Spatial arrangement of estuarine and coastal habitats and the implications for fisheries production and diversity

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Objectives

1. Develop appropriate methods for determining relationships between the spatial arrangement of estuarine habitats (habitat mosaics) and their biota including fish, crustaceans and molluscs and more sedentary epibenthos (e.g. comparing the fauna of isolated seagrass with that of seagrass close to mangroves);

2. Document patterns of abundance and diversity of fish and decapods in different habitat mosaics from degraded and relatively undisturbed areas of estuarine embayments, in two latitudinally separated areas (Moreton Bay and Hervey Bay) and taking into account position within an embayment (e.g. western side versus eastern side);

3. Increase our understanding of fisheries-habitat links using a combination of standardised survey methods in Queensland and make comparisons with the results obtained in Victoria using the same methods and gear types (Jenkins, 2001/036);

4. Identify the relative importance of different nearshore habitats for key fish and decapod species from recruitment to older life-history stages and compare these results with those obtained for similar species and trophic groups in Victoria (Jenkins, 2001/036);

Non Technical Summary:

Mangroves, seagrasses, saltmarsh and other complex habitats are recognised as being critically important as nurseries for commercially and recreationally important fish, crabs, prawns, squid and octopus (i.e. nekton). These habitats provide safe places for animals to hide from predators and to escape harsh environmental conditions, but also provide rich feeding grounds. Many of the juveniles of species that support recreational and commercial (trawl, net and pot) fisheries are found in these shallow water habitats. Although there is good understanding of the general importance of the physical characteristics of these different habitats to nekton (e.g. thick, lush seagrass supports more fish than sparse patchy beds), it is not clear why some patches of habitat are "better" than others in supporting these juvenile populations.

Much of the past research examining the benefits of complex habitats such as mangroves and seagrass to nekton has not taken into account that mobile animals such as fish and prawns move between different habitats. For example, on the rising tide, many fish move into mangrove forests to feed and escape larger predators. As the tide falls though, they are forced to retreat to the subtidal zone. If they are forced to move from the mangroves to an area that offers no protection, they may be vulnerable to large predators waiting for them. In contrast, fish leaving the mangroves and moving into a dense subtidal seagrass bed would have protection from these predators. It is not just the mangroves that may be important in providing a nursery for nekton, it is likely to be the combination of habitats – the mosaic – that may be critical to sustaining fisheries.

The focus of this research was in two parts. First, detailed surveys of the numbers and types of fish and crustaceans (prawns, crabs) using different habitats (mangroves, seagrass and mudflats) were done in Moreton Bay, southeast Queensland. Surveys were completed seasonally over 12 months at sites in the western and eastern regions of Moreton Bay. The western region is influenced by the intense urban and industry development of Brisbane, plus the inflows from several major river systems. The eastern region is less influenced by human development. The methods that were used were determined in conjunction with a study being done at the same time in Victoria and also funded by the FRDC. Use of the same methods in the same manner, in the same selection of habitats, provided a unique opportunity to compared the utilisation of critical estuarine nursery habitats by fisheries species in temperate (Victoria) and subtropical (SE Queensland) systems. The second component of this project involved an investigation of whether the spatial arrangement of different habitats affected their value to nekton in Moreton Bay. We examined issues such as whether the proximity between seagrass and mangroves affected the value of the seagrass and whether the type of habitat linking mangroves to subtidal areas changed the value of the mangroves to nekton.

Moreton Bay supports a rich and abundant assemblage of nekton using the shallow water habitats. A total of 65 species (20,990 individuals) of fish, 26 species (29,385 individuals) of decapod crustaceans (e.g. prawns, crabs and shrimp) and 5 species (1,859 individuals) of cephalopods (squid, cuttlefish) was sampled from mangroves, seagrass and mudflats in western Moreton Bay over the course of 12 months, from spring (September-October) 2002 through to winter (June-July),

2003. Similar numbers of species and individuals were sampled from eastern Moreton Bay over the same. More species of fish and decapods were caught in the mangroves in western than eastern Moreton Bay, even though larger numbers of individuals were caught on the eastern side. This pattern is consistent with the overall patterns of the distribution of different groups of marine animals across the embayment, based on the degree of marine influence which varies from east to west across the bay. Interestingly though, the opposite pattern was evident for intertidal seagrass, with more species caught in the eastern than western Bay, although many of these were only represented by a single individual. More individuals were caught in the eastern Bay, consistent with the pattern for the mangroves.

The number of species and abundance of fish in subtropical Moreton Bay was considerably greater than recorded for Victorian estuarine habitats, using precisely the same sampling gear over much the same time period. Our detailed studies in subtropical Moreton Bay confirm the presence of strong gradients in species richness and abundance of fish and decapods from tropical to temperate regions, but it is not possible to identify specific explanations for those patterns due to numerous confounding factors across the different studies (including levels of surrounding development and impact on the mangroves, and species complexity of the mangroves).

Different communities of nekton used the mangroves and nearby intertidal seagrass, with little overlap between the two habitats, on either side of Moreton Bay, consistent with previous studies. Our results suggest that as the fish and prawns move into the intertidal zone during high tide, some species access the seagrass beds, but other species move further upshore into the mangroves, either for shelter or to feed. Even though the numbers and types of fish, crabs and prawns using the seagrass beds and mangrove forests were different, there was overlap in the species that used the mangroves and those found in the mudflat. A previous study done in the region over 10 years ago, had found 27 species of fish exclusively in the mangroves but only 4 species of fish that were exclusive to seagrass. In marked contrast, we found 20 species exclusive to the mangroves, 14 species exclusive to the seagrass and 3 species exclusive to the mudflats. Thus, we found fewer species using only the mangroves and many more species using only the seagrass. The total number of species of fish was similar in both studies (56 in the earlier study, 65 in ours). It is possible that the considerable changes in the environment within Moreton Bay over the 13 years between these two studies have modified the nature of the communities using the bay.

We developed the first accurate maps showing the position of each of the major estuarine habitats across the gradient from shallow subtidal to terrestrial zones, covering a large (100 x 35 km) portion of Moreton Bay. Our approach relied only on three ingredients: aerial imagery, knowledge of the study environment and general competency with modern GIS software. Thus, this approach could be readily applied by managers and researchers to gain a better understanding of the landscape composition within other important coastal and estuarine regions.

The distance between patches of seagrass and nearby mangrove forest had a large effect on the catches of juvenile prawns in these seagrass beds. This distance is one measure of the connectivity of the habitats. Importantly, we found that the distance between seagrass and mangroves was a more important factor than the density of the seagrass in determining catches of greasyback, *Metapenaeus* *bennettae*, and eastern king, *Penaeus plebejus*, prawns. The estimated potential contribution of dense seagrass close to mangroves to postlarval and juvenile populations of *M. bennettae* and *P. plebejus* from Moreton Bay was at least 8-12 times greater than the other habitats. Likewise, sparse seagrass, previously considered to be a lesser quality habitat, could potentially contribute several times greater numbers of juvenile prawns to commercial populations when the beds were close to mangroves, than when dense or sparse seagrass was far from the forests.

The connectivity between seagrasses and nearby mangroves also had a marked effect on the numbers and types of fish and shrimp using the seagrass. Overwhelmingly, larger numbers of individuals and species were found in seagrass beds that were close to the mangroves than those further away. For many species of fish, this pattern was consistent in dense and sparse seagrass beds, indicating that for those species the influence of potential connectivity between seagrass and mangroves had a greater influence than the density of the seagrass. For other species, the effects were more complicated, but clearly, the way in which different habitats are used by nekton in these subtropical estuarine systems is influenced the nature of the surrounding habitats. The presence or absence of mangroves in the intertidal zone may be a critical feature in maintaining patterns of increased biodiversity and biomass of fish assemblages in other nearshore habitats (e.g. subtidal seagrass beds). These critical linkages between mangroves and other components of estuarine and coastal systems suggest that the loss or degradation of mangroves is therefore likely to have serious implications for the overall function and value of nearby habitats, if there is a reduction in habitat connectivity. That is, losing the mangroves will have greater implications for fisheries within an estuary than previously considered.

The nature of the corridor connecting subtidal seagrasses with intertidal mangroves also had marked effects on the numbers and types of fish and invertebrates (prawns, shrimp and crabs) that used the mangroves. Nekton moving into the mangroves on the rising tide have to cross different types of habitat as they leave the subtidal zone. During the day, almost twice as many individuals moved into the mangroves when the corridor had dense seagrass or was bare compared with when it was a patchy seagrass habitat. At night the pattern was similar, except the number of individuals moving across the unvegetated corridor was intermediate between the dense seagrass and patchy seagrass corridors. Some species were clearly more strongly associated with the unvegetated corridor (e.g the estuary perchlet, *Ambassis marianus*) whereas other species were associated with vegetated corridors (e.g. yellowfin bream, *Acanthopagrus australis*, gobies and the estuarine shrimp, *Palaemon debilis*). These results have important implications for understanding the linkages between nekton and the critical estuarine habitats that support them. Nekton do not utilise habitats such as mangroves independently of the surrounding landscape and, as such, conservation and management of nursery areas will not be effective unless this is taken into account.

All this work had been focussed in the northern sections of Moreton Bay. These areas are characterised by large open, expanses of coastline within the estuarine embayment. Our results indicate that characteristics such as the proximity between different habitats and the nature of the corridors connecting different habitats are extremely important in determining the numbers and types of species of nekton using nearshore nursery areas. But how general are these relationships and patterns? Are the effects of attributes such as connectivity among patches and the structure and composition of corridors between patch types on the use of estuarine mosaics by nekton consistent in other systems?

The final component of this project was to determine the extent to which these relationships between the abundance and composition of the nekton assemblage and the spatial arrangement of estuarine habitats occurred in a different system. Southern Moreton Bay was selected as the region for this independent validation of the application of landscape techniques because the configuration of habitats is markedly different from eastern and western Moreton Bay where all previous work has been done. In particular, the southern region of Moreton Bay is dominated by small, mangrove-covered islands separated by deep channels, as opposed to the extensive, elongated intertidal flats present in the western and eastern regions. Detailed sampling of 50 sites across southern Moreton Bay indicated that there were large differences in the average abundance of fish utilising the five major types of estuarine landscapes. The increase in abundance between the landscapes supporting the largest and smallest numbers of fish were commonly greater than 2-3 fold in vegetated and unvegetated habitats. These very large differences in abundance in different mosaics are consistent with the studies done in other parts of Moreton Bay.

OUTCOMES ACHIEVED

The outputs of this project are most likely to lead to improved outcomes in natural resource management benefiting natural resource managers, recreational and commercial harvesters and the general public.

Outputs of this project will be provided to the Queensland State Government (EPA) at a time when there is a statutory review of the Moreton Bay Zoning Plan occurring. EPA has requested information on the patterns of distribution and abundance of nekton (fish, crabs and prawns) in different regions of the Bay as part of their planning. Discussion of the results of this work with members of the Burnett Mary Regional Group for Natural Resource Management has led to development of plans for implementation of the methods and approaches for the assessment of fisheries resources in the Hervey Bay region. Similar discussions are under way with other local catchment authorities interested in the application of the approaches developed in this study.

The outputs of this project have considerable public good benefit in that they directly provide information on the ecological functioning of different estuarine nursery habitats and the use of these habitats by commercially and recreationally important finfish, crabs and prawns.

KEYWORDS: **nursery areas, estuarine habitats, habitat quality, spatial arrangement, proximity, SE Queensland, Moreton Bay**

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Background

The juveniles of many species of commercially and recreationally important finfish, crabs, prawns, squid and octopus are often closely associated with nearshore, shallow water habitats within estuaries (e.g. Bell and Pollard, 1989; Sogard et al., 1989; Bell et al., 1992). Extensive research has demonstrated the close relationship between the distribution of many of these epibenthic animals and the distribution of specific types of habitat, such as mangroves and seagrasses, within estuaries and along the coastline. Probably the clearest demonstrations of strong associations between epibenthos and coastal habitats are those for some species of juvenile penaeid prawns, with tiger prawns found mainly on seagrass and banana prawns along mangrove lined mudbanks (Staples et al. 1985, Loneragan et al. 1998, Vance et al. 1998). Finfish, crabs and prawns use these habitats as feeding and nursery areas and these "fisheries" habitats are considered critical to sustaining the species that support recreational and commercial (trawl, net and crab) fisheries (e.g. QFMA, 1996 a, b).

In the past, fisheries research has focussed on the importance of individual habitat types with the aim of comparing, for example, mangroves versus seagrasses (e.g. Robertson and Duke, 1987; Laegdsgaard and Johnson, 1995), or vegetated versus unvegetated areas (e.g. Ferrell and Bell, 1991; Edgar et al., 1994; Jenkins and Wheatley, 1998) in terms of their relative importance to finfish and crustaceans. Past research has also led to significant advances in our understanding of the importance of specific features of these habitats, such as the density and length of seagrass shoots, or the complexity of mangrove root systems, on finfish and crustacean abundance and diversity (e.g. Bell and Westoby, 1986a; Mullin, 1995; Halliday and Young, 1996).

This approach to fisheries habitat research has treated different types of habitat as isolated and separate patches, rather than as a 'mosaic' of inter-linked habitats (Skilleter, 1998), all connected by the water either continually subtidally or during tidal inundation. Many of the species using these estuarine mosaics are highly mobile and could easily move between multiple habitat types regularly over a tidal cycle or during the course of their life cycle (Irlandi and Crawford, 1997; Kneib, 1997; Cappo et al., 1998). Research has shown that there may be significant variation in the way shallow water habitats are utilised during, for example, a diel cycle (e.g. Bell and Pollard, 1989; Sogard et al., 1989; Vance and Staples, 1992). Moving between different habitat types on a daily basis or during the course of their life cycle provides an opportunity for an animal to utilise different resources, such as food or shelter, found in different parts of the mosaic (e.g. Hansson et al., 1995). Studies which only focus on a single type of habitat from within these mosaics, or those that simply compare one habitat with another, will not provide a good understanding of the reasons behind variation in the abundance of finfish and crustaceans from place to place, or from time to time (e.g. inter-annual variation) or how modification (natural or anthropogenic) of one habitat type affects overall fisheries production or diversity in an area.

It is clear that there needs to be a major shift in focus in fisheries research from a continuation of this past concentration on individual habitat types to an approach where nearshore intertidal and subtidal areas are examined in a manner which reflects the way in which they are used by fisheries species: as a mosaic of interconnected patches made up of many different types of habitat. At present, there is almost no information about the importance of the particular arrangement of the different patches of habitat within different mosaics on the abundance and diversity of finfish and crustacean communities. For example, it is unclear whether an area of seagrass, adjacent to intertidal mangrove, provides habitat of better quality for juvenile fish than a similar area of seagrass adjacent to intertidal mud (but see Irlandi and Crawford, 1997 for an overseas example for saltmarshes). There are also unresolved questions about how loss and change in these habitats influences fisheries productivity. Does the loss of an area of intertidal mangrove also affect the value to fish and crustaceans of nearby patches of previously connected subtidal seagrass?

Additionally, there is little information about the specific mechanisms which underlie the close relationships between the juveniles and adults of fisheries species and these estuarine habitats. It has long been known that two patches of seemingly similar habitat (for example, patches of mangrove forest) frequently support different communities of fish and crustaceans despite their close proximity to each other (e.g. Bell and Westoby, 1986b; Laegdsgaard and Johnson, 1995). Variation in recruitment associated with local hydrodynamics (e.g. McNeill et al, 1992; and see Cappo et al., 1998) is clearly important at some spatial scales, but is unlikely to account for the fact that often, patches of habitat relatively close together support quite different communities of finfish and crustaceans. Variation in the availability of food and shelter are two alternative explanations which need to be considered, incorporating information about the arrangement and abundance of different types of habitat within a mosaic. Traditional approaches have been of little use in determining the nature and strength of fisheries-habitat links and Cappo et al. (1998) recommended that more innovative approaches, such as stable isotope tracers for food web analysis be implemented.

Need

Estuarine and coastal habitats, such as seagrass and mangroves, provide critical habitat for many species of juvenile fish and crustaceans. Projects which have mapped the extent of habitats such as seagrass and mangroves have not provided any clear understanding of how the changes in these habitats affects fisheries production. What is needed is an approach which recognises that estuarine habitats are inter-connected and most fish and prawns use a combination of habitats during their life or even during a day. Such an approach has been recognised by the FRDC reviews of Habitat and Seagrass research, the FRDC R & D plan for 2000-2005, the Marine Science and Technology Plan and the recent QDPI Strategic Plan for Marine Habitat Research. The following four key areas are, therefore, being addressed in this proposal:

1. Understand the utilisation of different habitat types within a critical chain of habitats by fish and decapods at different ontogenetic stages, sampling with a suite of different gear types to maximise coverage of species and size ranges over regional spatial scales;
2. Understand the processes linking different estuarine habitats such as seagrass and mangroves with fish and decapod assemblages and predict the effects of habitat change on these fisheries and on diversity;
3. Assess the importance of the nature and proximity of habitats adjacent to seagrass beds (e.g. mangroves, mudflats) in studies of seagrass fish/decapod assemblages, implementing 'landscape' scale programs to obtain the information needed to conserve, restore and enhance fisheries values; and
4. Use approaches developed for landscape ecology to determine the effects of variables such as the size and shape of seagrass beds, edge to area ratios and proximity to major currents and other habitats on seagrass fish/decapods communities.

Objectives

1. Develop appropriate methods for determining relationships between the spatial arrangement of estuarine habitats (habitat mosaics) and their biota including fish, crustaceans and molluscs and more sedentary epibenthos (e.g. comparing the fauna of isolated seagrass with that of seagrass close to mangroves);
2. Document patterns of abundance and diversity of fish and decapods in different habitat mosaics from degraded and relatively undisturbed areas of estuarine embayments, in two latitudinally separated areas (Moreton Bay and Hervey Bay) and taking into account position within an embayment (e.g. western side versus eastern side);
3. Increase our understanding of fisheries-habitat links using a combination of standardised survey methods in Queensland and make comparisons with the results obtained in Victoria using the same methods and gear types (Jenkins, 2001/036); and
4. Identify the relative importance of different nearshore habitats for key fish and decapod species from recruitment to older life-history stages and compare these results with those obtained for similar species and trophic groups in Victoria (Jenkins, 2001/036).

Section 1: Utilisation of Different Nearshore Habitats in Moreton Bay by Nekton Assemblages

1.1. Introduction

Estuarine systems are increasingly exposed to the effects of human population growth (Hinrichsen, 1998; Edgar et al., 2000), especially in Australia where the greater proportion of the population lives close to the coast (Saenger, 1995). The decline in global fisheries (Jackson et al., 2001; Worm et al., 2006) and losses of biodiversity and ecosystem services in marine and estuarine systems (Adger et al., 2005; Danielson et al., 2005) is increasingly focussing attention of scientists and managers on the causes and consequences of such perturbations.

Habitat destruction, along with pollution and excessive exploitation of resources, is implicated as one of the key direct effects humans are having on coastal biodiversity (Jackson et al. 2001; Lotze et al., 2006). Historical analysis has shown that estuarine systems globally have suffered increasing levels of human-induced degradation, with losses of biodiversity, changes to ecological services and marked declines in wetland habitats (Lotze et al., 2006). The loss of key coastal habitats such as mangroves has also been recently implicated as a key feature in determining the ecological resilience of ecosystems to catastrophic disturbance (Adger et al., 2005).

Conservation and protection of estuarine habitats, including saltmarshes, mangroves and seagrass beds, has often been driven by their role in supporting early life history stages of recreationally and commercially important fish and crustacean (prawns and crabs) stocks (Beck et al., 2001). Even unvegetated mudflats and sand banks are now more widely recognised as being important in the life history of a wide range of estuarine fisheries species (Peterson and Bishop, 2005; Miller and Skilleter, 2006). These same habitats are also considered to be important because of the large numbers of other species that are found there, although it is not clear whether this is a function of the specific habitats, or of the estuarine environment itself (Manson et al., 2005b). Protection of estuarine habitats is seen as a mechanism to conserve biodiversity from anthropogenic disturbance and degradation (Banks and Skilleter, 2002; Pressey, 2004; Stevens and Connolly, 2004). A critical component in the capacity to understand the effects of disturbance on biodiversity, mediated through habitat loss or change, is knowledge on the specific relationships between animals and the various habitats in which they are found (Manson et al., 2005b). A greater understanding of these linkages is possible through examination of the temporal changes in the abundance of fauna over periods when the composition, quality or extent of estuarine habitats has changed (Manson et al., 2003; Morrisey et al., 2003).

Moreton Bay has been the focus of numerous studies in the past, documenting the patterns of utilisation of different estuarine habitats by nekton (primarily fish and decapod crustaceans) with information on life history and ecological interactions and processes (reviewed by Tibbetts and Connolly, 1998). The most recent detailed studies were, however, done over a decade ago. Laegdsgaard and Johnston (1995) examined the temporal variation of fish assemblages using mangroves and seagrass beds while Masel and Smallwood (2000a, b) examined the dynamics of prawn populations using different nearshore habitats, both in western Moreton Bay. Similarly, during the early 1970s, extensive surveys of benthic macroinvertebrates were done in the Moreton Bay region (e.g. Stephenson and Campbell, 1977; Stephenson and Cook, 1977; Stephenson *et al.*, 1978; Stephenson, 1980a, b, c), providing detailed information on the distribution and diversity of benthic fauna in Moreton Bay at a range of spatial and temporal scales (reviewed by Skilleter, 1998).

There have been dramatic changes in the environmental conditions within the region, due to the exceptional population growth within SE Queensland (Skinner et al., 1998; Australian Bureau of Statistics, 2004) and the associated increases in pollution and other anthropogenic impacts on the waterways (Lotze et al., 2006). Marked changes have occurred in the Brisbane region, including modifications to the catchment of the Brisbane River (Bunn, 1998), development of the Brisbane Port complex (WBM Oceanics, 1993; Capelin *et al.*, 1998) and expansion of the airport just north of the Brisbane River. These changes and continued urban and rural development within the Moreton Bay catchment are likely to have affected the distribution, abundance and composition of the estuarine assemblages in the region, either directly or through changes in the distribution and quality of critical habitats (Manson et al., 2003).

The work done in this section was included in the overall project in response to requests from the FRDC to complete surveys using similar sampling gear and methods and in similar types of habitats as were being used in FRDC Project 2001/036 (Assessment of the importance of different near-shore marine habitats to important fishery species in Victoria using standardised survey methods, and in temperate and sub-tropical Australia using stable isotope analysis) being done by Fisheries-Victoria (Dr Greg Jenkins). The focus was on obtaining data on utilisation of key habitats (mangroves, seagrass and unvegetated mudflats) to complement the comparative work being done in that project examining food webs in different regions of Australia. Importantly, this work also provided an opportunity to examine the distribution and abundance of nekton in the different nearshore habitats, more than a decade after previous intensive surveys were completed.

1.2. Methods

1.2.1. Study Area

Five different types of habitat were chosen for initial sampling in spring 2002. In the intertidal zone they were the high shore mangroves (*Avicennia marina*), seagrass beds (*Zostera capricorni*), and unvegetated mudflats, while in the subtidal, seagrass (*Zostera capricorni*) and unvegetated sediments were sampled (Table 1.1). *Avicennia marina* is the only species of mangrove that occurs in Victoria, so only stands of mangroves dominated by *Avicennia marina* in Moreton Bay were selected for sampling. This would allow more relevant and logical comparisons to be made between the results from the work being done in Victoria (FRDC 2001/036) and Queensland on utilisation of mangrove habitats. For similar reasons, sampling of seagrass in Queensland to date has focussed on beds dominated by *Zostera capricorni*. Work so far on the Victorian project has concentrated on *Zostera*/*Heterozostera* stands.

