, *R. tapirina*. Highest numbers of *R. tapirina* were collected in the unvegetated habitats at Queenscliff and St Leonards (Figure 10). Very few individuals were collected at Tin Can or North Jetty (Table 2). Settlement of this species occurred in late winter/early spring at a size of 5–10 mm SL, and individuals ranged up to approximately 90 mm SL (Figure 10). Juveniles of *A. rostratus*, although occurring in lower numbers (Table 2), showed a similar spatial and temporal pattern of settlement.

### Other species

Adult *Platycephalus laevigatus* were only collected from *Heterozostera* habitats of Swan Bay and Corner Inlet (Tables 2 and 3); however, small juveniles in Corner

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**Figure 5.** Mean biomass of fishes in bimonthly samples collected from seagrass and unvegetated habitats at sites in Swan Bay and Corner Inlet. Solid bars, seagrass; open bars, unvegetated. *Significant differences between habitats based on planned comparisons. Errors bars are standard errors.*
Figure 6. Time course of (a) abundance and (b) length-frequency distributions for King George whiting, *Sillaginodes punctata*, in the Swan Bay region of Port Phillip Bay, sampled monthly over 1 year. Solid symbols and bars, seagrass; open symbols and bars, unvegetated. Error bars are standard error.
Figure 7. Time course of (a) abundance and (b) length-frequency distributions for blue rock whiting, *Haletta semifasciata*, in the Swan Bay region of Port Phillip Bay, sampled monthly over 1 year. Solid symbols and bars, seagrass; open symbols and bars, unvegetated. Error bars are standard error.
Figure 8. Time course (a) abundance and (b) length-frequency distributions for the sixspine leatherjacket, *Meuschenia freycineti*, in the Swan Bay region of Port Phillip Bay, sampled monthly over 1 year. Solid symbols and bars, seagrass; open symbols and bars, unvegetated. Error bars are standard error.
Figure 9. Time course (a) abundance and (b) length-frequency distributions for the yellow-eye mullet, *Aldrichetta forsteri*, in the Swan Bay region of Port Phillip Bay, sampled monthly over 1 year. Solid symbols and bars, seagrass; open symbols and bars, unvegetated. Error bars are standard error.
Figure 10. Time course of (a) abundance and (b) length-frequency distributions for the greenback flounder, *Rhombosolea tapirina*, in the Swan Bay region of Port Phillip Bay, sampled monthly over 1 year. Solid symbols and bars, seagrass; open symbols and bars, unvegetated. Error bars are standard error.
Inlet were mainly found on unvegetated habitat (Table 3). Small (15–20 mm) juvenile *P. dentex* were collected at the Canoe Point and Toora Channel sites in December and February; no clear trend in habitat utilization was apparent (Table 3). Large numbers of newly settled (10–15 mm) *G. tricuspidata* were collected at the Canoe Point and Toora Channels sites in December (Table 3).

**Discussion**

The fish assemblages associated with seagrass habitats in Port Phillip Bay and Corner Inlet were similar to those described for many seagrass areas (Bell & Pollard, 1989), with a dominance of small cryptic species, such as syngnathids and gobiards, together with juveniles of larger species. This observation may be biased, however, because the small size of the seine net would reduce its effectiveness for sampling larger, mobile species. Relatively large (>200 mm LCF) individuals of species such as *M. freycineti*, *H. semifasciata* and *P. laevis* were collected frequently, suggesting that the net was capable of sampling larger specimens, albeit with reduced efficiency. These species can apparently complete their entire life cycle in seagrass beds, although the authors have observed larger individuals (but not recruits) of all three species on deeper reef/algal habitats (G. P. Jenkins & M. J. Wheatley, unpubl. data).

Fish communities associated with subtidal *Heterozostera* were consistent amongst sites in Swan Bay and Corner Inlet, as were communities associated with intertidal *Zostera*. However, there was a major difference in species composition and abundance between the two habitats. It is not possible at this stage to distinguish between the effects of habitat structure and water depth in causing these differences. The two species of seagrass in question are very similar morphologically; however, deeper subtidal beds were generally more complex (higher density, longer stems) than shallow subtidal or unvegetated beds. Therefore, deeper seagrass beds may have allowed greater protection from predators or increased food availability relative to unvegetated areas compared with shallow seagrass beds (Bell & Pollard, 1989). Differences with depth may also relate to the depth distribution of pre-settlement larvae (Bell & Westoby, 1986). Manipulative experiments utilizing artificial seagrass beds would be required to assess the importance of these factors (Bell & Pollard, 1989).