For each of these 5 habitat types, three sites were selected in each of eastern and western Moreton Bay (Table 1.1; Figure 1.1). Several criteria were used in selection of the sites. First, where possible, sites containing several of the different habitat types in the one area were selected. This was done to allow more direct comparisons of the utilisation by epibenthic species among the different habitat types, without confounding from spatial variation. Second, the future choice of sites for comparisons among different habitat mosaics (combinations of several habitat types) was taken into account. This was done so that the data from the general sampling programme would also serve to provide information on how the composition of the nekton communities in different habitats may correlate with variation in the composition of different mosaics. Third, the sites were selected to allow the analysis of spatial variation in the utilisation of the different habitat types at a range of scales, including 10's of metres (among replicate samples), 100-1000 m (among sites of the same habitat type) and 10's of kilometres (western and eastern Moreton Bay). The choice of these spatial scales was determined after discussion with MAFRI team (FRDC Project 2001/036) to allow more appropriate comparisons of the results from the two. Although the Victorian study compared two bays separated by 100's of kilometres, a much greater distance than was considered here, the comparisons between eastern and western Moreton Bay still represent significantly differently environments because of the marked differences in the levels of marine influence across the Bay, including variation in salinity, turbidity, water temperature and sediments **(**Young 1978; O'Brien 1994; Gabric et al. 1998**).**

Subsequent sampling in summer 2002/03, autumn and winter 2003 was only done in 3 of these 5 habitats. The subtidal seagrass and unvegetated sediments were not sampled after discussions with the MAFRI team who were not sampling these habitats as part of their study. Sampling was therefore focussed in the intertidal habitats (mangroves, seagrass and unvegetated mudflats) to maximise the comparative information that could be provided between the two bioregions (SE Queensland and Victoria).

1.2.2. Selection of Sampling Gear

Discussions were held at the beginning of the project with the members of the MAFRI team (Drs Greg Jenkins and Jeremy Hindell: FRDC Project 2001/036) to determine the types of gear to be used. Several different gear types were selected for use in both projects. Sampling in mangrove habitats was done using fyke nets (70 cm square rings, 6 mm honeycomb mesh, 70 cm high wings x 4.5 m long, with two 5 m metre detachable wings). Five replicate fyke nets were deployed approximately 10-20 metres in from the edge of the forest edge. This was different from the way the nets were deployed in the Victorian project where the diminutive nature of the mangrove trees prevented the nets from being set within the forest. The decision was taken to deploy the fyke nets within the mangrove forest for the Queensland work because this provides a more reliable estimate of the species that are utilising the forest habitat than samples collected along the edge of the mangroves, or using creeks that run through the forest.

The design of the fyke nets, with two detachable wings, allowed the nets to be positioned among the trees in such a way that animals retained in the net were those that were moving through the structure of the mangroves (Figure 1.2). Nets were deployed at least 5 metres away from any drainage channels through the mangroves, again so that the catch reflected those species utilising the main forest area. Each fyke was deployed at the top of the tide and retrieved once the tidal level has receded below the level of the end of the wings (approximately 3 hours soak time), with the total catch (soak) period recorded. All catches were standardised to a two hour soak time for comparisons among sites.

Intertidal and subtidal seagrass (*Zostera capricorni*) and unvegetated sediments were sampled with a 6 metre long seine net (2 m high x 1 mm mesh) hauled along a 25 metre path at a water depth of 0.75 – 1.0 m on either side of flood tide. The mouth of the seine was set at 4 m wide, so each haul of the seine sampled an area of 100 m^2. Four replicate samples using the seine net were collected at each of the sites in spring, but in subsequent seasons, six replicate samples were collected after initial analysis of the data (see Results).

1.2.3. Statistical Analysis

1.2.3.1. Utilisation of Different Habitats – Spring 2002

Spatial variation among the three sites in each habitat in the composition of the nekton assemblage (fish, decapod crustaceans and cephalopod molluscs) using each of the different habitat types was examined using non-metric multidimensional scaling ordination (nMDS) and analysis of similarities (ANOSIM) on untransformed data. Data on catches in the mangroves (using fyke nets) were standardised to a 2 hour soak time. Species contributing most to the separation of the sites were identified using SIMPER (Clarke, 1993). Differences in the abundance of species contributing at least 5% to the separation of the nekton assemblages among the three sites were then analysed with one-factor analyses of variance, after testing for homoscedasticity of variances with Cochran's test. Where necessary, data were transformed to $\log_e(x+1)$ to remove heteroscedasticity. If transformation was unsuccessful and variances were still heteroscedastic, untransformed data were analysed but a more conservative P-value of 0.01 was used to indicate significance. In most cases, the design was completely balanced (equal numbers of replicates and levels within treatments) and ANOVA is very robust to violations of the assumption of homoscedastic variances under these circumstances (Underwood, 1981).

1.2.3.2. Regional Comparisons – Western versus Eastern Moreton Bay, Spring 2002

The data from spring 2002 were used to compare the abundance and composition of the nekton assemblage between eastern and western Moreton Bay with the primary aim of determining if the two regions, which differ markedly in terms of the physical characteristics such as sediments (Williams 1958; Branford 1981; Rulifson 1981; Somers 1987) and salinity (Dall 1958; Gunter et al. 1964). Variation in the composition of the nekton assemblage was first examined using non-metric multidimensional scaling ordination (nMDS) and analysis of similarities (ANOSIM) on untransformed data. A two factor nested ANOSIM was used to examine the main effect of differences between regions and the nested effect of sites within the regions. Data on catches in the mangroves, using fyke nets, were standardised to a 2-hour soak time. Catches in the unvegetated habitats (intertidal and subtidal) were extremely patchy, with some hauls of the seine net only capturing a few individuals. Multivariate analysis of the complete dataset for these habitats were dominated by the large number of samples with very few specimens of relatively uncommon species (at least in these samples). To allow a more meaningful test of whether there were identifiable differences in the composition of the nekton assemblage between eastern and western Moreton Bay for these two habitats, only species that were represented by at least 5 individuals were included in the analyses. The small numbers of individuals caught in these two habitats and the large number of species that were only caught in one or other of the two regions (see Section 1.3 Results below) also precluded meaningful univariate analysis of the abundance of individual species. Species contributing most to the separation of the assemblages in eastern and western Moreton Bay were identified with SIMPER.

In the other three habitats (i.e. mangroves, intertidal and subtidal seagrass beds), the abundance of species identified by SIMPER as contributing at least 5 % to the separation of the assemblage for that habitat were then analysed using 2 factor analyses of variance. The factors were region (fixed) and sites within region (nested, random). Data were transformed to $\log_e(x+1)$ where necessary to meet the assumption of homoscedastic variances. Where transformation of the data did not remove heteroscedasticity, untransformed data were then analysed but using a more conservative P value of 0.01 to indicate significance.

1.2.3.3. Temporal Variation – Western and Eastern Moreton Bay

In each of the regions, differences in the composition of the nekton assemblage using each of the habitats during the four sampling periods (i.e. spring, summer, autumn and winter) were analysed using ANOSIM based on a dummy categorical variable created from the interaction between the main effect of region and season. This was done because it is not possible to test formally for interactions using ANOSIM (Clarke, 1993). Pairwise tests within ANOSIM for relevant combinations of the four sampling periods by sites within the region, were done to determine the presence of these interactions between the main effects. The primary focus here was to determine if there was evidence of greater or lesser spatial variation (i.e. differences among sites) in any period and if such differences were associated with any particular site. This restricted the number of relevant comparisons to 12, represented by a comparison of the R-statistic for each combination of sites (i.e. 3 values) in each of the 4 seasons. Examination of the values for the R-statistic from these pair-wise tests and the ordination plots (see below) was used to infer the presence of any interactions between the effects of proximity and density of seagrass bed. Differences in composition of the nekton assemblage were also examined graphically using non-metric multidimensional scaling (ordination) using the Bray-Curtis similarity measure on untransformed data (Clarke, 1993). The contributions of different taxa to the variation in the composition of the nekton assemblages among the seasons were examined using SIMPER. These analyses focussed on determining which species were most responsible for temporal shifts in the composition of the assemblage from season to season, rather than comparing all combinations of seasons (i.e. spring versus summer, summer versus autumn but not spring versus autumn).

1.2.3.4. Comparisons of Composition between Mangroves and Adjacent Seagrass

Differences in the sampling gear used in the mangroves (fyke nets) and intertidal seagrass and mudflat habitats (seine nets) precluded direct comparisons of the abundance of different species between the two habitats. Abundance in the mangroves was estimated using a catch-per-unit effort approach (catches corrected for soak time of the net) whereas abundance in the intertidal seagrasses and mudflats was estimated as a per unit area basis.

The composition of the assemblages using the mangroves and intertidal seagrass could be compared though, based on presence/absence of species was possible though, providing some indication of the extent to which specific habitats support discrete assemblages of fish and decapod crustaceans. The mangroves were compared with the intertidal seagrass, but not the mudflats, because of the spatial arrangement of the different sites that were used for sampling each of the habitats. In eastern and western Moreton Bay, the sites sampled in the mangroves and those for the intertidal seagrass were always adjacent to each other (Table 1.1), whereas those sampled for the intertidal mudflats were in other areas. This meant that when sampling was done in the mangroves and intertidal seagrass habitats, during high tide, nekton had access to either of the habitats. These analyses therefore identified the species that were primarily associated with one or other of the habitats when they had the capacity to utilise either of them.

Data on the composition of the nekton assemblage using the mangroves and intertidal seagrass beds were analysed using ordination (nMDS) and ANOSIM on presence-absence transformed data. ANOSIM was done using a one factor design based around a dummy variable that combined the effects of habitat (mangrove vs seagrass) and season (spring, summer, autumn, winter) into a single category (e.g. mangrove-spring, seagrass-winter). This meant there were 8 levels of this factor that were compared in the ANOSIM. Pairwise multiple comparisons were used to examine the different groups, but were restricted to comparisons of the two habitats for each of the four seasons (e.g. mangroves-spring versus seagrass-spring). There was no specific interest in comparing the use of the two habitats in different seasons (e.g. mangroves-spring versus seagrass-summer) because such comparisons would not provide information on how utilisation of the different habitats varied during periods when nekton had access to both habitats.

1.3. Results

1.3.1. General Characteristics of the Fauna – Western Moreton Bay

1.3.1.1. Mangrove Forests

A total of 4,796 individuals from 41 species was sampled in the western Moreton Bay mangroves in Spring 2002 (Table 1.2). The nekton assemblage comprised 29 species of fish and 12 species of invertebrates (decapods and cephalopods). The most abundant species of fish was the toadfish *Tetractenos hamiltoni* (1,858 individuals), while the greasyback prawn, *Metapenaeus bennettae* was the most abundant invertebrate (648 individuals).

The overall composition of the nekton assemblage using the mangroves in Spring 2002 varied significantly among the three sites (ANOSIM, $P < 0.001$; Figure 1.3A). Five to seven species contributed 60-70 % to the separation of each of the sites from each other, with variation in the abundance of the toadfish, *Tetractenos hamiltoni*, providing the best discrimination between pairs of sites (Figure 1.3B).

More individuals of each of these species (Figure 1.3B) occurred in the mangroves at Fisherman Islands-South than either of the other sites and, in most cases, these differences were substantial and significant (Figure 1.4 & 1.5). Separation of Fisherman Islands-North from Thorneside was a result of these species being relatively uncommon at one of these sites, but not the other (Figures 1.4 & 1.5).

1.3.1.2. Intertidal Seagrass

There was fewer species caught from the intertidal seagrass beds in the western Bay than in the mangroves, with only 17 species of fish and 14 species of invertebrates present (Table 1.3). Estimates of abundance are not directly comparable between the mangroves and intertidal seagrass beds because of the different sampling methods that were used (fyke nets in the mangroves and seine nets in the seagrass). Gobies were the most abundant fish caught (1,743 individuals), comprising mostly *Arenigobius frenatus* and *Favonigobius exquisitus*, followed by the common fortescue, *Centropogon australis* (317 individuals), the eastern striped trumpeter, *Pelates sexlineatus* (248 individuals) and the hairy pipefish, *Urocampus carinirostris* (147 individuals).

The most abundant invertebrate that was sampled was the commercially important brown tiger prawn, *Penaeus esculentus* (320 individuals). Several different carid shrimp, including *Latreutes compressus*, *Latreutes porcinus* and *Macrobrachium* sp., were also relatively abundant (> 150 individuals).

The composition of the nekton assemblage using the intertidal seagrass in Spring 2002 varied significantly among the three sites (ANOSIM, $P < 0.01$; Figure 1.6A). There was noticeably less variation in the composition of the individual samples from Thorneside than at either of the other sites (Figure 1.6A), primarily due to the dominance of gobies (*Arenigobius frenatus* and *Favonigobius exquisitus*) in the catch at this site and the relatively uniform abundance of these fish in those samples. Six to seven species contributed ~70 % to the separation of each of the sites from each other, with variation in the abundance of gobies providing the best discrimination between pairs of sites (Figure 1.6B).

There were more fish found in the seagrass beds at Thorneside than the other two sites (Figure 1.7A) with very few sampled from Fisherman Islands South. Similarly, there were significantly more species caught at Thorneside, with the smallest number of species occurring at Fisherman Islands South. The abundance of each of the abundant species of fish followed this same pattern, with more individuals occurring at Thorneside, followed by Fisherman Islands North, with the smallest numbers at Fisherman Islands South (Figure 1.7C-G).

The general pattern was similar for the abundant invertebrates in the intertidal seagrass beds (Figure 1.8), with the exception of *Metapenaeus ensis* (Figure 1.8B) which was more abundant at Fisherman Islands North although there was no significant different between that site and Thorneside.

1.3.1.3. Intertidal Unvegetated Mudflats

There were considerably fewer individuals and species caught from the intertidal mudflats than the other habitats in Spring 2002 with only 16 species of fish and 11 species of invertebrates occurring in the samples, although many of these only occurred as single individuals (Table 1.4). The most abundant species of fish were the sandy sprat, *Hyperlophus* sp. (53 individuals) and the puffer, *Marilyna pleurosticta* (12 individuals). The most abundant invertebrate was the small squid, *Sepioteuthis* sp. (103 individuals).

The composition of the nekton assemblage using the intertidal mudflats was significantly different at Nudgee Beach South compared with the other two sites (ANOSIM, $P < 0.01$) which were not different from each other. Two of the samples from Nudgee Beach North did not contain any individuals and only 6 individuals were collected from the other two samples. There appeared to be more variation in the composition of the assemblage at Lota than at Nudgee Beach South (Figure 1.9A), but only 20 individuals were sampled, so this pattern may not be biologically meaningful. Graphs of the abundance of individual species, comparing the three sites, are not shown given the small number of animals that were caught in this habitat in Spring 2002.

1.3.1.4. Subtidal Seagrass

Of the 5 habitats sampled in spring 2002, the subtidal seagrass supported the greatest number of species, with 29 species of fish and 25 species of invertebrates occurring in the three sites (Table 1.5). The most abundant fish were the six-lined trumpeter, *Pelates sexlineatus* (566 individuals), gobies (primarily *Arenigobius frenatus* and *Favonigobius exquisitus*), the fan-bellied leatherjacket, *Monacanthus chinensis* (190 individuals) and the common fortescue, *Centropogon australis* (159 individuals). The most common invertebrates were the carid shrimps, *Periclimenes obscurus* (966 individuals), *Hippolyte caradina* (571 individuals), *Latreutes pygmaeus* (224 individuals), *Hippolyte* sp. (109 individuals) and the prawns *Penaeus esculentus* (101 individuals) and *Penaeus plebejus* (116 individuals).

The composition of the nekton assemblage varied significantly among the different sites (ANOSIM, $P < 0.01$; Figure 1.10A). Six to seven species contributed ~70 % to the separation of the sites (SIMPER, Figure 1.10B), with *Periclimenes obscurus* and *Hippolyte caradina* providing the best discrimination between pairs of sites.

In contrast to the intertidal habitats, where one or other of the sites tended to have more individuals of most species of fish, there was no clear pattern in the subtidal seagrass habitat, where the different sites had more of some species but not others. For example, the abundance of gobies was significantly greater at Fisherman Islands North than the other two sites (Figure 1.11C), whereas Fisherman Islands South had the greatest number of the fan-bellied leatherjacket, *Monacanthus chinensis* (Figure 1.11F) and there was no significant difference among the three sites in species richness (Figure 1.11B) or the abundance of common fortescue, *Centropogon australis* (Figure 1.11D) and the six-lined trumpeter, *Pelates sexlineatus* (Figure 1.11E). A similar pattern was evident for the decapod crustaceans, with either Fisherman Islands North or South having the greatest abundance of shrimp (*Periclimenes obscurus* – Figure 1.12B; *Latreutes pygmaeus* – Figure 1.12C; *Penaeus plebejus* – Figure 1.12D; *Penaeus esculentus* – Figure 1.12E).

1.3.1.5. Subtidal Unvegetated Mudflats

Only 10 species and a total of 98 individuals were caught in the samples from the unvegetated, subtidal sediments and all but three of these species were represented by less than 5 individuals across the 12 samples (4 samples from each of 3 sites). The estuarine anchovy (*Thryssa aesturia*) was the most abundant species (34 individuals) followed by the eastern king prawn (*Penaeus plebejus*) with 23 individuals and mysids with 21 individuals. No formal analyses were done on these data, given the small numbers of individuals caught and the small number of samples in which these individuals occurred.

1.3.2. Regional Comparisons – Western versus Eastern Moreton Bay, Spring 2002

1.3.2.1. Mangrove Forests

There were more species of both fish and invertebrates caught in the mangroves on the western side of Moreton Bay in Spring 2002 than on the eastern side (Table 1.6). An additional 15 species were caught in the western side mangroves that did not occur on the eastern side. In contrast, only 4 species found on the eastern side, did not occur on the western side (Table 1.6). Overall, more individuals were caught on the eastern side than the western, but this was a function of very large numbers of two species of schooling fish, the estuary perchlet (*Ambassis marianus*) and Ogilby's hardyhead (*Atherinomorus ogilbyi*) and a carid shrimp (*Palaemon debilis*) (Table 1.6).

Not surprisingly, given the larger number of species that occurred in western than eastern Moreton Bay, the composition of the nekton assemblage using the mangroves varied significantly between the two regions (Figure 1.13), with complete separation of the two groups. Five species contributed 77 % to the separation of the two regions in the multivariate analysis, with the carid shrimp, *Palaemon debilis*, accounting for 34 % of the differences (Figure 1.13B).

Despite the large differences in the composition of the assemblage between the western and eastern sides of Moreton Bay, there were few significant differences in the abundance of individuals or the number of species. Generally speaking, one of the three sites within each region had significantly more individuals than the other sites and this smaller scale (sites within regions) variation dominated the analyses, precluding the detection of regional differences. In the western Bay, it was usually the same site that supported significantly larger numbers of animals and species (Figure 1.14A-D), but this was not the case in the eastern Bay where differences were species specific. Only *Palaemon debilis* was significantly more abundant on one side (East) of the Bay compared with the other (West) (Figure 1.14E).

1.3.2.2. Intertidal Seagrass

In contrast to the situation for the mangroves, many more species were caught in the intertidal seagrasses of eastern than western Moreton Bay, the opposite was true for the intertidal seagrass beds (Table 1.7). Of the 33 different species sampled, 10 of these only occurred in the eastern Bay, whereas only 2 species were found exclusively in the western Bay (Table 1.7). The total numbers of individuals were quite similar on both sides of the Bay, but this did not reflect differences in abundance for individual species.

Despite the differences in which species were found in the samples from each side of Moreton Bay, there was no overall significant difference in the composition of the nekton assemblage between the two regions (ANOSIM, P > 0.20), with considerable overlap in samples plotted by ordination (Figure 1.15A). There was, however, significant differences in the composition of the assemblage among the sites within each region (ANOSIM, P < 0.001) and this dominated the analysis examining differences between the regions. Four species contributed 74 % to the differences between the two regions (SIMPER analysis), with the eastern king prawn, *Penaeus plebejus*, accounting for 35 % (Figure 1.15B).

Analyses of the abundance of individual species were also dominated by very large and significant variation among the three sites within each of the two regions. In some cases, there was significant variation among the sites in each of the two regions (i.e. eastern and western Moreton Bay; e.g. total number of individuals – Figure 1.16A, *Centropogon australis* – Figure 1.16C, *Pelates sexlineatus* – Figure 1.16E). For other variables, variation was more pronounced on the western side of the Bay (e.g. species richness – Figure 1.16B, *Urocampus carinirostris* – Figure 1.16F) than on the eastern side. Finally for *Latreutes porcinus* (Figure 1.16G) and *Penaeus plebejus* (Figure 1.16H), most of the individuals were only sampled from a single site on in the western or eastern side of the Bay respectively.

1.3.2.3. Intertidal Unvegetated Mudflats

The number of individuals caught on the intertidal unvegetated mudflats was generally small in both eastern (397) and western (103) Moreton Bay although a greater number of individuals and species was caught in the eastern than western region (Table 1.8). Of the 28 species that were sampled, 13 of these only occurred in the eastern Bay, whereas only 9 species were found exclusively in the western bay (Table 1.8). The abundance of individual species was also generally very small, with many species represented by fewer than 5 individuals.

There was very little difference in the composition of the nekton assemblage using the unvegetated mudflats in eastern and western Moreton Bay (ANOSIM, P > 0.90; Figure 1.17A). Four species contributed greater than 5 % to the small amount of separation between the assemblages in the eastern and western regions (Figure 1.17B), the eastern king prawn (*Penaeus plebejus*) contributing 32 %. None of these species provided a useful indicator for the specific assemblages though because they tended to be abundant in one region but were either not present in the other, or in very small numbers. Given the very small numbers of individual species that were caught in one or other of the two regions, no formal univariate analyses were done comparing the abundances of individual species.

1.3.2.4. Subtidal Seagrass

Of the five habitats that were sampled, subtidal seagrass beds, primarily *Zostera capricorni*, supported the largest number of species and individuals of nekton in western and eastern Moreton Bay (Table 1.9), with 57 species caught in the western region and 38 species caught in the eastern region. Of the 66 species in total that were caught, 28 species occurred exclusively in the western region, while only 9 species were exclusive to the eastern region (Table 1.9).

There was almost complete separation of the nekton assemblages in the two regions (ANOSIM, $P < 0.001$; Figure 1.18A). Seven species contributed at least 5 % each to the separation of the assemblages, combined accounting for 69 % of the distinction of the two regions. Only two of these species were fish (the gobies and the eastern striped trumpeter, *Pelates sexlineatus*). Decapod crustaceans and mysid shrimp accounted for the other five species responsible for separation of the nekton assemblages between the regions.

The number of species was significantly greater in the western than the eastern region (ANOVA, $P < 0.004$; Figure 1.19A), with very little variation in species richness among the sites in either region (ANOVA, $P < 0.43$). Analyses of the abundance of individual species were often dominated by very large and significant variation among the three sites within one or other of the two regions. In some cases, this small-scale variation obscured apparent regional differences in abundance. The total number of individuals caught was 66.5 % greater in the western than eastern regions (Figure 1.19B), but this difference was not significant (ANOVA, $P > 0.22$). There was significant variation among the sites ($P < 0.05$), especially in the western Bay. Similarly for the fan-bellied leatherjacket, *Monocanthus chinensis*, there was an overall increase of 459 % in the abundance from eastern to western Moreton Bay (Figure 1.19E), but this was not significantly different (ANOVA, $P > 0.17$) because of the highly significant variation among the sites ($P < 0.001$), especially in the western region.

In contrast, the number of common fortescue (*Centropogon australis*) was significantly greater in the western than eastern region (an increase of 489 %; ANOVA, $P < 0.05$), despite the variation among the sites on both sides of the Bay (Figure 1.19C). Similarly, there was a significant increase of 1,430 % in the abundance of *Pelates sexlineatus* from eastern to western Moreton Bay (Figure 1.19F). Three species of carid shrimp were sufficiently abundant to analyse separately. *Palaemon obscurus* was extremely abundant in the western region, but only occurred in very small numbers in the eastern region (Figure 1.19G). *Hippolyte caradina* was very abundant at two of the three sites in western Moreton Bay but did not occur at the third site, so the average abundance was similar between eastern and western Moreton Bay (Figure 1.19H; ANOVA Region – $P > 0.81$; Sites (Region) – $P < 0.002$). Finally, the abundance of *Latreutes pygmaeus* was relatively consistent at all sites (Figure 1.19I), with no significant differences in the numbers of this shrimp between regions (ANOVA, $P > 0.58$) or among sites within regions (ANOVA, $P > 0.07$).

1.3.2.5. Subtidal Unvegetated Mudflats

Only small numbers of animals were caught in the unvegetated subtidal sediment habitat, on either side of Moreton Bay (Table 1.10), although considerably more individuals were present in the western than eastern Bay. A total of 26 species occurred in the samples, but 13 of these were represented by fewer than 5 individuals and 15 species occurred exclusively in the western Bay (Table 1.10). The clear distinction between eastern and western Moreton Bay in which species were caught in the two regions was reflected in the ordination of the data (Figure 1.20A) which showed clear separation of the two assemblages, confirmed by multivariate analysis (ANOSIM, $P < 0.001$). Seven species contributed greater than 5 % to the separation of the assemblages (Figure 1.20B), but only *Penaeus plebejus* could be considered useful in terms of characterising the different groupings because the other species were rare or absent in one or other of the two regions. Given the very small numbers of individual species that were caught in one or other of the two regions, no formal univariate analyses were done comparing the abundances of individual species.

1.3.3. Temporal Comparisons

1.3.3.1. Mangroves

Western Moreton Bay: There were significant changes in the composition of the nekton assemblages using the mangroves over the course of the 12 months sampling was done (ANOSIM, Seasons, $P < 0.001$), but the amount of change in the assemblage varied from site to site. Ordinations of the data show clear groupings of samples within each of the four seasons (Figure 1.21A), but there is also considerable overlap of the samples collected in different seasons. There is no obvious temporal progression in the composition of the assemblage: samples collected in, for example, spring were not more similar to those caught in summer (the next set of samples collected) than those from either autumn or winter (Figure 1.21B). This lack of clear temporal progression in the ordinations is a function of the interaction between Season x Site, where the amount of change from season to season varied among the three sites. This is most easily seen in the ordination of the sites (taken as the centroids of the samples within an site) which clearly indicates that the composition of the assemblage is as different among the three sites sampled within a season as among the four seasons (Figure 1.21B).

At Fisherman Islands North, the composition of the assemblage was most similar between spring and summer, but at each of Fisherman Islands South and Thorneside, the least amount of change in composition was seen between summer and autumn. At Thorneside in particular, there was very little change in the composition of the assemblage between summer and autumn (Table 1.11A). Thus, the composition of the assemblage using any particular site changes considerably over the course of a year, but in a different way from the other sites.

Despite this small-scale (among sites) spatial variation in the overall composition of the nekton assemblage, some species did provide good distinction among the assemblages found in the mangroves at different times of the year. Changes in the abundance of five different species (*Tetractenos hamiltoni, Palaemon debilis*, *Palaemon serenus*, *Metapenaeus bennettae* and *Atherinomorus ogilbyi*) provided good discrimination in the composition of the nekton assemblage using the mangroves over the course of the year. The common toadfish, *Tetractenos hamiltoni*, contributed the most to the separation of the assemblage in successive seasons from spring to winter (Table 1.12), with a progressive decline in abundance into the cooler months of the year. The carid shrimps, *Palaemon debilis* and *P. serenus*, increased in numbers from spring through to summer but were very rare in samples in autumn, before returning to large numbers again in winter. Mysid shrimp and the flat-tailed mullet, *Liza argentea*, also contributed to the separation through time of the assemblage using the mangroves, but neither taxon was a useful discriminator of temporal changes because of the large standard deviation around the dissimilarity values indicating extremely patchy distribution among replicate fyke samples and among sites. *Liza argentea* (decrease in abundance) and the golden-lined whiting, *Sillago analis* (3-fold increase in abundance), provided discrimination between the assemblage from autumn to winter. Palaemonid shrimp (those not identified to species level) also contributed to the separation of the assemblages over this period but did not provide useful discrimination because of a patchy distribution.

Eastern Moreton Bay: A similar pattern of temporal change was seen in eastern Moreton Bay, with the assemblage using the mangroves changing significantly over the course of the year (ANOSIM, Seasons, $P < 0.001$) but these changes were not synchronous among the three sites. The significant variation among the sites obscured any clear patterns in the ordination of the data (Figure 1.22A), although the seasonal variation was evident. The interaction between season and site is evident from the ordination based on the site centroids (Figure 1.22B), showing that in some seasons (e.g. spring and winter) the sites are more similar to each other than they are in the other seasons (summer and autumn).

At One Mile, the assemblage sampled in each season changed increasingly as the year progressed (i.e. spring and summer were very similar to each other, summer and autumn were less similar, autumn and winter were very different). In contrast, at Wallen Pools South there was a more even distribution in terms of the similarity in the assemblage from season to season (Table 1.11B). The most obvious difference in the patterns of temporal change between western and eastern Moreton Bay was in the contribution of different species to the changes from season to season. In eastern Moreton Bay, only two species provided the discrimination between the assemblages from season to season – the estuary perchlet, *Ambassis marianus*, and the carid shrimp, *Palaemon debilis* (Table 1.13). Ogilby's hardyhead, *Atherinomorus ogilbyi*, contributed 7-14 % to the separation of the assemblages among the seasons, but did not provide a useful discrimination because of its patchy distribution.

1.3.3.2. Intertidal Seagrass

Western Moreton Bay: There was considerable change in the composition of the nekton assemblage using the intertidal seagrass beds over the course of the year, but as noted above for mangroves, the variation among the three sites precluded any simple interpretation based on seasonal patterns. Some grouping of samples based on season is evident in the ordination (Figure 1.23A), but there was also considerable overlap among the seasons. The magnitude of changes in the composition between successive seasons (e.g. spring to summer) varied among the three sites. For example, Fisherman Islands North showed little change in the composition of the assemblage from Spring to Summer ($R = 0.23$) compared with Thorneside which showed a large amount of change ($R = 0.94$) (Table 1.11C). A similar level of variation among the three sites was seen for each of the periods over which change in composition was assessed (spring-summer, summer-autumn, autumn-winter).

Despite the spatial variation among the sites, there was significant shifts in the nekton assemblage detected among successive seasons (ANOSIM, $P < 0.001$; pairwise comparisons, P from 0.002 - 0.019). Discrimination in the composition of the nekton assemblage between successive seasons was primarily due to changes in the abundance of prawns (*Metapenaeus bennettae* and *Penaeus plebejus*), carid shrimp (*Palaemon serenus* and *Periclimenes obscurus*), the six-lined trumpeter, *Pelates sexlineatus* and gobies (Table 1.14). *Palaemon debilis* contributed to the separation of the assemblages but was not a particularly useful discriminator due to a patchy distribution.

Eastern Moreton Bay: A similar pattern of large variation among the sites in the temporal changes between successive seasons was evident in eastern Moreton Bay. There were pronounced changes in the composition of the nekton assemblage using the intertidal seagrass over the course of the year (Figure 1.23B), but the ordination of the data was dominated by considerable variation among replicate samples and the sites. The degree of change in the composition of the assemblage between successive seasons also varied considerably among the three sites. For example, both Myora Springs and Wallen Pools South showed relatively small amounts of change from spring to summer (R = 0.27 and 0.26 respectively) compared with One Mile (R = 0.87). One Mile consistently showed a greater amount of change in the composition of the assemblage from season to season than the other two sites (Table 1.11D).

The most notable difference in the pattern of change between western and eastern Moreton Bay was that in the latter, fewer species contributed to the distinction in the composition of the assemblage between successive seasons. On each occasion, *Penaeus plebejus* and gobies provided good discrimination in the composition of the assemblages between successive seasons (Table 1.15). *Periclimenes obscurus* contributed to the separation of the assemblages but did not provide good discrimination due to a patchy distribution

1.3.3.3. Intertidal Unvegetated Mudflats

Western Moreton Bay: Fewer individuals and species of nekton were caught over the intertidal mudflats than the other habitats at any time during the year. There were significant changes during the year in the composition of the assemblage over the mudflats (ANOSIM, $P < 0.001$; pairwise comparisons, P from 0.001 - 0.003). The extent to which the composition of the assemblage change between successive seasons varied considerably among the three sites, indicative of a Season x Site interaction, but the rank order was the same from site to site, with the greatest amount of change between spring and summer and the least amount of change from autumn to winter (Table 1.11E). The change in composition between successive seasons was primarily due to changes in the abundance of only a few species (Table 1.16).

Eastern Moreton Bay: The pattern of change of usage of the intertidal mudflats in the eastern Bay was similar to that in the western region, with significant shifts in the composition of the assemblage between successive seasons (ANOSIM, $P < 0.001$), but also considerable variation in the extent of change among the three sites (Figure 1.24A), indicative of a Season x Site interaction. In contrast to western Moreton Bay though, the level of change between successive seasons varied from site to site (Table 1.11F). A larger number of species contributed to the change in the composition of the assemblage between successive seasons in the eastern compared with the western Bay (Table 1.17), although several species provided poor discrimination because of extremely patchy distribution (e.g. planktivorous fish such as Hardyhead – *Atherinomorus ogilbyi* and estuary perchlet – *Ambassis marianus*).

1.3.3.4. Comparisons of Composition between Mangroves and Adjacent Seagrass

The was a clear distinction in the composition of the nekton assemblage using the mangroves and intertidal seagrass in western (Figure 1.25A) and eastern

(Figure 1.25B) Moreton Bay, with no overlap evident in the ordinations. These analyses were based on presence/absence data so reflect the overall composition of the nekton assemblage found in the two adjacent habitats.

In western Moreton Bay, the distinction in the composition of the assemblage between the two habitats was greater in autumn and winter ($R = 0.84$ and $R = 0.99$ respectively) than spring and summer ($R = 0.57$ and $R = 0.54$ respectively). A large number of species contributed to the separation of the assemblages in the mangroves and seagrass, with no single species contributing more than 5 % to the overall distinction between the two habitats. Mangrove-specific species were primarily fish whereas the seagrass-specific species were primarily carid and penaeid shrimp.

In eastern Moreton Bay, the distinction in the composition of the assemblage between the two habitats was equally as strong in each of the four seasons ($R > 0.93$). As for western Moreton Bay, a large number of species contributed to the distinction in the assemblages between the habitats, but no species contributed more than 5 %. Again, mangrove-specific species were primarily fish whereas the seagrass-specific species were primarily carid and penaeid shrimp.

1.4. Discussion

Despite the considerable changes that have occurred in the Brisbane region over the last decade, including expansion of the port, increased urban development and changed rural activities along the foreshores (e.g. Albert et al., 2005) and problems with pollution entering western Moreton Bay (Udy and Dennison, 1997; Dennison and Abal., 1999; Shaw et al., 2004), the nearshore estuarine habitats in this subtropical embayment still support a rich and abundant assemblage of nektonic fish and decapod crustaceans. A total of 65 species (20,990 individuals) of fish, 26 species (29,385 individuals) of decapod crustaceans and 5 species (1,859 individuals) of cephalopods was sampled from mangroves, seagrass and mudflats in western Moreton Bay over the course of 12 months, from spring (September-October) 2002 through to winter (June-July), 2003. Similar numbers of species and individuals were sampled from eastern Moreton Bay over the same period and a similar number of species of fish was sampled from western Moreton Bay several decades ago (Stephenson and Dredge, 1976).

More species of fish and decapods were caught in the mangroves in western than eastern Moreton Bay, even though larger numbers of individuals were caught on the eastern side. Western Moreton Bay is impacted by a range of urban and industrial pollutants entering the water directly or via the Brisbane River (e.g. Connell and Miller, 1998; Shaw et al., 2004), whereas eastern Moreton Bay receives relatively clean water directly from the Tasman Sea, via entrances between Moreton and North Stradbroke Islands and North and South Stradbroke Islands (Neil, 1998). Given the potential impacts of pollution, greater species richness in western Moreton Bay may seem surprising but is consistent with the overall patterns of species densities across the embayment (Davie and Hooper, 1998). There are broad scale differences in the composition of the assemblages, based on the degree of marine influence which varies from east to west across the bay (e.g. Young and Wadley, 1979). Interestingly though, the opposite pattern was evident for intertidal seagrass, with more species caught in the eastern than western Bay, although many of these were only represented by a single individual. More individuals were caught in the eastern Bay, consistent with the pattern for the mangroves.

The species richness and abundance off nekton in subtropical Moreton Bay was considerably greater than recorded for Victorian estuarine habitats, using precisely the same sampling gear over much the same time period (Hindell et al., 2004; Hindell and Jenkins, 2004). For example, one of the more abundant species in the Victorian study was the smooth toadfish, *Tetractenos glaber*. Abundance per hour of fishing time using fyke nets ranged from 1-3 individuals. The ecological equivalent species in Moreton Bay, the common toadfish, *Tetractenos hamiltoni*, reached abundances of 25-150 individuals per hour of fishing time using the same design fyke nets. Overall abundances (i.e. all individuals) were also considerably less in Victoria than in subtropical Moreton Bay (see also Clynick and Chapman, 2002, who sampled mangroves in the temperate Sydney region). Until recently, most of our knowledge about the dynamics of nekton in mangrove forests had been based on work done in tropical systems (e.g. Robertson and Duke, 1987; Blaber et al., 1989, 1995; Chong et al., 1990; Vance et al., 1996 among many others), with fewer detailed studies from subtropical (e.g. Laegdsgaard and Johnson, 1995; Halliday and Young, 1996; Lin and Shao, 1999) and temperate environments (Bell et al., 1984; Clynick and Chapman, 2002; Hindell and Jenkins, 2004). Our detailed studies in subtropical Moreton Bay confirm the presence of strong gradients in species richness and abundance of fish and decapods from tropical to temperate regions, but it is not possible to identify specific explanations for those patterns due to numerous confounding factors across the different studies (including variation in gear type, levels of surrounding development and impact on the mangroves, and species complexity of the mangroves). Two of these that deserve closer consideration though are (1) the effect of urbanisation and development on utilisation of mangroves and (2) the influence of different types of mangroves on the value of the habitat in supporting nekton.

The nekton assemblage using the mangrove habitats in western Moreton Bay was dominated by the common toadfish, *Tetractenos hamiltoni*, which often reached very large numbers and moved long distances into the forest (personal observations and Section 6 below). Although they did not dominate the catch, toadfish were also very abundant in mangrove forests in eastern Moreton Bay. Toadfish usually moved from subtidal habitats into the mangroves at the very edge of the tidal front so were among the first large animals to enter the forest on the rising tide and were often the last to depart as the tide receded. They are active predators in mangroves (Bell et al., 1984; Hughes, 1984; Warren, 1990; Roach, 1998) and their stomachs are usually full of small snails and crabs that are common in this habitat (Skilleter, unpublished data). *Acanthopagrus australis*, yellowfin bream, were also relatively abundant within the mangrove forests (see also Halliday and Young, 1996) and have also been implicated in controlling the abundance of snails within mangrove systems (e.g. temperate Sydney – Bell et al., 1984; Roach, 1998). Both these species may be important in transferring energy from the mangrove forests to subtidal habitats, via their predation on macrobenthic organisms (Sheaves and Molony, 2000). Other top-level predators in tropical mangroves, such as *Platycephalus fuscus* (Sheaves and Molony, 2000), were rare in the subtropical forests of Moreton Bay, although this may be a function of the specific sampling method (fyke nets) we used. Baker and Sheaves (2006) found that the abundance of large piscivores such as *P. fuscus* were routinely under-estimated in surveys using standard methods such as netting.

The greasyback prawn, *Metapenaeus bennettae*, was the most abundant, commercially important, decapod crustacean sampled from the mangroves, but this species was even more abundant in the downshore intertidal seagrass beds, during spring and summer. *M. bennettae* may take advantage of the substantial protection and cover provided by the vegetation within the mangroves and seagrass, especially as small juveniles (Laegdsgaard and Johnson, 1995; Manson et al., 2005b), moving between the two habitats on the rising tide, as has been suggested for other species of prawns (Robertson, 1988; Vance et al., 1996; Bishop and Khan, 1999). *M. bennettae* was rarely sampled from unvegetated mudflat habitats, using a seine net, and are also are rare in benthic core samples from these habitats (Skilleter, unpublished data). Two other species of prawns, the eastern king prawn, *Penaeus plebejus*, and the brown tiger prawn, *Penaeus esculentus*, were also relatively abundant, but only for a restricted period of time and either only in the seagrass (*P. plebejus*) or in seagrass and mudflat habitats (*P. esculentus*). Both these species were only rarely caught in the mangroves. More detail on the distribution and abundance of these prawns is found in Section 4 below (and Skilleter et al., 2005).

The overall abundance of fish using the mangroves in eastern Moreton Bay was heavily dominated by numbers of two species, the estuary perchlet, *Ambassis marianus*, and Ogilby's hardyhead, *Atherinomorus ogilbyi*, both schooling planktivores (Prince et al., 1982; Halliday and Young, 1996; Miller and Skilleter, 2005). They are often found associated with structure in seagrass and mangroves (e.g. Burchmore et al., 1984), but feed in the open water. They also move long distances into the intertidal zone, accessing saltmarsh areas (Thomas and Connolly, 2001) where they feed extensively on crab zoea (Hollingsworth and Connolly, 2006; Mazumder et al., 2006). There are few detailed descriptions of the abundance of larger zooplankton in Moreton Bay (Greenwood, 1998), so it is not clear whether there is a gradient in prey abundance across the embayment. Furthermore, the dynamics and interactions between populations of planktivorous fishes and their zooplankton prey are complex (Morgan, 1990) so there may not be any simple relationships over large spatial scales. More detailed studies are needed on the trophic resources available to these fishes, especially in relation to the distribution and abundance of the habitats in which they are found.

The use of methods such as fyke nets for sampling in mangroves has been criticised as not being quantitative (Sheridan and Hays, 2003; Smith and Hindell, 2005) based on the claim that the data are not presented as area-based density estimates. This is erroneous. There is no implicit or explicit requirement that quantitative measures are area-based. Catches from fyke nets provide quantitative estimates of abundance based on catch-per-unit-effort, usually expressed as soak time for the net. Comparisons among sites, times, treatments, etc. are valid as long as catch data are standardised for variable soak times and any sampling biases are consistent across these factors (Peterson and Black, 1994). Most methods of sampling, including those considered to be quantitative by Sheridan and Hays (2003), such as visual censuses, plankton nets and block nets, have associated biases, especially in complex habitats such as mangroves. For example, the effectiveness (catch efficiency) of visual censuses varies considerably with changes in water clarity, area of transect (or area of patch surveyed and observer experience (Sale and Sharp, 1983; Fowler, 1987; McCormick and Choat, 1987; Samoilys, 1991), but also fish behaviour (Kulbicki, 1998). Breen and Ruetz (2006) examined gear bias for fyke nets and found that variation in soak time had the largest effect on catches. They recommended standardising catches by soak time where variation among samples was less than 1 day. Variation in soak times for our catches was usually less than 60 minutes, so biases from this source are likely to be minimal. Butcher et al. (2005) specifically compared the efficiency of various gear types (ring, fyke and fence nets and beam trawls) for sampling two species that commonly occur in subtropical mangrove forests, *Sillago ciliata* (sand whiting) and *Platycephalus fuscus* (dusky flathead). They found that fyke (and ring) nets provided a 95 % effectiveness at capturing these species, compared with a method such as beam trawling.

Fyke nets (and similar gear types such as block nets: Thayer et al., 1987) are useful for sampling in highly structured environments, such as mangrove forests, because they can be deployed in and around features such as trees and prop-roots, without the need for removal or damage to the vegetation (Rozas and Minello, 1997). They have been widely used in many different environments, including freshwater lakes (Weaver et al., 1993), seagrass beds (Joseph et al., 2006) and saltmarsh systems (Crinall and Hindell, 2004; Hampel et al., 2004). Despite the capacity for use of these gears to sample within the mangrove forest, many previous studies have examined the use of the mangroves by nekton based on sampling in adjacent areas such as the pneumatophore zone or unvegetated mudflats (Laegdsgaard and Johnson, 1995; Primavera, 1998; Hindell and Jenkins, 2004) or in tidal creeks draining the forest area (Bell et al., 1984; Blaber et al., 1995; Robertson and Duke, 1987; Laegdsgaard and Johnson, 1995; Kuo et al, 1999), rather than sampling within the actual habitat. Similar issues have been identified for work on fish utilising saltmarshes (Connolly, 1999). There are limitations associated with the interpretation of data from such studies because it is not possible to determine the numbers and types of nekton that are actually using the forest habitat. Rozas and Minello (1997) discussed some of the issues associated with sampling nekton in shallow water estuarine habitats, including issues with the interpretation of data based on sampling adjacent areas, outside the line of shoreline vegetation. Clearly, knowledge about the interactions between nekton and critical estuarine habitats, such as mangroves and seagrass, requires sampling done within those habitats.

Fyke nets proved to be extremely effective at capturing nekton from within the mangrove forests of subtropical Moreton Bay, with large numbers of individuals of large and small fish and decapods occurring in the catches. These catches provided unambiguous data on what species were moving into the forest at different times of the year and the variation in utilisation of the mangroves at different spatial scales (see also Clynick and Chapman, 2002). From the few studies where sampling has been done specifically in the forest, it is clear that many species venture into the structurally complex habitat on the rising tide (Thayer et al., 1987; Morton, 1990; Vance et al, 1996) so it is important that sampling be focussed inside the forest itself.

The composition of the nekton assemblage using the mangroves and adjacent patches of intertidal seagrass showed very little overlap between the two habitats, for landscapes on either side of Moreton Bay, consistent with previous work by Laegdsgaard and Johnson (1995). Our analyses were based on presence-absence data to allow comparisons between the two different sampling methods that were used in the different habitats, but still provide a strong indication that a different suite of animals is using the mangroves from that using the adjacent, downshore seagrass beds, during periods when both habitats are accessible (see also Morton, 1990). During high tide, fish using seagrass beds move upshore into mangroves to feed (Thayer et al., 1987; Robertson and Duke, 1990; Vance et al., 1996), although the extent to which this occurs depends on the proximity between the two habitats (Nagelkerken and van der Velde, 2004b and see Sections 4 and 5- this report). Our data suggest that as the nekton move into the intertidal zone during high tide, some species stay within the seagrass beds, but other species move further upshore into the mangroves, either for shelter or to feed (Manson et al., 2005b). What these data do not address is the extent to which utilisation of the mangroves varies as a function of variation in the quality of the seagrass corridor that the nekton cross before reaching the mangroves. This question is addressed in Section 6 of this report.

Even though the composition of the nekton assemblage in the seagrass beds and mangrove forests were quite distinct from each other, there was overlap between the species that used the mangroves and those found in the mudflat. Laegdsgaard and Johnson (1995) obtained similar results and suggested that there was separation of the assemblage between mangroves and the mudflats based on size of individual species. A notable result in that study, which had a large influence on their conclusions about the relative importance of mangroves compared with other habitats, was the fact that they found 27 species of fish exclusive to the mangroves but only 4 species of fish were exclusive to seagrass. In marked contrast, we found 20 species exclusive to the mangroves, 14 species exclusive to the seagrass and 3 species exclusive to the mudflats (Table 1.19). Thus, we found fewer species using only the mangroves and many more species using only the seagrass. The total number of species of fish was similar in both studies (56 in the earlier study, 65 in ours). The seine net used by Laegdsgaard and Johnson (1995) was similar in size to the one used in the present study, although the length of tow was slightly shorter in the earlier work. It is unlikely that this alone could account for the different results. The two studies were done 13 years apart, so clearly considerable changes in the environment within Moreton Bay are likely to have occurred, but the specific changes that could account for the different results cannot easily be identified. The specific methods used to sample the mangroves and the actual habitat sampled also varied between the two studies though.