The pattern of dominance differed from previous Victorian studies using fine-mesh beach seines in seagrass areas of Swan Bay (Jessop, 1988), Westernport (Robertson, 1980) and Gippsland Lakes (Ramm, 1986). Dominant species in these studies included *A. forsteri*, *A. microstoma*, *G. marmoratus*, smooth toadfish, *Tetractenos glaber*, *R. tapirina* and gobies, *Favonigobius* sp. The species composition is therefore most similar to the present intertidal and shallow subtidal sites, and include species found on unvegetated and/or seagrass habitat. The major factor influencing the species composition in the present study compared to previous studies was the fact that, in the present study, the seine was deployed from a boat, whilst in previous studies, the seine was hauled onto the shore. Many of the dominant species in previous studies tend to occur in shallow water following the movement of the tide, and therefore would be highly susceptible to seines hauled onto shore. For example, Jessop (1988) sampled a number of intertidal sites in Swan Bay; the dominants, including *A. microstoma*, *F. lateralis* and *R. tapirina*, were very similar to those on seagrass and unvegetated habitat at the shallow Queenscliff site in the present study, whilst dominants in subtidal sites of Swan Bay in the present study, such as *S. argus* and *A. spilomelanurus*, were recorded rarely. In a study that was conducted in Westernport concurrently with the present study, and where the seine was also deployed from a boat, dominants included species that also dominated at many of the present study’s subtidal sites, such as *Stigmatopora* spp., *V. phillipi*, *H. perspicillatus*, *M. freycineti* and *H. semifasciata* (Edgar & Shaw, 1995a). These results suggest that the composition of fishes collected from seagrasses of Victoria will probably show much greater variation with the depth of sampling compared with location or time of sampling.

Important differences also occurred between fish assemblages associated with each habitat. The species dominating on unvegetated habitat were often quite distinct from those on the adjacent vegetated habitat. Ferrell and Bell (1991) found that unvegetated areas adjacent to seagrass constituted a specific habitat for a number of species that was distinct from unvegetated areas some distance (>100 m) from seagrass. Also, the number of individuals on sand adjacent to seagrass was significantly higher than sand distant from seagrass. The presence of seagrass may lead to organic enrichment of unvegetated sediments nearby, enhancing food production for fishes (Shaw & Jenkins, 1992). Furthermore, some species may use unvegetated areas as long as refuge is available nearby for refuge from predators (Robertson, 1980; Summerson & Peterson, 1984). Therefore, comparing seagrass and adjacent unvegetated habitat may not be relevant for predicting changes in fish assemblages where the majority of seagrass is lost, because both organic
enrichment of sediments, and availability of refuge, would be reduced substantially.

It is difficult to generalize on the question of whether seagrass has a higher diversity and abundance of fishes in southern Victoria. The hypothesis was supported to some extent in terms of species richness. Overall, there were significantly more species in seagrass, but on many sampling occasions, there was no significant difference between the two habitats. Support for the hypothesis that abundances of fish are higher in seagrass depended on the site examined. Whilst most sites conformed with the prediction, particularly the deeper, subtidal Heterozostera sites, the shallower Queenscliff and Bennison sites did not. These sites were characterized by having relatively low abundances of fish in seagrass and high abundances of F. lateralis on the shallow unvegetated habitat. These results suggest that loss of both intertidal and subtidal seagrasses would result in a significant decline in species diversity; however, loss of deeper subtidal Heterozostera would have greater consequences for the abundance of fishes compared with loss of intertidal or shallow subtidal seagrasses.

Conclusions from the analysis of variance based on biomass did not differ greatly from those based on abundances with the exception of the Canoe Point site. Abundances at this site were dominated by the relatively small P. olorum. The effect of particularly large animals on biomass relative to abundance estimates was clear at Toora Channel, where the collection of a small number of adult P. laevigatus caused major biomass differences on certain dates.