Laegdsgaard and Johnson (1995) used trap nets deployed in small tidal creeks draining the mangroves, compared with fyke nets deployed within the main structure of the forest in our study. The specific design of their trap nets was not described though. Trap nets used in freshwater fisheries in the USA are quite similar in design and work in a similar manner to our fyke nets. Trap nets used in Australian penaeid fisheries (for example) are very different from fyke nets and work in a different way (e.g. Broadhurst et al., 2004). If the nets used by Laegdsgaard and Johnson (1995) were similar to those used in Australian penaeid fisheries, then this could account for the different results. If their trap nets were more like those used in USA freshwater fisheries, the gear differences seem unlikely to explain the differences in utilisation of the different habitats.

The deployment within creeks rather than in the actual mangrove forest could, however, account for the differences in patterns of species utilisation of the mangroves over the 13 year period. The water in tidal creeks crossing the mangroves is deeper and often contains less structure (pneumatophores, trees, etc) than the surrounding sediments. These creeks provide a corridor into the mangroves for larger species of nekton, but also those that have an association with unstructured habitats such as mudflats. The species caught using creeks may not be the same as those moving across the main forest floor (see Connolly, 1999 for a similar argument in relation to saltmarshes). This form of corridor is also very different from one where fish entered into the forest across the broad expanse of flat, shallower sediments. The effects of corridor type on the composition of the assemblage using mangroves is specifically examined in Section 6 of this report.

Analyses on data from either side of Moreton Bay, were dominated by considerable variation among the different sites selected to represent the major habitats (seagrass, mangroves and mudflats) and a lack of synchrony among the sites in the fluctuations in abundance through the year. That is, the nekton assemblage using the different habitats at any site was often quite different from that in nearby sites and the populations were fluctuating through time in different ways. Selection of sites was done to ensure as much as possible that the broad physical characteristics (overall coverage of seagrass, species composition of mangroves within the forest, etcetera) of the target habitat at each site were similar to each other. The selection of sites used a fairly typical set of criteria, including presence/absence of channels, relative height on the shore and density of vegetation (e.g. Laegdsgaard and Johnson, 1995; Hindell and Jenkins, 2004). Such variability among sites, even where sites are only separated by small distances, has been a consistent feature of descriptive studies examining utilisation of estuarine habitats by nekton (Robertson and Duke, 1987; Loneragan and Potter, 1990; Skilleter et al., 2005). Laegdsgaard and Johnson (1995) suggested this may be due to differences in advection of larvae to different sites (see also Stockhausen and Lipcius, 2003), but an alternative explanation lies with variation in the composition of the landscapes in which the individual patches of habitat are found. The proximity between different types of habitat (Sections 4 and 5; Skilleter et al., 2005) and the nature of the corridors through estuarine landscapes (Micheli and Peterson, 1999; Section 6) have been shown to affect the dynamics of nekton using specific habitats, but rarely are such factors taken into consideration when designing surveys to document utilisation of estuarine habitats (Laegdsgaard and Johnson, 1995; Hindell and Jenkins, 2004; Smith and Hindell, 2005; but see Irlandi and Crawford, 1997; Raposa and Oviatt, 2000). Consideration of the influence of landscape characteristics on the utilisation of different estuarine habitats within subtropical Moreton Bay formed the primary basis of the remainder of this study, as described below.

Table 1.1: List of sites used for sampling epibenthic finfish and crustaceans for each of the five habitat types sampled in Spring 2002. Position within Moreton Bay of each site is shown in Figure 1.1.

HABITAT TYPE	Western Moreton Bay	Eastern Moreton Bay
Mangroves (*Avicennia*)	Fisherman Islands North	Myora Springs 1
	Fisherman Islands South	One Mile
	Thorneside	Wallen Pools
Seagrass (*Zostera*) - intertidal	Fisherman Islands North	Myora Springs 1
	Fisherman Islands South	One Mile
	Thorneside	Wallen Pools South
Unvegetated - intertidal	Nudgee Beach North	Chiggill 1
	Nudgee Beach South	Chiggill 2
	Lota	Amity Point
Seagrass (*Zostera*) - subtidal	Fisherman Islands North	Myora Springs 1
	Fisherman Islands South	Myora Springs 2
	Thorneside	Wallen Pools South
Unvegetated - subtidal	Nudgee Beach North	Wallen Pools South
	Nudgee Beach South	Chiggill 3
	Lota	Chiggill 4

Site		**Habitat**			
	Mangroves	**Seagrass**		**Unvegetated**	
		Intertidal	**Subtidal**	**Intertidal**	**Subtidal**
Western Bay					
Fisherman Islands North	✓	✓	✓		
Fisherman Islands South	✓	✓	✓		
Thorneside	✓	✓	✓		
Nudgee Beach North				✓	✓
Nudgee Beach South				✓	✓
Lota				✓	✓
Eastern Bay					
One Mile	✓	✓			
Wallen Pools	✓	✓			
Myora Springs 1	✓		✓		
Myora Springs 2			✓		
Wallen Pools South		✓	✓		✓
Chiggill 1				✓	
Chiggill 2				✓	
Chiggill 3					✓
Chiggill 4					✓
Amity Point				✓	

Table 1.2: List of species sampled from the mangroves in western Moreton Bay in spring 2002. The total number of individuals for each species is shown, for each site separately and all sites combined. FI-North = Fisherman Islands North, FI-South = Fisherman Islands South and Thorn = Thorneside (see Figure 1.1).

Species	Spring			
	Mangroves			
	FI-North	FI-South	Thorn	Total
Acanthopagrus australis	122	250	15	387
Ambassis marianus	0	35	7	42
Atherinomorus ogilbyi	17	34	7	58
Gerres subfasciatus	3	9	2	14
Girella tricuspidata	0	0	1	1
Gobiidae	7	17	0	24
Hippichthys pencillus	0	3	0	3
Hyporhampus regularis ardelio	0	0	2	2
Liza argentea	2	221	21	244
Marilyna pleurosticta	0	0	7	7
Microcanthus strigatus	0	0	1	1
Mugil cephalus	0	3	0	3
Mugilidae	0	95	0	95
Mugilogobius stigmaticus	8	223	15	246
Omobranchus verticalis	1	3	1	5
Paramonacanthus otiensis	0	1	0	1
Pelates sexilineatus	1	11	0	12
Petroscirtes lupus	1	0	0	1
Pseudomugil signifer	1	1	0	2
Rhabdosargus sarba	4	6	4	14
Sillago analis	0	0	4	4
Sillago ciliata	0	16	4	20
Sillago maculata	0	0	2	2
Sphyraena obtusata	9	20	11	40
Sygnathidae	0	2	0	2
Tetractenos hamiltoni	273	1171	414	1858
Thryssa aesturia	4	1	14	19
Tylosurus gavialoides	0	1	1	2
Valamugil georgii	2	6	30	38
Leander tenuicornis	4	0	0	4
Metapenaeus bennettae	2	543	103	648
Metapenaeus ensis	2	341	69	412
Mysidacea	2	0	10	12
Palaemon debilis	149	239	27	415
Palaemon serenus	44	0	0	44
Penaeidae	0	47	7	54
Penaeus plebejus	2	5	7	14
Periclimenes andamanensis	1	0	0	1
Periclimenes obscurus	23	0	0	23
Scylla serrata	10	7	2	19

Idiosepius notoides	2	1	0	3
Number of individuals =				4796
Number of Species =				41
Number Fish Species =				29
Number of Invert Species =				12

Table 1.3: List of species sampled from the intertidal seagrass (*Zostera capricorni*) in western Moreton Bay in spring 2002. The total number of individuals for each species is shown, for each site separately and all sites combined. FI-North = Fisherman Islands North, FI-South = Fisherman Islands South and Thorn = Thorneside (see Figure 1.1).

	FI-North	FI-South	Thorn	Total
Acanthopagrus australis	6	0	28	34
Centropogon australis	49	0	268	317
Gerres subfasciatus	0	2	0	2
Gobiidae	302	0	1441	1743
Hippocampus whitei	0	1	0	1
Omobranchus verticalis	0	0	3	3
Paramonacanthus otiensis	0	1	3	4
Pelates quadrilineatus	0	11	3	14
Pelates sexilineatus	20	2	226	248
Petroscirtes lupus	0	1	5	6
Sphyraena obtusata	0	7	0	7
Stigmatopora nigra	0	1	0	1
Suggrundus jugosus	0	0	1	1
Sygnathidae	0	1	0	1
Tetractenos hamiltoni	8	1	46	55
Thryssa aesturia	0	0	2	2
Urocampus carinirostris	29	0	118	147
Latreutes porcinus	1	0	220	221
Latreutes pygmaeus	0	0	183	183
Metapenaeus bennettae	58	0	196	254
Metapenaeus ensis	62	0	28	90
Palaemon debilis	0	1	101	102
Palaemon serenus	0	0	10	10
Penaeidae	0	7	0	7
Penaeus esculentus	0	0	18	18
Penaeus plebejus	31	0	289	320
Periclimenes andamanensis	3	0	0	3
Periclimenes holthuisi	0	6	0	6
Periclimenes obscurus	0	1	113	114
Portunus pelagicus	2	0	0	2
Idiosepius notoides	16	4	47	67
Number of individuals =				3983
Number of Species =				31
Number Fish Species =				17
Number of Invert Species =				14

Table 1.4: List of species sampled from the intertidal mudflats (unvegetated) in western Moreton Bay in spring 2002. The total number of individuals for each species is shown, for each site separately and all sites combined. FI-North = Fisherman Islands North, FI-South = Fisherman Islands South and Thorn = Thorneside (see Figure 1.1).

Species	Spring			
	Intertidal Seagrass			
	Nudgee North	Nudgee South	Lota	Total
Centropogon australis	0	0	1	1
Dasyatus fluvorium	0	1	0	1
Hyperlophus sp.	0	52	1	53
Marilyna pleurosticta	0	12	0	12
Paramonacanthus otiensis	0	0	1	1
Pelates sexilineatus	0	0	7	7
Petroscirtes lupus	0	0	1	1
Sillago maculata	0	1	0	1
Sphyraena obtusata	0	0	3	3
Tetractenos hamiltoni	1	5	1	7
Thryssa aesturia	0	1	0	1
Carid sp A	0	0	1	1
Metapenaeus ensis	1	0	0	1
Metapenaeus macleayi	0	3	0	3
Penaeus esculentus	1	5	4	10
Sepioteuthis sp	3	80	20	103
Number of individuals =				206
Number of Species =				16
Number Fish Species =				11
Number of Invert Species =				5

Table 1.5: List of species sampled from subtidal seagrass (*Zostera capricorni*) in western Moreton Bay in spring 2002. The total number of individuals for each species is shown, for each site separately and all sites combined. FI-North = Fisherman Islands North, FI-South = Fisherman Islands South and Thorn = Thorneside (see Figure 1.1).

Species	Spring			
	Intertidal Seagrass			
	FI-North	FI-South	Thorn	Total
Acanthopagrus australis	0	0	3	3
Ambassis marianus	0	0	2	2
Apogon fasciatus	0	1	0	1
Centropogon australis	47	31	81	159
Dicotylichthys punctulatus	0	1	0	1
Gerres subfasciatus	0	0	1	1
Gobiidae	157	38	5	200
Hippichthys pencillus	0	1	0	1
Hippichthys whitei	0	2	0	2
Hyporhampus regularis ardelio	4	0	0	4
Microcanthus strigatus	0	1	0	1
Monacanthus chinensis	53	122	15	190
Mugilidae	5	0	0	5
Mugilogobius stigmaticus	2	0	0	2
Paramonacanthus otiensis	26	10	26	62
Pegasus volitans	0	0	1	1
Pelates sexilineatus	174	279	113	566
Petroscirtes lupus	6	4	5	15
Pseudorhombus sp	0	0	1	1
Rhabdosargus sarba	0	0	2	2
Sillago maculata	7	0	4	11
Sphyraena obtusata	1	0	2	3
Stigmatopora nigra	16	2	2	20
Suggrundus jugosus	0	1	0	1
Sygnathidae	0	1	0	1
Sygnathoides biaculeatus	0	2	0	2
Tetractenos hamiltoni	4	0	0	4
Tetractenos pleurogramma	0	1	0	1
Urocampus carinirostris	31	5	3	39
unidentifiable carid	0	27	0	27
Carid sp A	0	0	5	5
Caradina maccullochi	0	0	9	9
Chlorotocella nr gracilis	0	2	5	7
Hippolyte caradina	257	314	0	571
Hippolyte sp	0	109	0	109
Latreutes porcinus	9	3	27	39
Latreutes pygmaeus	130	46	48	224
Leander tenuicornis	1	0	23	24
Macrobrachium sp	0	1	0	1
Metapenaeus bennettae	1	0	3	4
Metapenaeus ensis	2	0	5	7

Mysidacea	3	71	23	97
Palaemonidae	0	1	0	1
Palaemon serenus	7	0	28	35
Penaeus esculentus	10	89	2	101
Penaeus plebejus	64	14	38	116
Periclimenes andamanensis	0	3	0	3
Periclimenes holthuisi	0	36	21	57
Periclimenes obscurus	375	514	77	966
Portunus pelagicus	1	3	5	9
Euprymna tasmanica	0	1	0	1
Idiosepius notoides	16	23	2	41
Sepia plangon	0	4	0	4
Sepiadarium austrinum	0	5	0	5
Number of individuals =				3764
Number of Species =				54
Number Fish Species =				29
Number of Invert Species =				25

Table 1.6: List of species sampled from mangroves in western and eastern Moreton Bay in spring 2002. The total number of individuals for each species is shown for each side of the bay and combined. Species that occurred in mangroves in western Moreton Bay but not in eastern Moreton Bay are marked as ✹; those in eastern Moreton Bay but not in western Moreton Bay are marked as ◊.

Species	Presence-Absence	WEST	EAST	TOTAL
Acanthopagrus australis		387	58	445
Ambassis marianus		42	1436	1478
Atherinomorus ogilbyi		58	1115	1173
Gerres subfasciatus		14	73	87
Girella tricuspidata	✹	1	0	1
Gobiidae		24	19	43
Hippichthys pencillus	✹	3	0	3
Hyporhampus regularis ardelio	✹	2	0	2
Liza argentea		244	129	373
Marilyna pleurosticta		7	1	8
Microcanthus strigatus		1	5	6
Mugil cephalus		3	13	16
Mugilidae		95	52	147
Mugilogobius stigmaticus		246	18	264
Omobranchus punctatus	◊	0	2	2
Omobranchus verticalis	✹	5	0	5
Paramonacanthus otiensis	✹	1	0	1
Pelates sexilineatus	✹	12	0	12
Petroscirtes lupus	✹	1	0	1
Pseudomugil signifer		2	3	5
Rhabdosargus sarba	✹	14	0	14
Sillago analis	✹	4	0	4
Sillago ciliata		20	22	42
Sillago maculata		2	3	5
Sphyraena obtusata		40	4	44
Tetractenos hamiltoni		1858	756	2614
Thryssa aesturia	✹	19	0	19
Valamugil georgii		38	3	41
Leander tenuicornis		4	0	4
Macrobrachium sp	◊	0	1	1
Metapenaeus bennettae		648	1	649
Metapenaeus ensis	✹	412	0	412
Mysidacea	✹	12	0	12
Palaemon debilis		415	2866	3281
Palaemon serenus	◊	44	68	112
Penaeidae	✹	54	0	54
Penaeus plebejus		14	3	17
Periclimenes andamanensis	✹	1	0	1
Periclimenes obscurus	✹	23	0	23
Portunus pelagicus	◊	0	6	6
Scylla serrata		19	6	25
Idiosepius notoides	◊	3	2	5

Number of individuals =		4792	6665	11457
Number of Species =		39	26	42
Number Fish Species =		27	18	28
Number of Invert Species =		12	8	14

Table 1.7: List of species sampled from intertidal seagrass in western and eastern Moreton Bay in spring 2002. The total number of individuals for each species is shown for each side of the bay and combined. Species that occurred in intertidal seagrass in western Moreton Bay but not in eastern Moreton Bay are marked as ✹; those in eastern Moreton Bay but not in western Moreton Bay are marked as ◊.

Species	Presence-Absence	WEST	EAST	TOTAL
Acanthopagrus australis		34	14	48
Atherinomorus ogilbyi	◊	0	37	37
Centropogon australis		317	103	420
Filicampus tigris	◊	0	8	8
Gerres subfasciatus	✹	2	0	2
Gobiidae		1743	1109	2852
Hippocampus sp	◊	0	1	1
Hippocampus whitei	✹	1	0	1
Paramonacanthus otiensis		4	1	5
Pelates sexilineatus		248	184	432
Petroscirtes lupus		6	9	15
Pseudorhombus sp	◊	0	1	1
Sphyraena obtusata		7	5	12
Tetractenos hamiltoni		55	16	71
Torquigener pleurogramma	◊	0	11	11
Urocampus carinirostris		147	320	467
Valamugil georgii	◊	0	2	2
Vanacampus sp	◊	0	1	1
Athanas japonicus	◊	0	1	1
Chlorotocella nr gracilis	◊	0	29	29
Latreutes porcinus		221	5	226
Latreutes pygmaeus		183	1	184
Palaemon debilis		102	1	103
Palaemon serenus		10	3	13
Penaeidae		7	8	15
Penaeus esculentus		18	1	19
Penaeus plebejus		320	2055	2375
Periclimenes andamanensis		3	40	43
Periclimenes holthuisi		6	1	7
Periclimenes obscurus		114	4	118
Pontophilus sp 1	◊	0	1	1
Portunus pelagicus		2	9	11
Idiosepius notoides		67	69	136
Number of individuals =		3617	4050	7667
Number of Species =		23	31	33
Number Fish Species =		11	16	18
Number of Invert Species =		12	15	15

Table 1.8: List of species sampled from intertidal mudflats in western and eastern Moreton Bay in spring 2002. The total number of individuals for each species is shown for each side of the bay and combined. Species that occurred in intertidal mudflats in western Moreton Bay but not in eastern Moreton Bay are marked as ✹; those in eastern Moreton Bay but not in western Moreton Bay are marked as ◊.

Species	Presence-Absence	WEST	EAST	TOTAL
Acanthopagrus australis	◊	0	4	4
Atherinomorus ogilbyi	◊	0	11	11
Centropogon australis		1	1	2
Dasyatus fluvorium	✹	1	0	1
Gobiidae	◊	0	12	12
Hippocampus sp.	◊	0	1	1
Hyperlophus sp.	✹	53	0	53
Marilyna pleurosticta	✹	12	0	12
Microcanthus strigatus	◊	0	2	2
Paramonacanthus otiensis	✹	1	0	1
Pelates sexilineatus		7	26	33
Petroscirtes lupus		1	18	19
Sillago maculata	✹	1	0	1
Sphyraena obtusata		3	2	5
Stigmatopora nigra	◊	0	2	2
Tetractenos hamiltoni		7	10	17
Thryssa aesturia	✹	1	0	1
Torquigener pleurogramma	◊	0	5	5
Urocampus carinirostris	◊	0	2	2
Fish species 11	◊	0	1	1
Caradina maccullochi	✹	1	0	1
Chlorotocella nr gracilis	◊	0	93	93
Latreutes pygmaeus	◊	0	1	1
Metapenaeus macleayi	✹	1	0	1
Mysidacea	✹	3	0	3
Palaemon serenus	◊	0	1	1
Portunus plebejus		10	189	199
Idiosepius notoides	◊	0	16	16
Number of individuals =		103	397	500
Number of Species =		15	19	28
Number Fish Species =		11	14	20
Number of Invert Species =		4	5	8

Table 1.9: List of species sampled from subtidal seagrass in western and eastern Moreton Bay in spring 2002. The total number of individuals for each species is shown for each side of the bay and combined. Species that occurred in subtidal seagrass in western Moreton Bay but not in eastern Moreton Bay are marked as ✹; those in eastern Moreton Bay but not in western Moreton Bay are marked as ◊.

Species	Presence-Absence	WEST	EAST	TOTAL
Acanthopagrus australis	✹	3	0	3
Ambassis marianus	✹	2	0	2
Apogon fasciatus		1	1	2
Callinomyus macdonaldi	✹	2	0	2
Centropogon australis		159	27	186
Cyclichthys jaculiferus	✹	4	0	4
Dicotylichthys punctulatus	✹	1	0	1
Filicampus tigris	◊	0	2	2
Gerres subfasciatus		1	1	2
Gobiidae		200	239	439
Hippichthys pencillus	✹	1	0	1
Hippocampus whitei		2	1	3
Hyporhampus regularis ardelio	✹	4	0	4
Lutjanus fulvifamma	◊	0	1	1
Microcanthus strigatus	✹	1	0	1
Monacanthus chinensis		190	34	224
Mugilidae		5	3	8
Mugilogobius stigmaticus		2	2	4
Paramonacanthus otiensis		62	4	66
Pegasus volitans	✹	1	0	1
Pelates sexilineatus		566	37	603
Petroscirtes lupus		15	16	31
Pseudorhombus jenysii	◊	0	1	1
Pseudorhombus sp	✹	1	0	1
Rhabdosargus sarba	✹	2	0	2
Scaridae	◊	0	1	1
Sillago maculata	✹	11	0	11
Sphyraena obtusata		3	29	32
Stigmatopora nigra		20	7	27
Suggrundus jugosus	✹	1	0	1
Sygnathidae	✹	1	0	1
Sygnathoides biaculeatus		2	1	3
Tetractenos hamiltoni	✹	4	0	4
Torquigener pleurogramma	✹	1	0	1
Urocampus carinirostris		39	2	41
Vanacampus sp	◊	0	1	1
Athanas japonicus	◊	0	11	11
Atyidae sp	✹	27	0	27
unidentifiable carid	✹	5	0	5
Carid sp A	✹	9	0	9

Caradina maccullochi		7	30	37
Chlorotocella nr gracilis		571	235	806
Hippolyte caradina		109	9	118
Latreutes compressus		39	32	71
Latreutes porcinus		224	280	504
Latreutes pygmaeus		24	23	47
Leander tenuicornis	✸	1	0	1
Macrobrachium sp	✸	4	0	4
Metapenaeus bennettae	✸	7	0	7
Metapenaeus macleayi		97	401	498
Mysidacea	✸	1	0	1
Palaemonidae	◊	0	84	84
Palaemon debilis		35	1	36
Penaeidae	✸	101	0	101
Penaeus esculentus		116	485	601
Penaeus plebejus		3	160	163
Periclimenes andamanensis	✸	57	0	57
Periclimenes holthuisi		966	4	970
Pontophilus sp 3		9	15	24
Scylla serrata	◊	0	6	6
Thalamita sp	✸	1	0	1
Euprymna tasmanica	◊	0	1	1
Hapalochlaena sp		41	77	118
Photololigo etheridgei		4	1	5
Sepia plangon	✸	5	0	5
Sepioteuthis sp	✸	3770	0	3770
Number of individuals =		7540	2265	9805
Number of Species =		57	38	66
Number Fish Species =		31	20	36
Number of Invert Species =		26	18	30

Table 1.10: List of species sampled from subtidal unvegetated sediments in western and eastern Moreton Bay in spring 2002. The total number of individuals for each species is shown for each side of the bay and combined. Species that occurred in subtidal seagrass in western Moreton Bay but not in eastern Moreton Bay are marked as ✸; those in eastern Moreton Bay but not in western Moreton Bay are marked as ◊.

Species	Presence-Absence	WEST	EAST	TOTAL
Atherinomorus ogilbyi	✸	15	0	15
Centropogon australis		0	1	1
Gobiidae	✸	283	0	283
Microcanthus strigatus		1	1	2
Mugilidae		1	1	2
Pelates sexilineatus		8	7	15
Petroscirtes lupus	✸	2	0	2
Pseudorhombus arsius	✸	1	0	1
Sillago ciliata	◊	0	2	2
Sphyraena obtusata	✸	1	0	1
Stigmatopora nigra	✸	7	0	7
Tetractenos hamiltoni		7	3	10
Thryssa aesturia	◊	0	34	34
Torquigener pleurogramma	✸	1	0	1
Torquigener squamicauda	✸	5	0	5
Urocampus carinirostris	✸	2	0	2
Valamugil georgii	◊	0	1	1
Fish sp. 11	✸	22	0	22
Chlorotocella nr *gracilis*	✸	85	0	85
Hippolyte sp.	✸	11	0	11
Mysidacea		36	21	57
Palaemonidae	◊	0	4	4
Penaeus plebejus		201	23	224
Pontophilus sp. 1	✸	76	0	76
Pontophilus sp. 3	✸	45	0	45
Portunus pelagicus	✸	2	0	2
Number of individuals =		812	98	910
Number of Species =		21	11	26
Number Fish Species =		14	8	18
Number of Invert Species =		7	3	8

Table 1.11: Results of one factor ANOSIM analyses based on a dummy categorical variable created for the interaction between the main effects of Season and Site. Values are for the R-statistic of pairwise tests for the relevant comparisons between consecutive seasons for each site. A larger value for the R-statistic indicates greater separation of those two seasons compared with the other comparisons within the sequence. The comparison of the two consecutive seasons that show the least amount of change in composition at each site is shaded in the table. R-values are directly comparable within any set, that is, among successive seasons and among sites.

(A) Western Moreton Bay - Mangroves	Spring vs Summ	Summ vs Aut	Aut vs Win
Fisherman Islands North	0.68	0.94	0.94
Fisherman Islands South	0.65	0.51	0.80
Thorneside	0.41	0.02	0.68

(B) Eastern Moreton Bay - Mangroves	Spring vs Summ	Summ vs Aut	Aut vs Win
Myora Springs	0.05	0.49	0.96
One Mile	0.49	0.03	0.42
Wallen Pools South	0.64	0.83	0.64

(C) Western Moreton Bay – Seagrass	Spring vs Summ	Summ vs Aut	Aut vs Win
Fisherman Islands North	0.23	0.60	0.28
Fisherman Islands South	0.51	0.33	0.33
Thorneside	0.94	0.95	0.40

(D) Eastern Moreton Bay – Seagrass	Spring vs Summ	Summ vs Aut	Aut vs Win
Myora Springs	0.27	0.54	0.33
One Mile	0.87	0.66	0.97
Wallen Pools South	0.26	0.31	0.15

(E) Western Moreton Bay – Mudflat	Spring vs Summ	Summ vs Aut	Aut vs Win
Nudgee North	0.96	0.66	0.28
Nudgee South	0.97	0.76	0.28
Lota	0.53	0.05	0.00

(F) Eastern Moreton Bay – Mudflat	Spring vs Summ	Summ vs Aut	Aut vs Win
Chiggil 1	0.70	0.35	0.03
Chiggil 2	0.19	0.59	0.32
Amity Point	0.92	0.24	0.31

Table 1.12: Results of SIMPER analyses showing those species contributing at least 5 % to the separation of the nekton assemblages between successive seasons in mangroves in western Moreton Bay. The % contribution to the overall separation of the assemblage and the average abundance in each of the seasons being compared is shown. Species that are unlikely to provide useful discrimination between the seasons (i.e. where there is relatively large average dissimilarity between groupings but also a large standard deviation of the dissimilarity between pairs of samples from the two groupings, Clarke, 1993) are shown as shaded cells in the table.

Spring versus Summer	% Contribution	Spring $\overline{X}$ Abundance	Summer $\overline{X}$ Abundance
Tetractenos hamiltoni	23.88	95.25	87.99
Palaemon debilis	13.62	22.65	58.42
Palaemon serenus	9.67	2.09	45.22
Metapenaeus bennettae	8.60	33.37	14.01
Mysidacea	7.61	0.42	97.26
Metapenaeus ensis	5.75	21.13	11.63
Atherinomorus ogilbyi	5.56	2.51	15.10

Summer versus Autumn	% Contribution	Summer $\overline{X}$ Abundance	Autumn $\overline{X}$ Abundance
Tetractenos hamiltoni	24.10	87.99	75.65
Palaemon debilis	12.44	58.42	0.00
Palaemon serenus	9.39	45.22	0.13
Palaemonidae	9.08	0.38	31.84
Mysidacea	7.07	97.26	0.00
Liza argentea	5.81	0.24	24.61
Atherinomorus ogilbyi	5.39	15.10	0.61
Metapenaeus bennettae	5.09	14.01	0.00

Autumn versus Winter	% Contribution	Autumn $\overline{X}$ Abundance	Winter $\overline{X}$ Abundance
Palaemon debilis	25.37	0.00	71.28
Tetractenos hamiltoni	23.19	75.65	8.25
Liza argentea	11.34	24.61	16.87
Palaemonidae	10.66	31.84	0.00
Sillago analis	6.37	5.34	13.83

Table 1.13: Results of SIMPER analyses showing those species contributing at least 5 % to the separation of the nekton assemblages between successive seasons in mangroves in eastern Moreton Bay. The % contribution to the overall separation of the assemblage and the average abundance in each of the seasons being compared is shown. Species that are unlikely to provide useful discrimination between the seasons (i.e. where there is relatively large average dissimilarity between groupings but also a large standard deviation of the dissimilarity between pairs of samples from the two groupings, Clarke, 1993) are shown as shaded cells in the table.

Spring versus Summer	% Contribution	Spring $\overline{X}$ Abundance	Summer $\overline{X}$ Abundance
Ambassis marianus	39.8	59.4	505.1
Palaemon debilis	34.4	119.4	384.7
Atherinomorus ogilbyi	7.5	53.0	5.3

Summer versus Autumn	% Contribution	Summer $\overline{X}$ Abundance	Autumn $\overline{X}$ Abundance
Ambassis marianus	43.6	505.1	155.4
Palaemon debilis	40.3	384.7	226.8

Autumn versus Winter	% Contribution	Autumn $\overline{X}$ Abundance	Winter $\overline{X}$ Abundance
Palaemon debilis	52.7	226.8	616.1
Ambassis marianus	20.9	155.4	140.8
Atherinomorus ogilbyi	14.3	1.0	118.2

Table 1.14: Results of SIMPER analyses showing those species contributing at least 5 % to the separation of the nekton assemblages between successive seasons in intertidal seagrass in western Moreton Bay. The % contribution to the overall separation of the assemblage and the average abundance in each of the seasons being compared is shown. Species that are unlikely to provide useful discrimination between the seasons (i.e. where there is relatively large average dissimilarity between groupings but also a large standard deviation of the dissimilarity between pairs of samples from the two groupings, Clarke, 1993) are shown as shaded cells in the table.

Spring versus Summer	% Contribution	Spring $\overline{X}$ Abundance	Summer $\overline{X}$ Abundance
Gobiidae	27.2	145.3	24.4
Palaemon serenus	17.6	0.8	58.3
Metapenaeus bennettae	8.4	21.2	16.8
Penaeus plebejus	5.9	26.7	4.0
Pelates sexlineatus	5.4	20.7	8.8
Centropogon australis	5.0	26.4	3.4

Summer versus Autumn	% Contribution	Summer $\overline{X}$ Abundance	Autumn $\overline{X}$ Abundance
Periclimenes obscurus	26.7	0.0	241.1
Palaemon serenus	16.0	58.3	83.1
Metapenaeus bennettae	10.3	16.8	52.2
Gobiidae	10.0	24.4	52.3
Palaemon debilis	7.6	1.5	60.7

Autumn versus Winter	% Contribution	Autumn $\overline{X}$ Abundance	Winter $\overline{X}$ Abundance
Periclimenes obscurus	36.1	241.1	201.3
Gobiidae	11.2	52.3	89.4
Palaemon serenus	9.7	83.1	18.1
Metapenaeus bennettae	7.8	52.2	57.7
Palaemon debilis	7.1	60.7	1.3
Pelates sexlineatus	6.5	35.6	42.2
Penaeus plebejus	5.6	14.1	39.2

Table 1.15: Results of SIMPER analyses showing those species contributing at least 5 % to the separation of the nekton assemblages between successive seasons in intertidal seagrass in eastern Moreton Bay. The % contribution to the overall separation of the assemblage and the average abundance in each of the seasons being compared is shown. Species that are unlikely to provide useful discrimination between the seasons (i.e. where there is relatively large average dissimilarity between groupings but also a large standard deviation of the dissimilarity between pairs of samples from the two groupings, Clarke, 1993) are shown as shaded cells in the table.

Spring versus Summer	% Contribution	Spring $\overline{X}$ Abundance	Summer $\overline{X}$ Abundance
Penaeus plebejus	36.8	171.3	17.2
Gobiidae	20.0	92.4	26.0
Periclimenes obscurus	14.8	0.3	146.4
Urocampus carinirostris	8.8	26.7	1.8

Summer versus Autumn	% Contribution	Summer $\overline{X}$ Abundance	Autumn $\overline{X}$ Abundance
Periclimenes obscurus	38.5	146.4	274.2
Penaeus plebejus	26.2	17.2	132.9
Gobiidae	17.2	26.0	98.1

Autumn versus Winter	% Contribution	Autumn $\overline{X}$ Abundance	Winter $\overline{X}$ Abundance
Periclimenes obscurus	42.6	274.2	8.0
Penaeus plebejus	21.7	132.9	131.9
Gobiidae	16.4	98.1	69.8

Table 1.16: Results of SIMPER analyses showing those species contributing at least 5 % to the separation of the nekton assemblages between successive seasons in intertidal mudflats in western Moreton Bay. The % contribution to the overall separation of the assemblage and the average abundance in each of the seasons being compared is shown. Species that are unlikely to provide useful discrimination between the seasons (i.e. where there is relatively large average dissimilarity between groupings but also a large standard deviation of the dissimilarity between pairs of samples from the two groupings, Clarke, 1993) are shown as shaded cells in the table.

Spring versus Summer	% Contribution	Spring $\overline{X}$ Abundance	Summer $\overline{X}$ Abundance
Mysids	38.9	0.4	30.8
Hyperlophus sp.	19.5	7.6	0.0
Penaeus plebejus	10.7	1.4	6.4
Metapenaeus bennettae	8.4	0.0	17.3

Summer versus Autumn	% Contribution	Summer $\overline{X}$ Abundance	Autumn $\overline{X}$ Abundance
Mysids	35.2	30.8	0.1
Sillago maculata	23.9	0.5	14.3
Penaeus plebejus	11.7	6.4	4.3
Metapenaeus bennettae	11.2	17.3	2.8

Autumn versus Winter	% Contribution	Autumn $\overline{X}$ Abundance	Winter $\overline{X}$ Abundance
Penaeus plebejus	37.7	4.3	17.6
Sillago maculata	30.7	14.3	0.5
Gobies	8.4	1.1	3.4
Tetractenos hamiltoni	7.5	0.6	1.6

Table 1.17: Results of SIMPER analyses showing those species contributing at least 5 % to the separation of the nekton assemblages between successive seasons in intertidal mudflats in eastern Moreton Bay. The % contribution to the overall separation of the assemblage and the average abundance in each of the seasons being compared is shown. Species that are unlikely to provide useful discrimination between the seasons (i.e. where there is relatively large average dissimilarity between groupings but also a large standard deviation of the dissimilarity between pairs of samples from the two groupings, Clarke, 1993) are shown as shaded cells in the table.

Spring versus Summer	% Contribution	Spring $\overline{X}$ Abundance	Summer $\overline{X}$ Abundance
Penaeus plebejus	43.6	17.2	9.1
Chlorotocella nr. *gracilis*	21.7	8.5	5.8
Ambassis marianus	6.6	0.0	15.1
Pelates sexlineatus	6.2	2.4	0.0
Atherinomorus ogilbyi	5.5	1.0	0.0

Summer versus Autumn	% Contribution	Summer $\overline{X}$ Abundance	Autumn $\overline{X}$ Abundance
Penaeus plebejus	33.8	9.1	17.2
Ambassis marianus	18.1	15.1	7.4
Chlorotocella nr. *gracilis*	17.8	5.8	2.6
Atherinomorus ogilbyi	12.8	0.0	3.9

Autumn versus Winter	% Contribution	Autumn $\overline{X}$ Abundance	Winter $\overline{X}$ Abundance
Penaeus plebejus	26.0	17.2	6.3
Ambassis marianus	23.7	7.4	19.1
Gobies	17.3	6.4	5.3
Atherinomorus ogilbyi	11.2	3.9	0.3
Chlorotocella nr. *gracilis*	8.7	2.6	0.1

Table 1.18: Results of SIMPER analyses showing species that were specific to either the mangroves or the adjacent seagrass over the course of the four seasons in which sampling was done. The % contribution to the overall separation of the assemblage is shown.

WESTERN MORETON BAY	
Mangrove specific species	% contribution
Liza argentea	3.08
Atherinomorus ogilbyi	2.90
Thalamita crenata	2.75
Sillago analis	2.30
Palaemon serenus	1.87
Hyporhampus regularis ardelio	1.28
Tylosaurus gavialoides	0.99
Seagrass specific species	% contribution
Macrobrachium sp.	4.59
Hapalochlaena sp.	4.14
Upeneus tragula	3.69
Latreutes compressus	2.69
Latreutes porcinus	1.99
Periclimenes nr. andamanensis	1.96
Monocanthus chinensis	1.80
Metapenaeus macleayi	0.67

EASTERN MORETON BAY	
Mangrove specific species	% contribution
Sillago ciliata	3.89
Liza argentea	2.23
Mugilogobius stigmaticus	1.91
Pseudomugil signifer	1.87
Scylla serrata	1.62
Hyporhampus regularis ardelio	1.38
Sillago maculata	1.31
Thalamita crenata	1.21
Seagrass specific species	% contribution
Periclimenes nr. *obscurus*	4.00
Centropogon australis	3.08
Chlorotocella nr. *gracilis*	3.02
Latreutes porcinus	2.69
Petroscirtes lupus	2.08
Periclimenes holthuisi	1.98
Latreutes pygmaeus	1.93

Table 1.19: List of species sampled from each of the habitats in western Moreton Bay over the course of 12 months sampling. The total number of individuals for each species is shown. M denotes species only sampled from mangroves, S from seagrass and Mu from mudflats.

Species		Mangrove	Seagrass	Mudflat	Total
Acanthopagrus australis		541	76	1	618
Ambassis marianus		140	3	0	143
Arrhampus sclerolepis	M	2	0	0	2
Atherinomorus ogilbyi	M	1270	0	0	1270
Butis sp	M	3	0	0	3
Cantherhinus pardalis	S	0	2	0	2
Centropogon australis		5	677	1	683
Chiloscyllum punctatum	M	1	0	0	1
Cottapistus praepositus	S	0	1	0	1
Dasyatus fluvorium		0	1	1	2
Eurysthmus leptus	M	14	0	0	14
Gerres subfasciatus		199	17	1	217
Girella tricuspidata	M	3	0	0	3
Gobiidae		178	4735	150	5063
Herklotsichthys castelnaui		21	8	0	29
Hippichthys pencillus		9	0	0	9
Hippocampus whitei	S	0	1	0	1
Hyperlophus sp	Mu	0	0	53	53
Hyperlophus translucidus		69	55	2	126
Hyporhampus regularis ardelio	M	80	0	0	80
Liza argentea	M	1135	0	0	1135
Marilyna pleurosticta		17	0	30	47
Microcanthus strigatus	M	4	0	0	4
Monacanthus chinensis		1	74	4	79
Monodactylus argenteus		3	0	2	5
Mugil cephalus	M	115	0	0	115
Mugilidae	M	96	0	0	96
Mugilogobius stigmaticus		636	6	0	642
Myxus elongatus	M	6	0	0	6
Omobranchus amolius		3	1	0	4
Omobranchus verticalis		9	3	1	13
Paramonacanthus otiensis		1	5	1	7
Pelates quadrilineatus	S	0	14	0	14
Pelates sexilineatus		439	1806	21	2266
Petroscirtes lupus		4	118	3	125
Platycephalus caeruleopunctatus	S	0	2	0	2
Platycephalus fuscus	M	1	0	0	1
Plotosus lineatus	M	61	0	0	61
Pseudomugil signifer	M	24	0	0	24
Pseudorhombus arsius	S	0	2	0	2
Repomucenus sp	Mu	0	0	4	4
Rhabdosargus sarba		14	6	0	20
Selenotoca multifasciata	M	122	0	0	122

Scobinichthys granulatus	S	0	1	0	1
Siganus fuscescens	S	0	3	0	3
Sillago analis	M	501	0	0	501
Sillago ciliata		143	0	2	145
Sillago maculata		59	13	98	170
Solegnathus sp	S	0	1	0	1
Sphyraena obtusata		83	17	3	103
Stigmatopora nigra	S	0	3	0	3
Suggrundus jugosus	S	0	1	0	1
Suggrundus sp	M	1	0	0	1
Sygnathidae		2	1	0	3
Sygnathoides biaculeatus	S	0	4	0	4
Terapon jarbua	S	0	1	0	1
Tetractenos hamiltoni		5808	193	20	6021
Thryssa aesturia		315	52	7	374
Torquigener pleurogramma	S	0	1	3	4
Torquigener squamicauda	Mu	0	0	14	14
Tylosurus gavialoides	M	32	0	0	32
Urocampus carinirostris		0	437	3	440
Valamugil georgii		54	1	0	55
Vanacampus sp		1	1	0	2
Fish sp 11	S	0	1	0	1
unidentifiable carid	S	0	1	0	1
Carid sp A	Mu	0	0	1	1
Caradina maccullochi		1	0	2	3
Chlorotocella nr gracilis	S	0	3	0	3
Latreutes porcinus	S	0	611	0	611
Latreutes pygmaeus	S	0	510	0	510
Leander tenuicornis		4	87	0	91
Macrobrachium sp	Mu	0	0	364	364
Metapenaeus bennettae		1056	2534	16	3606
Metapenaeus ensis		754	498	1	1253
Metapenaeus macleayi		0	22	279	301
Mysidacea		1743	292	0	2035
Palaemonidae	M	4	0	0	4
Palaemon debilis		4563	1246	0	5809
Palaemon serenus		1075	2880	0	3955
Penaeidae		72	293	0	365
Penaeus esculentus		36	416	417	869
Penaeus plebejus		26	1350	0	1376
Periclimenes andamanensis		1	3	0	4
Periclimenes holthuisi		0	6	3	9
Periclimenes obscurus		23	8076	0	8099
Periclimenes sp	S	0	5	0	5
Pontophilus sp 3	Mu	0	0	4	4
Portunus pelagicus		2	6	0	8
Scylla serrata	M	95	0	0	95
Thalamita crenata	M	4	0	0	4
Euprymna tasmanica	S	0	3	0	3

Hapalochlaena sp	Mu	0	0	3	3
Idiosepius notoides		3	327	0	330
Sepioteuthis lessoniana		2	1	0	3
Sepioteuthis sp		4	1	1515	1520

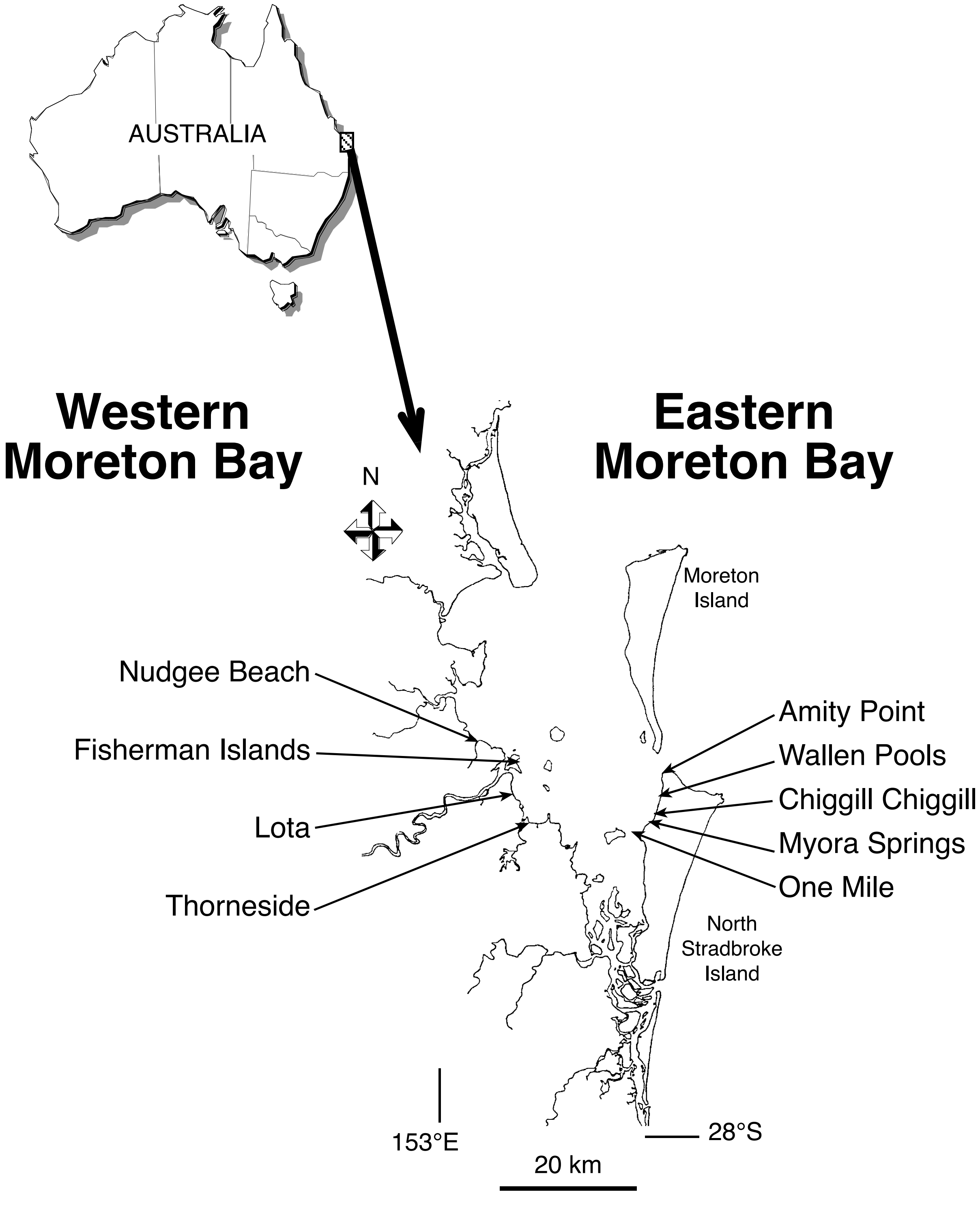

Figure 1.1: Map of Australia and the Moreton Bay region of SE Queensland showing the different sites sampled in western and eastern Moreton Bay as part of the broad-scale general sampling programme in different nearshore habitats.