Commercial species

The larval stage of a number of commercial species, including blue rock whiting, sixspine leatherjacket and rough leatherjacket, settled directly to deeper, subtidal Heterozostera beds. Juveniles of these species are cryptically coloured for life in seagrass, and individuals may remain in seagrass habitat throughout life. In contrast to the species above, rock flathead were generally represented by large individuals (Jenkins et al., 1993b). The diet of these species is dominated by seagrass-associated biota (Klumpp & Nichols, 1983; Burchmore et al., 1984; Edgar & Shaw, 1995b). Although a causal linkage between seagrass decline and fish decline in Westernport has not been demonstrated, and a third agent may have been responsible for both, the fact that the species mentioned above have shown the most dramatic decline would be consistent with the effects of habitat loss.

Other commercial species, however, tended to occur at intertidal and shallow subtidal sites, and showed no strong preference for seagrass habitat in the juvenile stage. Juvenile yellow-eye mullet feed primarily on zooplankton (Robertson, 1980; Edgar & Shaw, 1995b). This result may explain why yellow-eye mullet was one of the few species for which catches increased over the period of seagrass loss in Westernport. The greenback and long-snouted flounder showed a strong preference for shallow unvegetated habitats. At least in the case of flounder, there is evidence that juveniles may benefit indirectly from seagrass through organic enrichment of sediments and corresponding elevation of food production (Shaw & Jenkins, 1992; Jenkins et al., 1993a).

Settlement of King George whiting over the September/October period at a size of approximately 20 mm is consistent with previously recorded results for Westernport (Robertson, 1977), Port Phillip Bay (Jessop, 1988) and Gippsland Lakes (Ramm, 1986). The unvegetated patches amongst Heterozostera habitat in Swan Bay were the major habitat of newly settled juveniles of King George whiting. This contrasts with the results for Port Phillip Bay (Jenkins et al., 1993b), Westernport (Robertson, 1977) and Barker Inlet (Connolly, 1994a), where recruits were only collected in seagrass. The result that habitat preference appears to vary with locality may relate to the relative abundance and vulnerability of prey items in each habitat, because the major link between this species and seagrass habitat is probably food supply rather than protection from predators (Connolly, 1994b). The decline in King George whiting catches in Westernport may be related to seagrass loss and subsequent decline in food availability or other factors; however, this fishery may also be strongly influenced by variability in recruitment of larvae from Bass Strait (Jenkins & Black, 1994; Jenkins & May, 1994). Appearance of older juveniles in February/March on intertidal unvegetated habitat is consistent with the ontogenetic movement of this species to intertidal unvegetated habitat with growth in Westernport (Robertson, 1977).

Canoe Point was quite different from the other sites in terms of species composition and abundance, apparently related to its greater estuarine influence. In terms of commercial species, newly settled juveniles of luderick were only collected at this site, with an apparent strong preference for seagrass. Zostera, in estuaries of New South Wales, has previously been found to be an important juvenile habitat for this species (Middleton et al., 1984; Ferrell & Bell, 1991). Ramm (1986) found that juveniles of luderick were most common in seagrass with moderate estuarine influence.
In summary, although seagrass was an important habitat for some commercial species, other commercial species were found to utilize unvegetated habitats, and the juveniles of *S. punctata* may show flexible use of habitat, probably related to prey availability. The utilization of unvegetated habitats by juvenile commercial fish may be greater in areas where sediments are enriched with seagrass detritus or where seagrass refuge is nearby.

**Conclusions**

Fish assemblages in intertidal *Zostera* habitats were more similar to shallow unvegetated habitats than to subtidal *Heterozostera* habitat. Species richness was higher in seagrass compared with unvegetated habitats. Abundance and biomass of fishes were generally significantly higher in seagrass compared with unvegetated habitat in deeper subtidal areas. However, abundance and biomass were not significantly higher in seagrass relative to unvegetated habitat in some intertidal and shallow subtidal areas. As a consequence, any loss of seagrass will probably result in a decrease in species richness, and will also often result in a significant decrease in abundance and biomass of fishes, particularly in deeper subtidal areas. Seagrass is an important habitat for juveniles of some commercial species, but other commercial species utilize alternative habitats, particularly intertidal unvegetated habitats.

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**References**


Ferrell, D. J. & Bell, J. D. 1991 Differences among assemblages of fishes associated with *Zostera capricorni* and bare sand over a large spatial scale. *Marine Ecology Progress Series* 72, 15–24.


Sutherland, C. R. 1994 *What is the Role of Habitat Complexity in Determining the Distribution of Young Fishes?* B.Sc. (Hons.) Thesis, University of Melbourne, Victoria.