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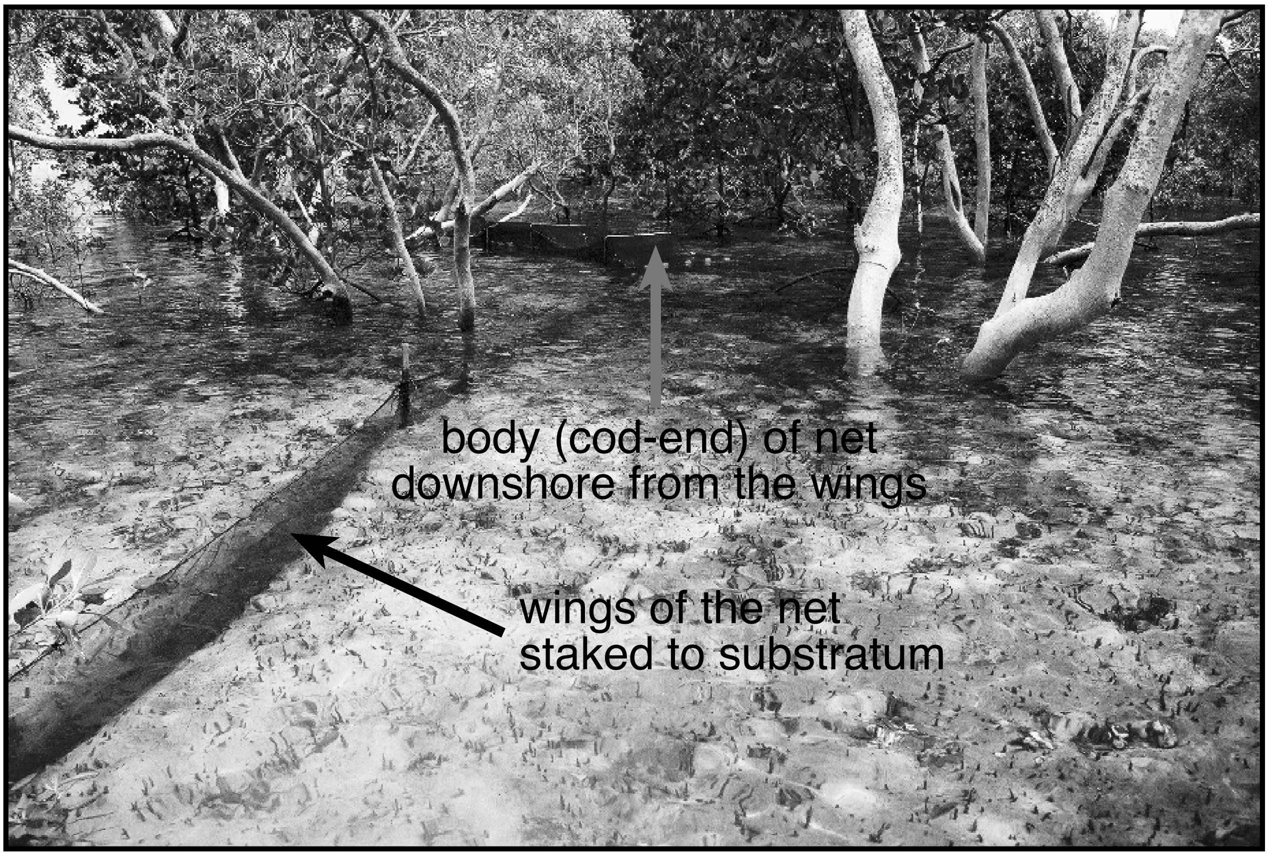
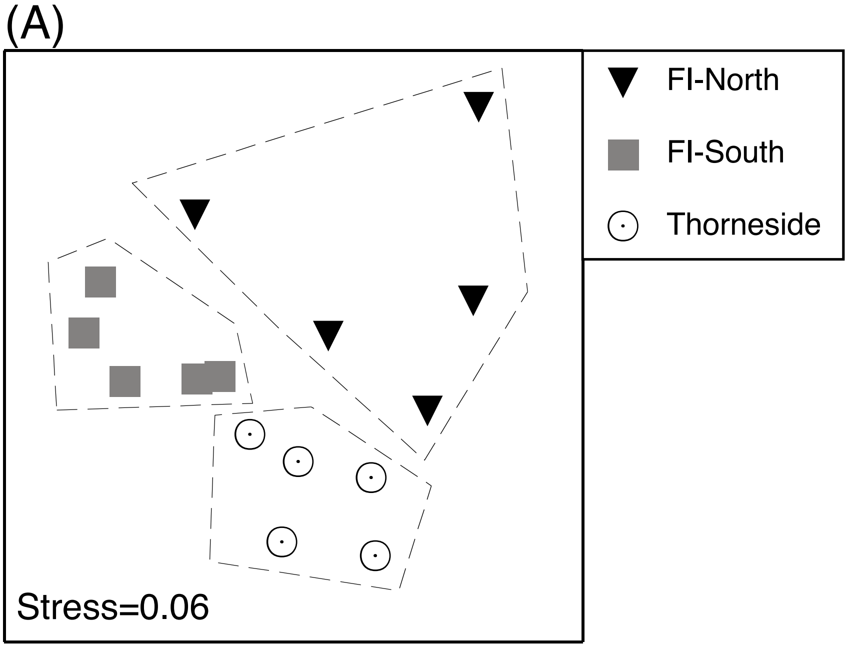


Figure 1.2: Photographs showing the deployment of fyke nets within the mangroves in western Moreton Bay. The design of the nets allowed the wings to be positioned around trees and other structures within the forest (see text for further details).



(B)

FI-North vs FI-South	*Tetractenos hamiltoni*	31 %
	Metapenaeus bennettae	18 %
	Metapenaeus ensis	11 %
	Palaemon debilis	08 %
	Liza argentea	07 %
	Mugilogobius stigmaticus	07 %
	Acanthopagrus australis	06 %
FI-North vs Thorneside	*Tetractenos hamiltoni*	33 %
	Metapenaeus bennettae	11 %
	Palaemon debilis	10 %
	Acanthopagrus australis	10 %
	Metapenaeus ensis	08 %
FI-South vs Thorneside	*Tetractenos hamiltoni*	30 %
	Metapenaeus bennettae	17 %
	Metapenaeus ensis	11 %
	Acanthopagrus australis	08 %
	Liza argentea	07 %
	Mugilogobius stigmaticus	07 %
	Palaemon debilis	06 %

Figure 1.3: (A) nMDS ordination of the composition of the nekton assemblage utilising mangroves in western Moreton Bay during Spring 2002. Data are catches standardised for a 2 hour soak time for fyke nets set ~5 m into the mangrove forest from the lower edge (n=5 fyke nets deployed at each site). (B) Results of SIMPER analysis, showing those species contributing 6 % or more to the separation of the sites in the ordination.

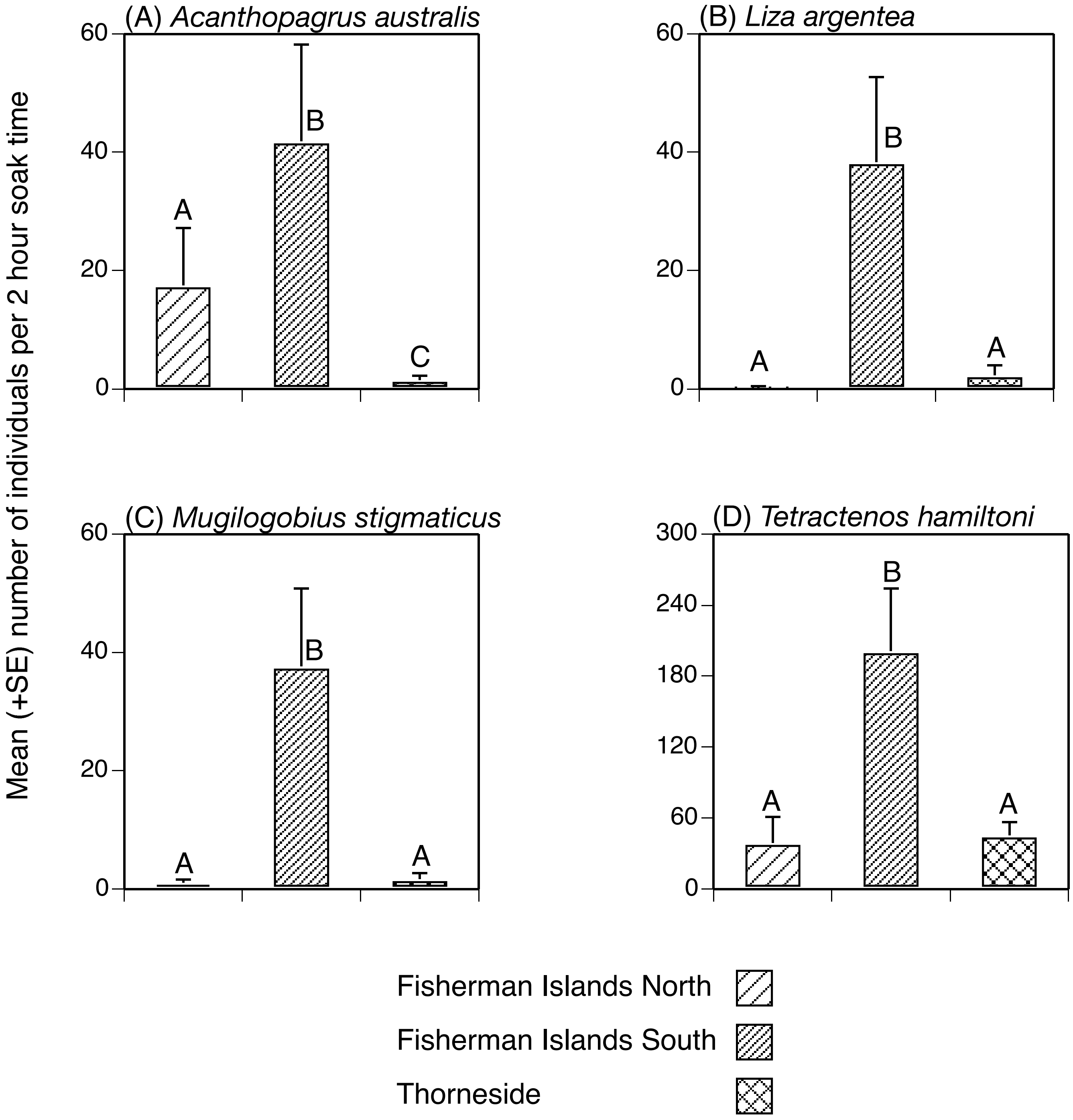


Figure 1.4: Mean (+SE) abundance of (A) *Acanthopagrus australis*, (B) *Liza argentea*, (C) *Mugilogobius stigmaticus*, and (D) *Tetractenos hamiltoni* per 2 hours soak time of fyke nets deployed ~ 5 metres into the mangrove forest above the pneumatophore zone. Five replicate fyke nets were deployed in each site during Spring 2002. The letters above the bars show the results of SNK post-hoc contrasts after ANOVA: means topped by the same letter were not significantly different from each other (P > 0.05). Note the differences in the scale on the Y-axis.

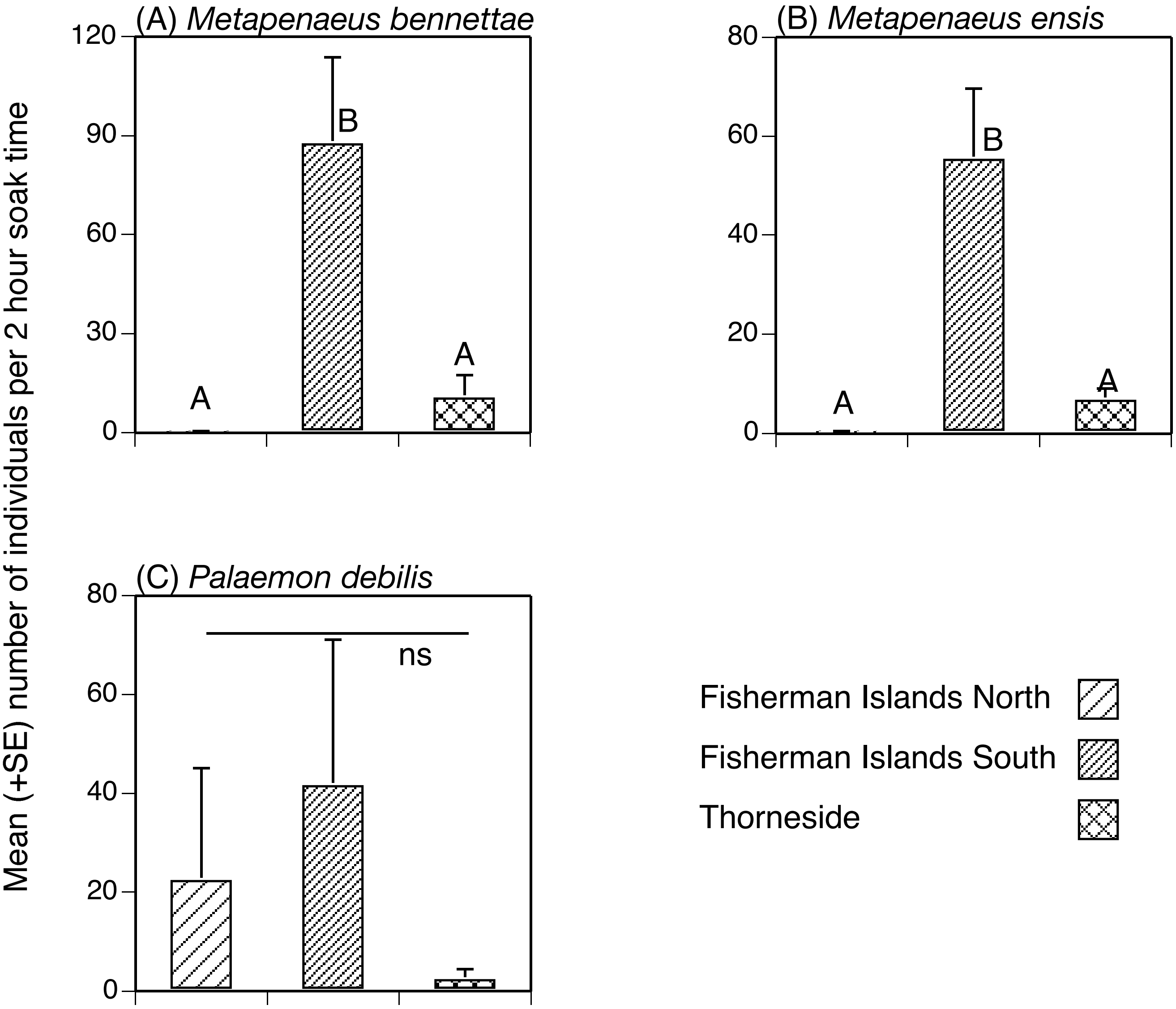
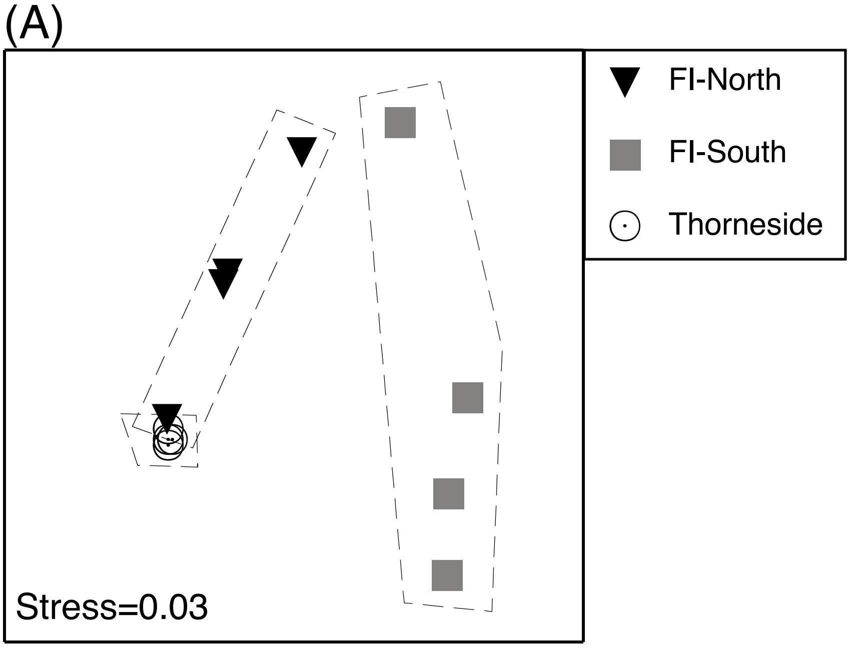


Figure 1.5: Mean (+SE) abundance of (A) *Metapenaeus bennettae*, (B) *Metapenaeus ensis* and (C) *Palaemon debilis* per 2 hours soak time of fyke nets deployed ~ 5 metres into the mangrove forest above the pneumatophore zone. Other details as in Figure 1.4. Note the differences in the scale on the Y-axis.



(B)

FI-North vs FI-South	Gobies	27 %
	Metapenaeus bennettae	20 %
	Pelates sexlineatus	07 %
	Centropogon australis	07 %
	Penaeus plebejus	06 %
	Urocampus carinirostris	06 %
	Metapenaeus ensis	05 %
FI-North vs Thorneside	Gobies	41 %
	Penaeus plebejus	09 %
	Centropogon australis	08 %
	Latreutes porcinus	07 %
	Pelates sexlineatus	07 %
	Latreutes pygmaeus	06 %
	Metapenaeus bennettae	05 %
FI-South vs Thorneside	Gobies	43 %
	Penaeus plebejus	09 %
	Centropogon australis	08 %
	Pelates sexlineatus	06 %
	Latreutes porcinus	06 %
	Metapenaeus bennettae	06 %
	Latreutes pygmaeus	05 %

Figure 1.6: (A) nMDS ordination of the composition of the nekton assemblage utilising intertidal seagrass in western Moreton Bay during Spring 2002. Data are number of animals caught in a seine net hauled across 100 m^2 of habitat (n = 4 hauls completed at each site). (B) Results of SIMPER analysis, showing those species contributing 5 % or more to the separation of the sites in the ordination.

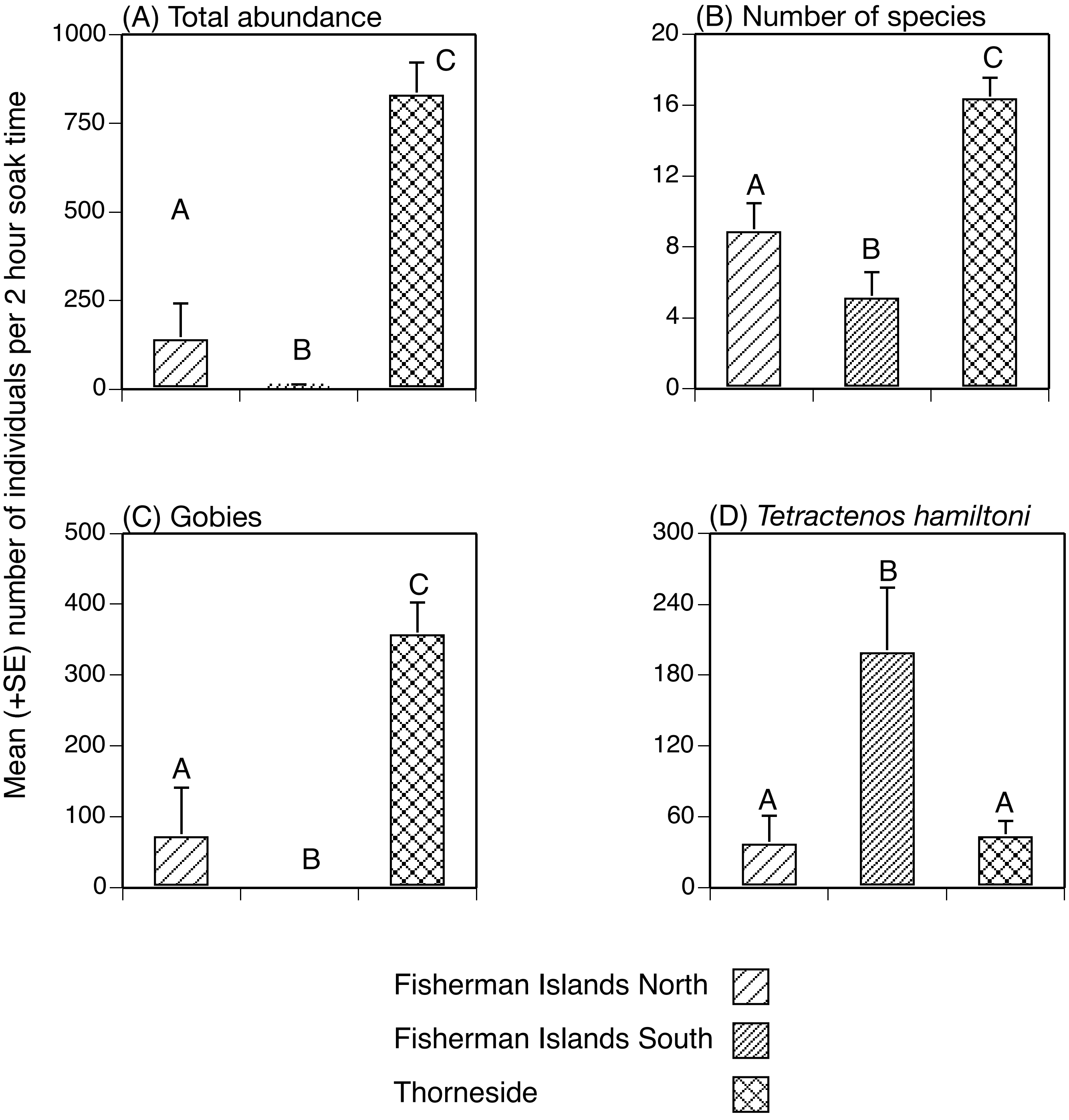


Figure 1.7: Mean (+SE) (A) number of species and abundance of (B) all nekton, (C) gobies, (D) *Tetractenos hamiltoni*, (E) *Pelates sexlineatus*, (F) *Centropogon australis* and (G) *Urocampus cariniros* per 100m^2 of intertial seagrass. Four replicate hauls of a seine nets were done in each site during Spring 2002. The letters above the bars show the results of SNK post-hoc contrasts after ANOVA: means topped by the same letter were not significantly different from each other (P > 0.05). Note the differences in the scale on the Y-axis.

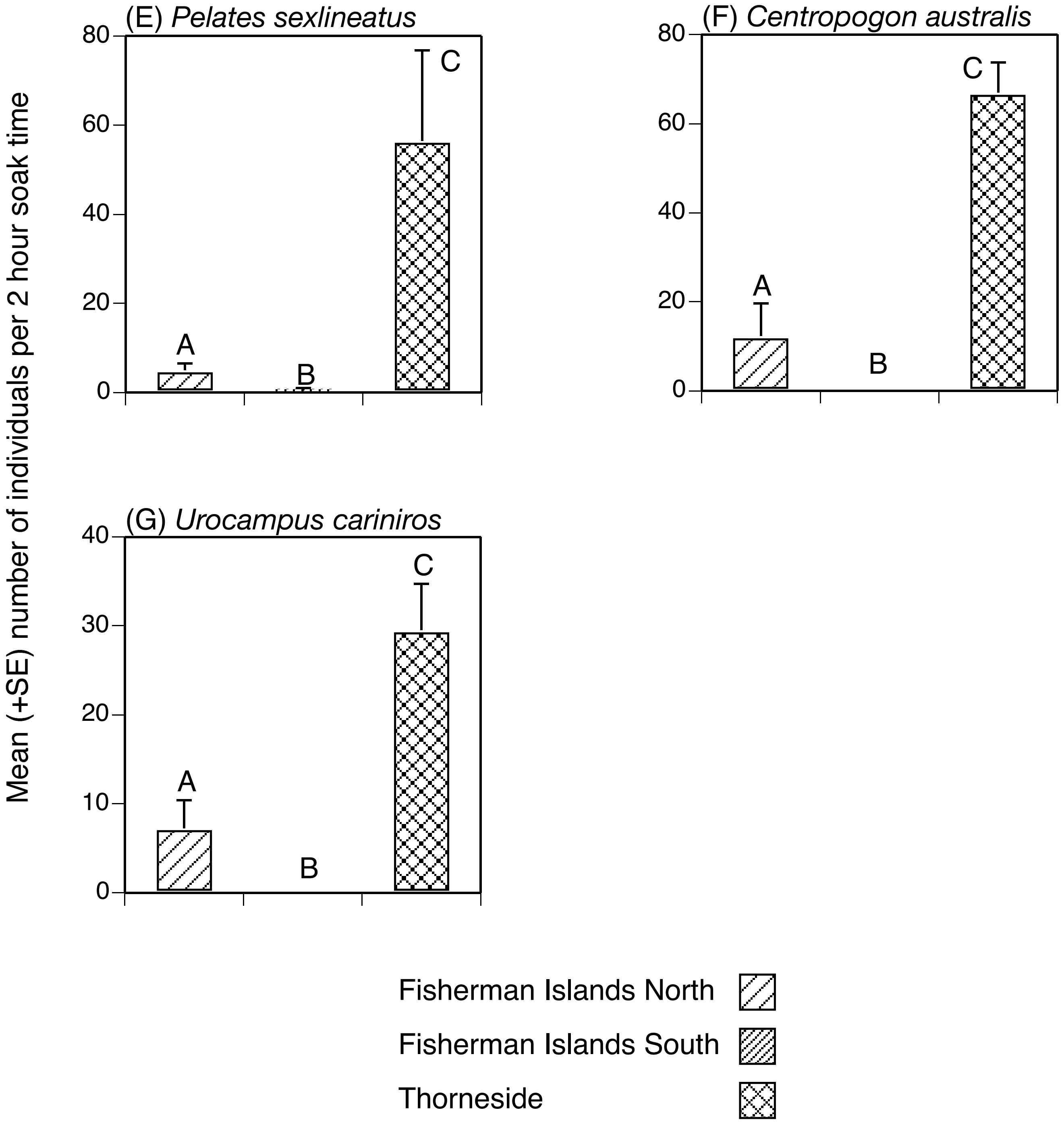


Figure 1.7: Continued..../

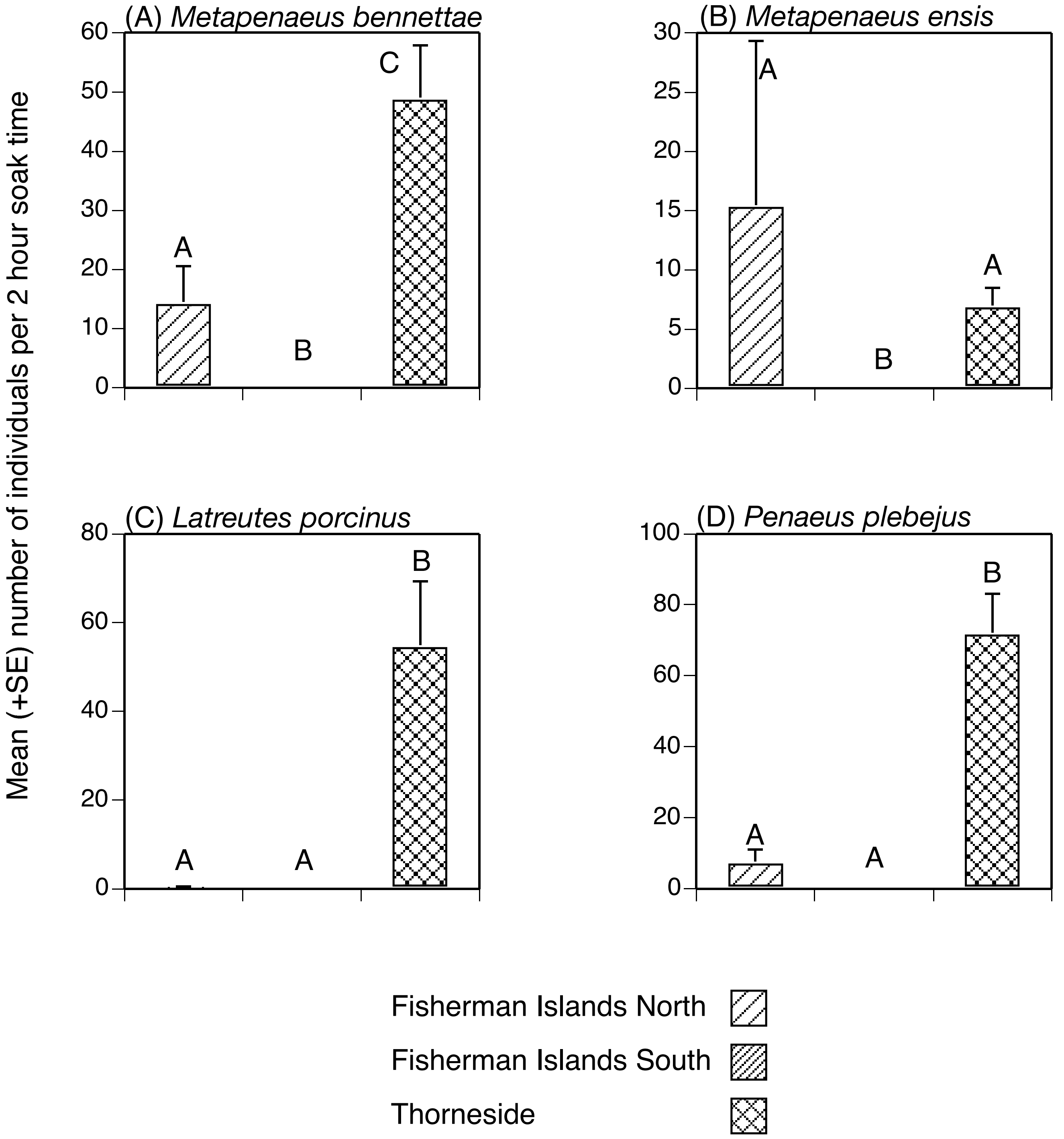


Figure 1.8: Mean (+SE) abundance of (A) *Metapenaeus bennettae*, (B) *Metapenaeus ensis*, (C) *Penaeus plebejus* and (D) *Latreutes porcinus* per 100m^2 of intertial seagrass. Four replicate hauls of a seine net were done in each site during Spring 2002. The letters above the bars show the results of SNK post-hoc contrasts after ANOVA: means topped by the same letter were not significantly different from each other (P > 0.05). Note the differences in the scale on the Y-axis.

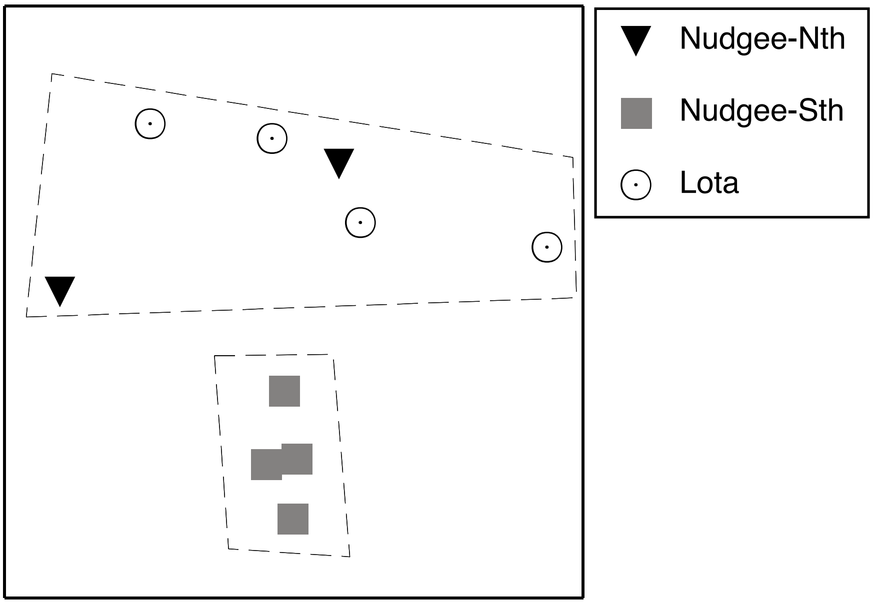
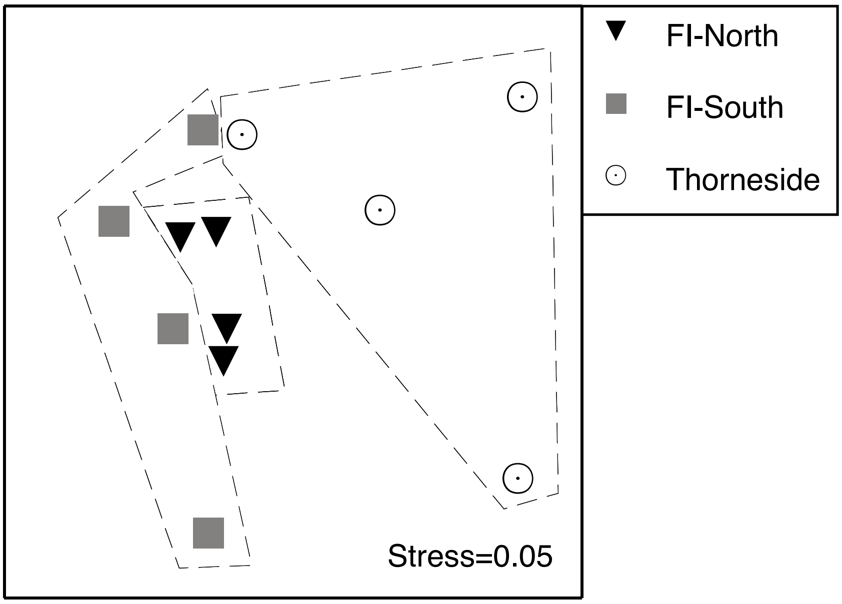


Figure 1.9: nMDS ordination of the composition of the nekton assemblage utilising intertidal mudflats in western Moreton Bay during Spring 2002. Data are the number of animals caught in a seine net hauled across 100 m^2 of habitat (n=4 hauls completed at each site). Only two samples are shown for Nudgee Beach North because the other two samples were empty.



FI-North vs FI-South	*Hippolyte caradina*	18 %
	Periclimenes obscurus	16 %
	Pelates sexlineatus	08 %
	Gobies	08 %
	Penaeus esculentus	06 %
	Latreutes pygmaeus	06 %
FI-North vs Thorneside	*Periclimenes obscurus*	24 %
	Hippolyte caradina	19 %
	Gobies	11 %
	Pelates sexlineatus	09 %
	Latreutes pygmaeus	07 %
	Pelates sexlineatus	07 %
	Latreutes pygmaeus	06 %
FI-South vs Thorneside	*Periclimenes obscurus*	26 %
	Hippolyte caradina	14 %
	Pelates sexlineatus	11 %
	Hippolyte sp.	07 %
	Monacanthus chinensis	07 %
	Penaeus esculentus	06 %

Figure 1.10: (A) nMDS ordination of the composition of the nekton assemblage utilising subtidal seagrass in western Moreton Bay during Spring 2002. Data are the number of fish caught in a seine net hauled across 100 m^2 of habitat (n=4 hauls completed at each site). (B) Results of SIMPER analysis, showing those species contributing 5 % or more to the separation of the sites in the ordination. FI-North = Fisherman Islands North, FI-South= Fisherman Islands South.

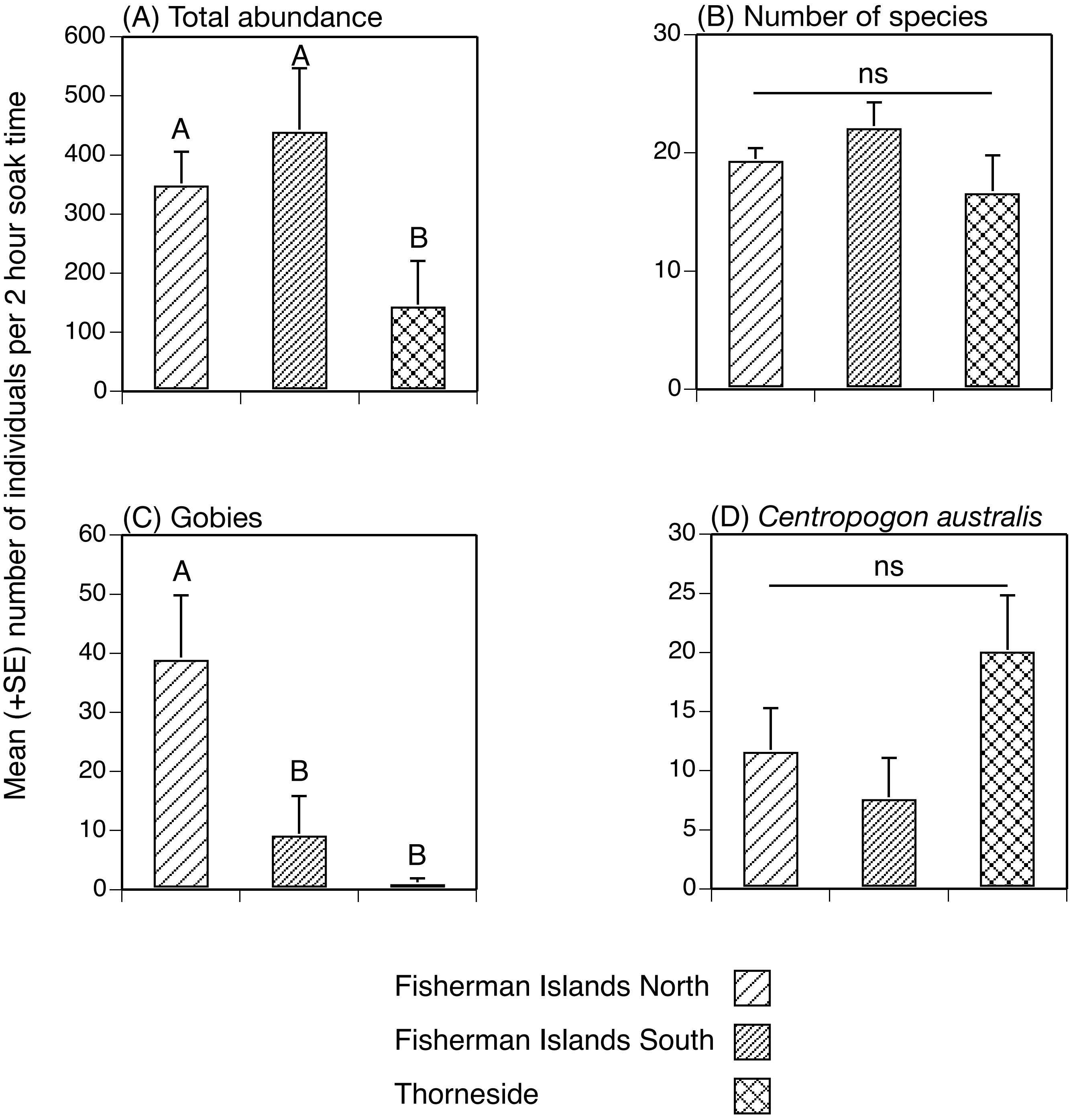


Figure 1.11: Mean (+SE) (A) number of species and abundance of (B) all nekton, (C) gobies, (D) *Centropogon australis*, (E) *Pelates sexlineatus* and (F) *Monocanthus chinensis* per $100m^2$ of subtidal seagrass. Four replicate hauls of a seine nets were done in each site during Spring 2002. The letters above the bars show the results of SNK post-hoc contrasts after ANOVA: means topped by the same letter were not significantly different from each other ($P > 0.05$); ns indicates not significantly different ($P > 0.05$) in ANOVA. Note the differences in the scale on the Y-axis.

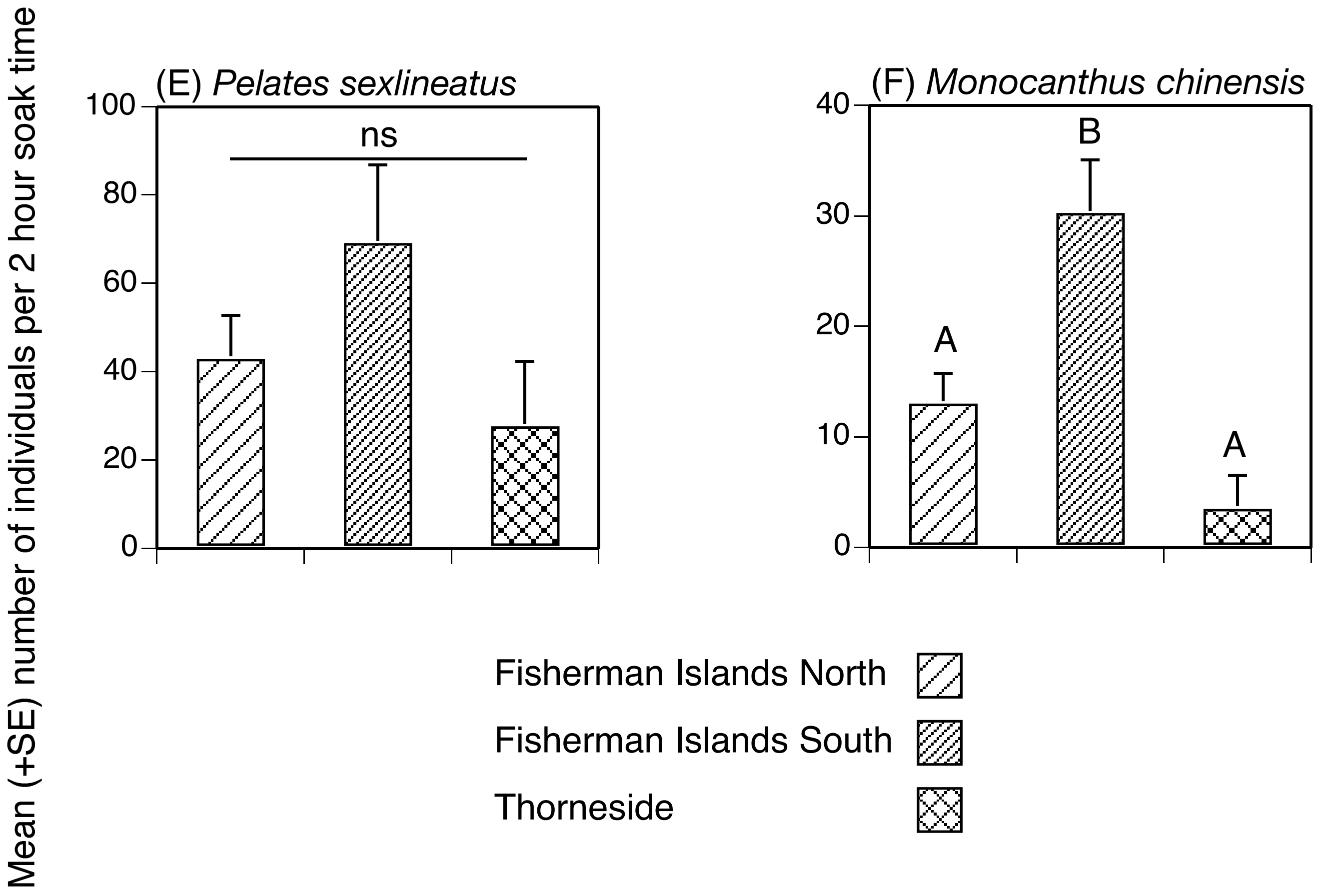


Figure 1.11: Continued..../

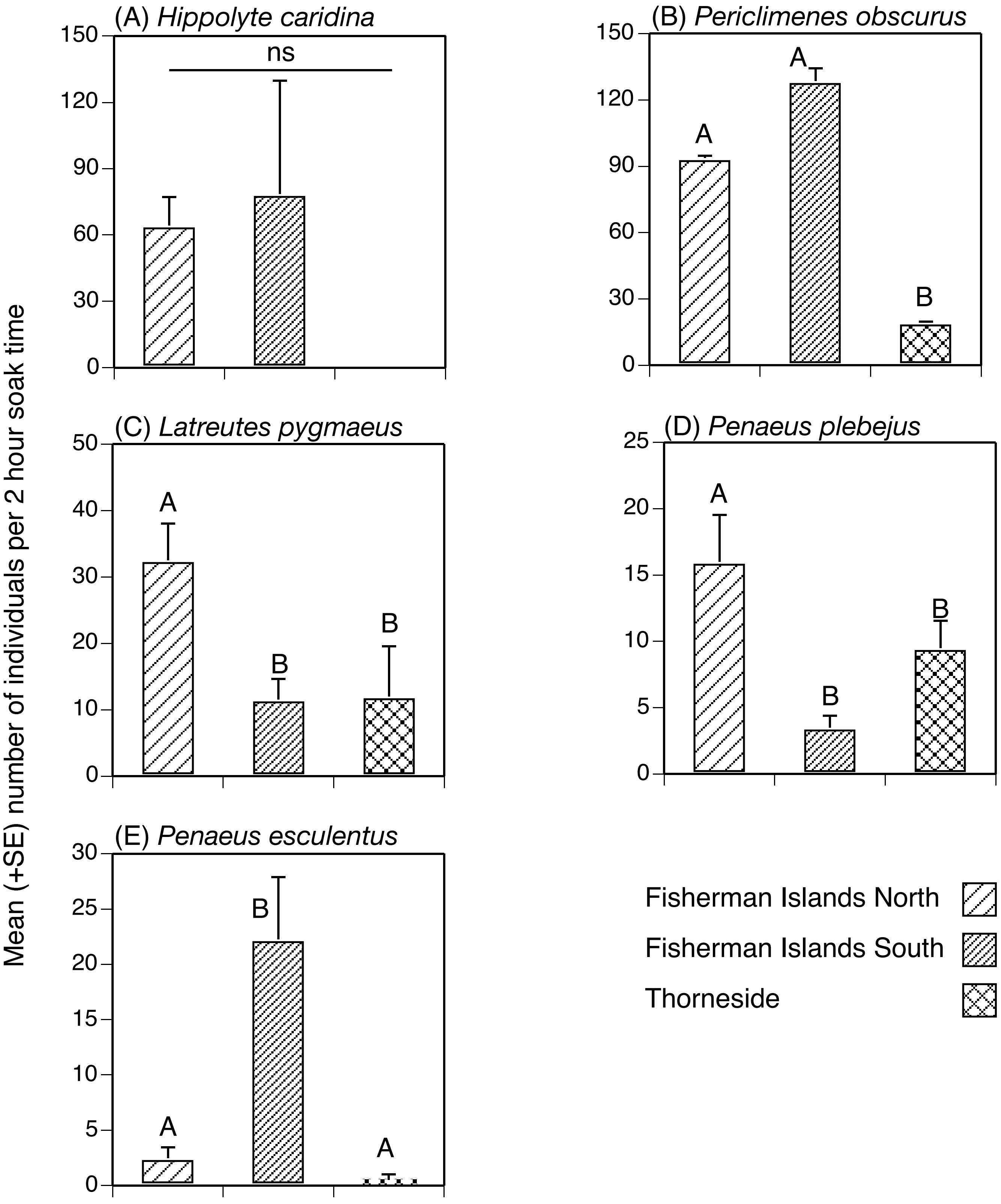
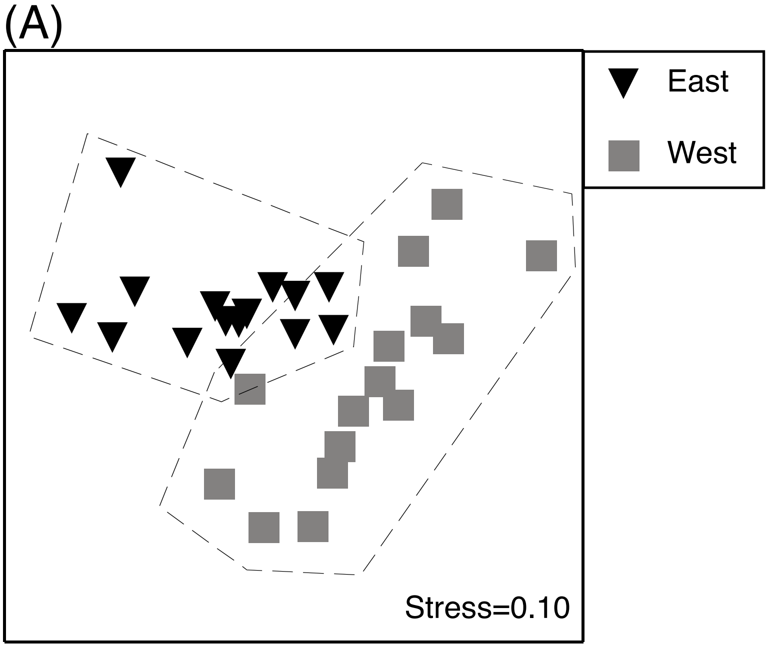


Figure 1.12: Mean (+SE) number of (A) *Hippolyte caradina*, (B) *Periclimenes obscurus*, (C) *Latreutes pygmaeus*, (D) *Penaeus plebejus* and (E) *Penaeus esculentus* per 100m^2 of subtidal seagrass. Four replicate hauls of a seine nets were done in each site during Spring 2002. The letters above the bars show the results of SNK post-hoc contrasts after ANOVA: means topped by the same letter were not significantly different from each other ($P > 0.05$); ns indicates not significantly different ($P > 0.05$) in ANOVA. Note the differences in the scale on the Y-axis.



(B)

East MB vs West MB	*Palaemon debilis*	**34 %**
	Tetractenos hamiltoni	16 %
	Ambassis marianus	12 %
	Atherinomorus ogilbyi	09 %
	Metapenaeus bennettae	06 %

Figure 1.13: (A) nMDS ordination of the composition of the nekton assemblage utilising mangroves in western and eastern Moreton Bay during Spring 2002. Data are catches standardised for a 2 hour soak time for fyke nets set ~ 5 m into the mangrove forest from the lower edge (n = 5 fyke nets deployed at each site). (B) Results of SIMPER analysis, showing the those species contrinuting more than 5 % to the separation of the two regions from each other.

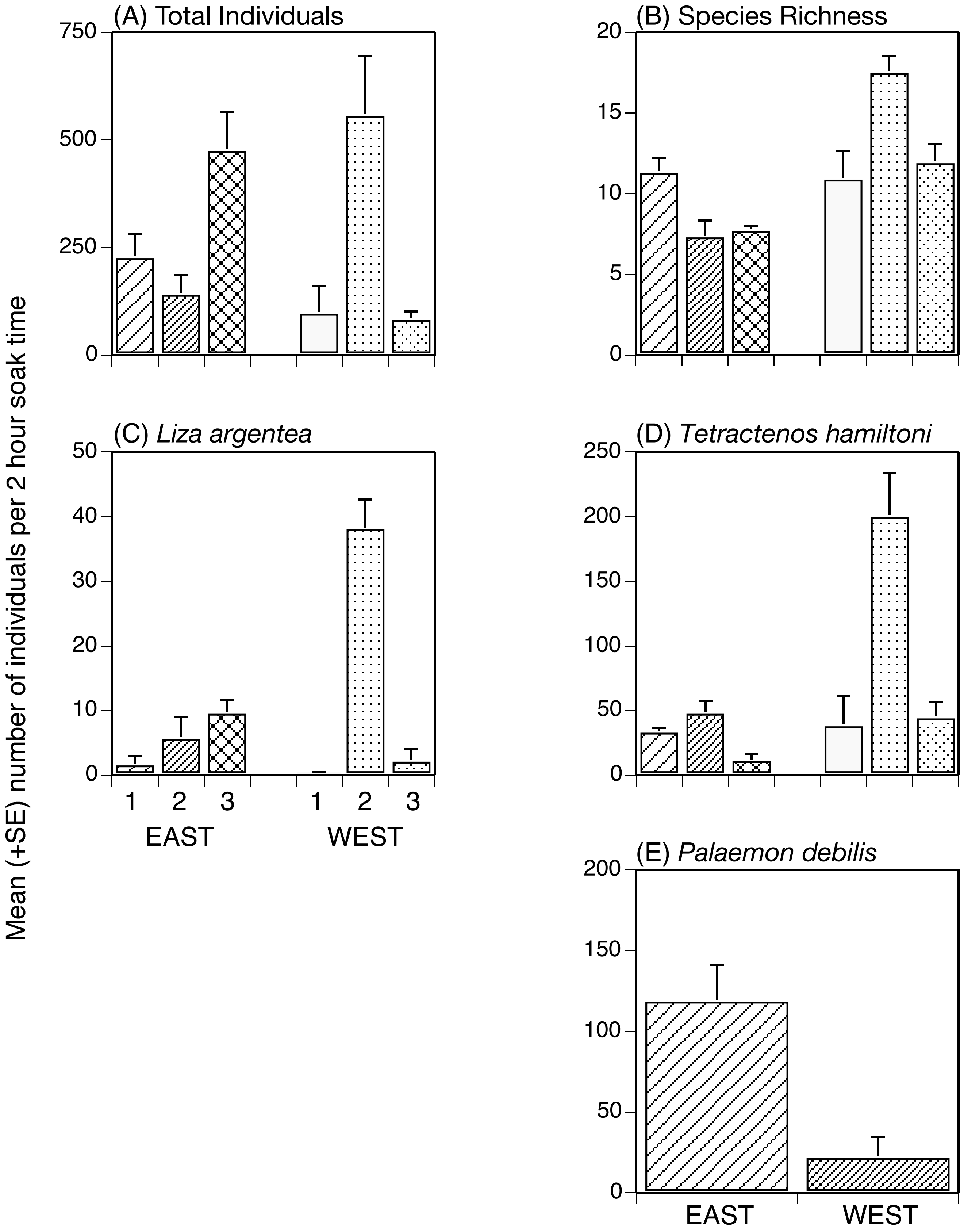
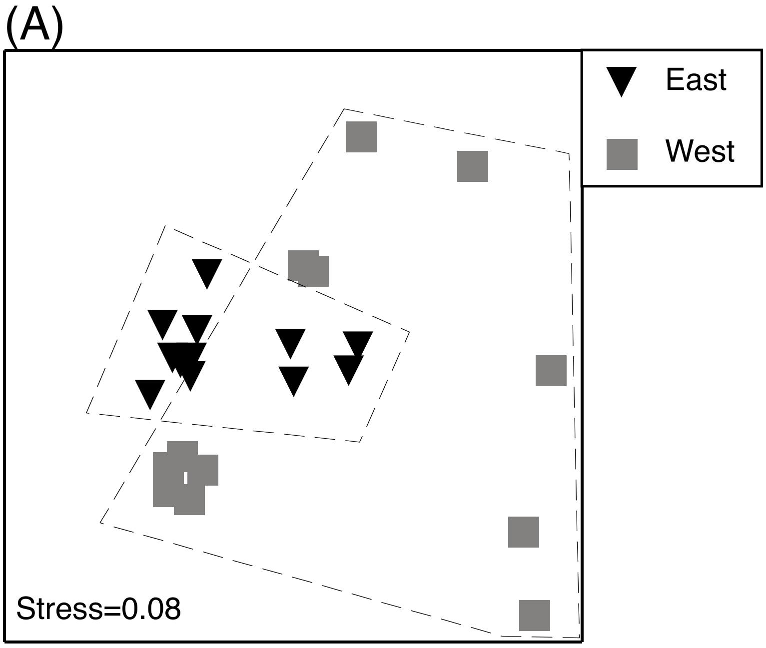


Figure 1.14: Mean (+SE) number of (A) individuals, (B) species, (C) *Liza argentea*, (D) *Tetractenos hamiltoni* and (E) *Palaemon debilis* per 2 hours soak time of fyke nets deployed ~ 5 m into the mangrove forest above the pneumatophore zone in each of three sites in eastern and western Moreton Bay. Five replicate fyke nets were deployed in each site in Spring 2002. See text for further details. Note the differences in the scale on the Y-axis.



(B)

East MB vs West MB	*Penaeus plebejus*	**35 %**
	Gobies	27 %
	Urocampus carinirostris	07 %
	Pelates sexlineatus	05 %

Figure 1.15: (A) nMDS ordination of the composition of the nekton assemblage utilising intertidal seagrass beds in western and eastern Moreton Bay during Spring 2002. Data are number of animals caught in a seine net hauled across 100 m^2 of habitat (n=4 hauls completed at each of three sites in each region. (B) Results of SIMPER analysis, showing those species contributing more than 5 % to the separation of the two regions from each other.

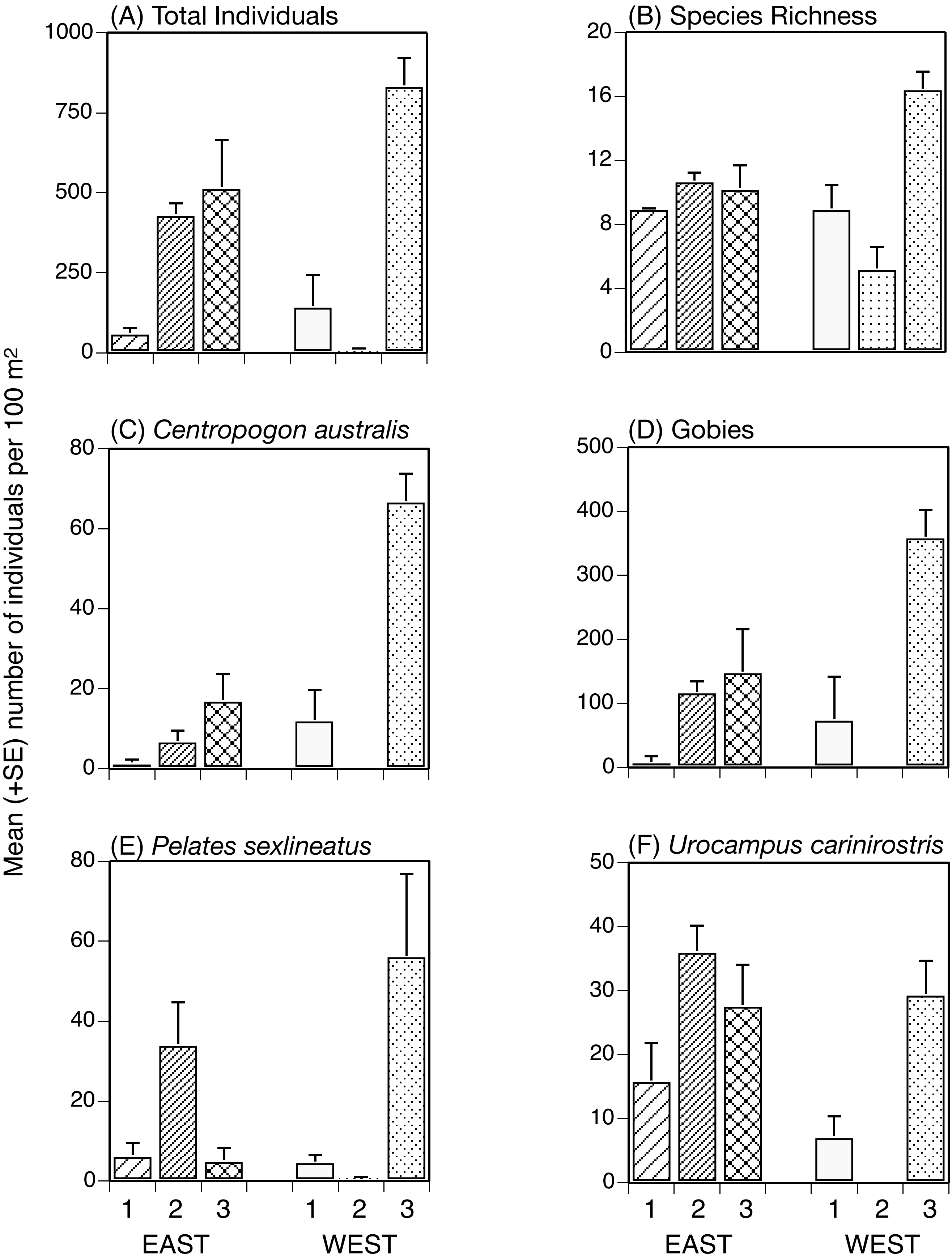


Figure 1.16: Mean (+SE) number of (A) individuals, (B) species, (C) *Centropogon australis,* (D) gobies, (E) *Pelates sexlineatus* and (F) *Urocampus carinirostris* per 100 m^2 of seagrass sampled in each of three sites in eastern and western Moreton Bay. Four replicate samples were collected in each site in Spring 2002. See text for further details. Note the differences in the scale on the Y-axis.

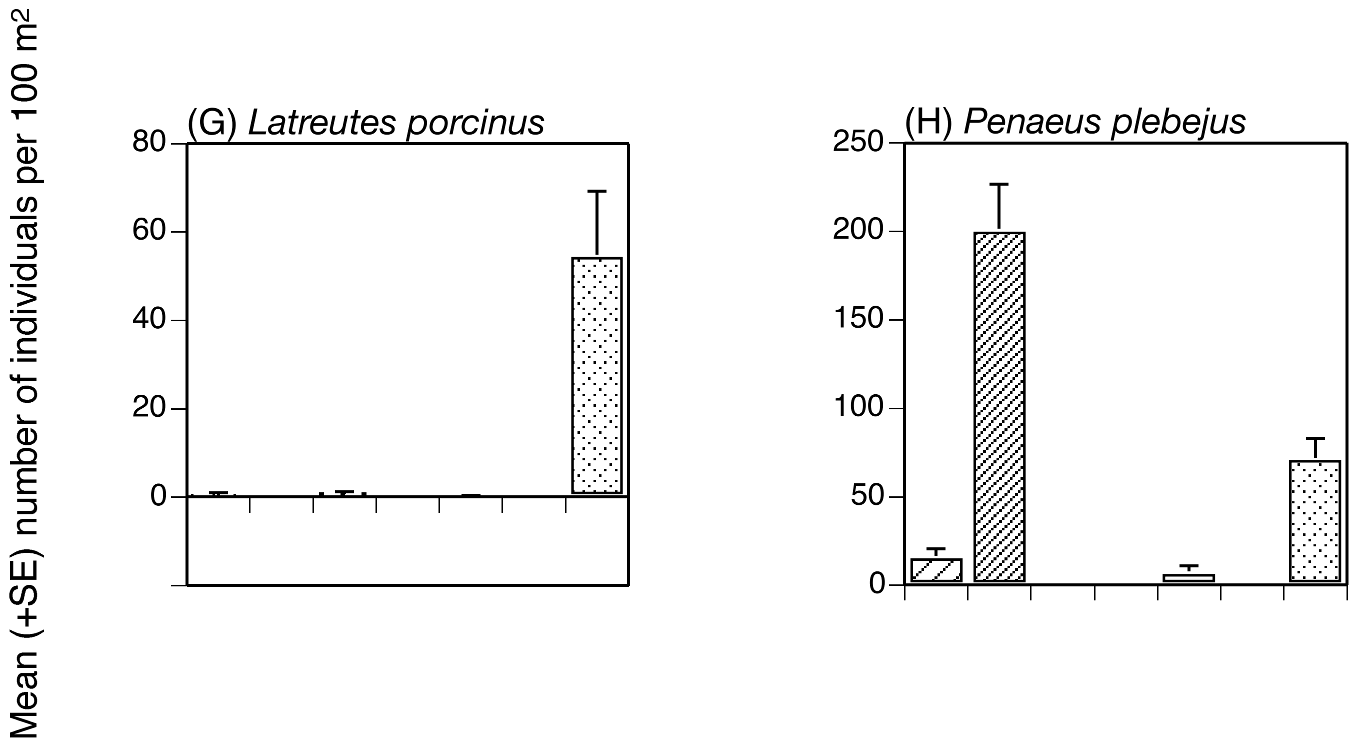
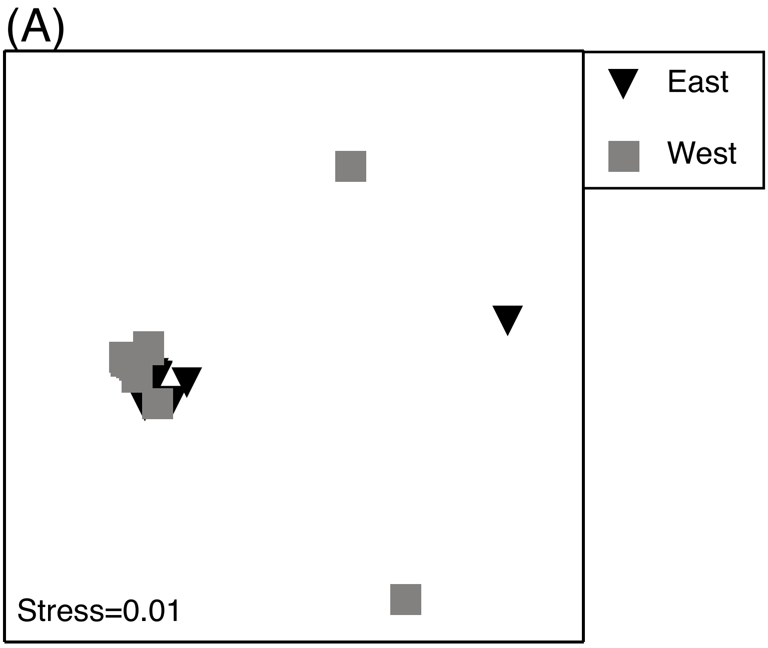


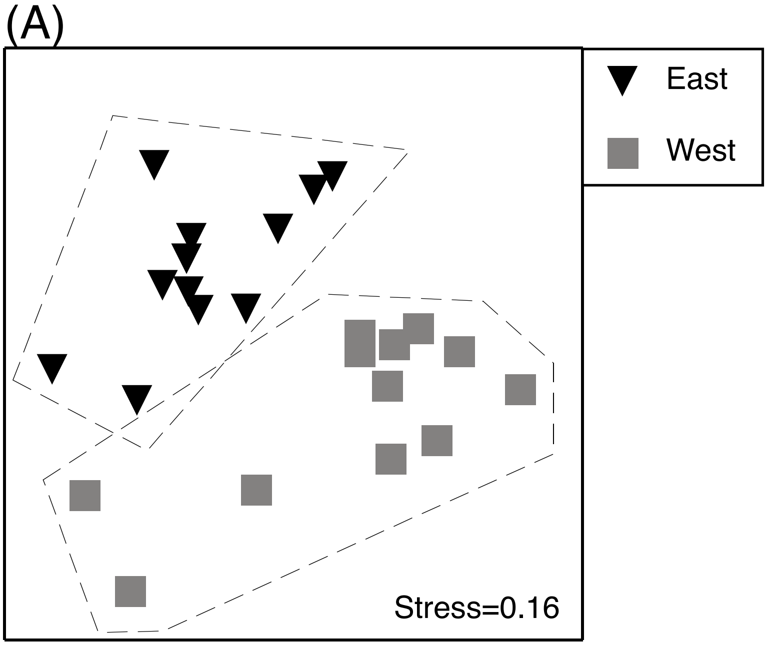
Figure 1.16 continued: Mean (+SE) number of (G) *Latreutes porcinus* and (H) *Penaeus plebejus* per 100 m^2 of seagrass sampled in each of three sites in eastern and western Moreton Bay. Four replicate samples were collected in each site in Spring 2002. See text for further details. Note the differences in the scale on the Y-axis.



(B)

East MB vs West MB	*Penaeus plebejus*	32 %
	Hyperlophus sp.	14 %
	Chlorotocella nr gracilis	13 %
	Pelates sexlineatus	08 %
	Atherinomorus ogilbyi	05 %

Figure 1.17: (A) nMDS ordination of the composition of the nekton assemblage utilising intertidal unvegetated mudflats in western and eastern Moreton Bay during Spring 2002. Data are number of animals caught in a seine net hauled across 100 m^2 of habitat (n=4 hauls completed at each of three sites in each region. Several samples did not catch any individuals so these samples have been excluded from the multivariate analysis. (B) Results of SIMPER analysis, showing those species contributing more than 5 % to the separation of the two regions from each other.



(B)

East MB vs West MB	*Palaemon obscurus*	17 %
	Pelates sexlineatus	11 %
	Hippolyte caradina	10 %
	Penaeus plebejus	10 %
	Mysidacea	09 %
	Gobiidae	06 %
	Latreutes pygmaeus	05 %

Figure 1.18: (A) nMDS ordination of the composition of the nekton assemblage utilising subtidal seagrass beds in western and eastern Moreton Bay during Spring 2002. Data are number of animals caught in a seine net hauled across 100 m^2 of habitat (n=4 hauls completed at each of three sites in each region. (B) Results of SIMPER analysis, showing those species contributing more than 5 % to the separation of the two regions from each other.

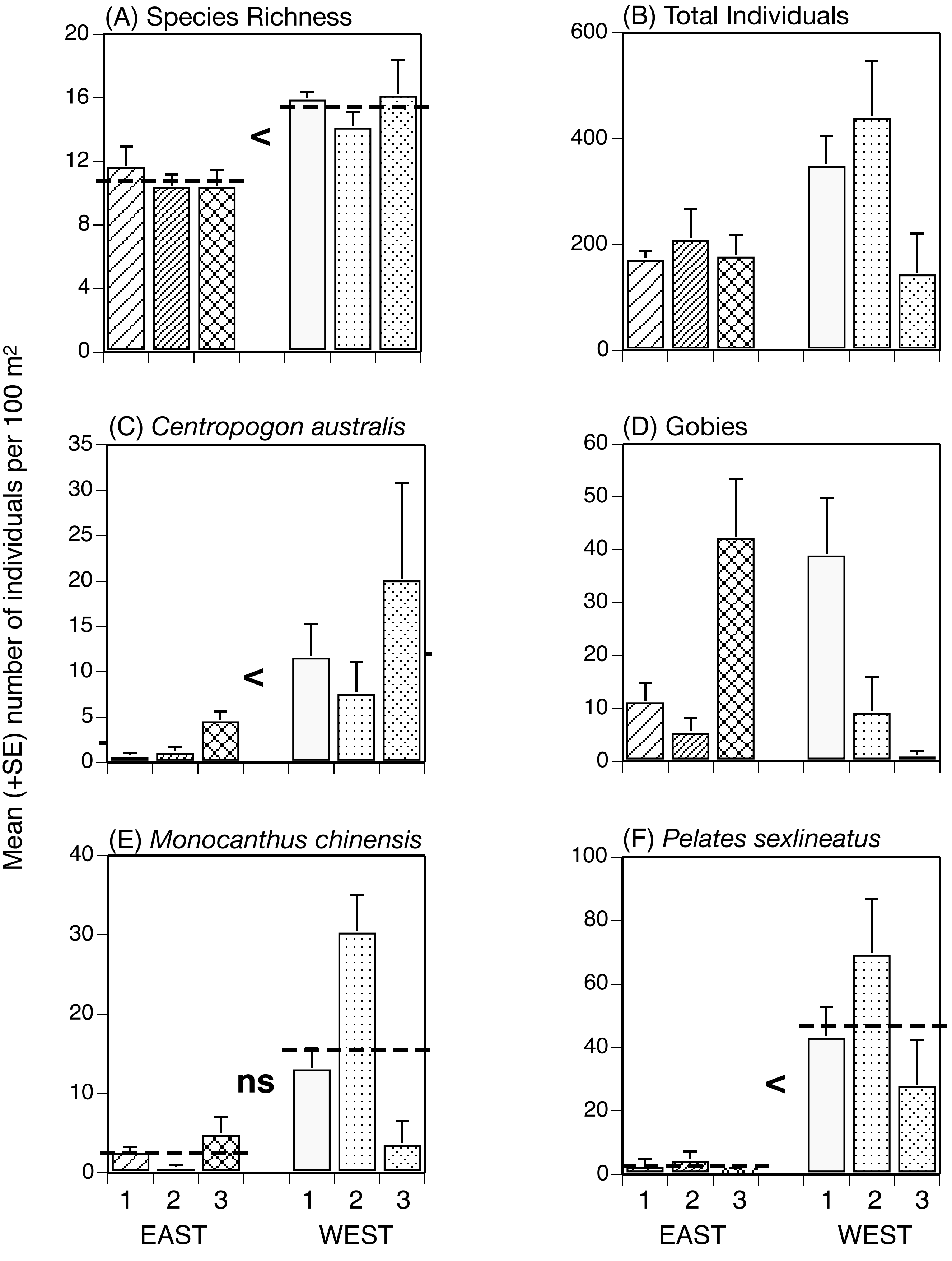


Figure 1.19: Mean (+SE) number of (A) species, (B) individuals, (C) *Centropogon australis*, (D) gobies, (E) *Monocanthus chinensis* and (F) *Pelates sexlineatus* per 100 m^2 of subtidal seagrass sampled in each of three sites in eastern and western Moreton Bay. Four replicate samples were collected in each site in Spring 2002. See text for further details. Note the differences in the scale on the Y-axis. The mean abundance for each region is shown as a dark dashed line across the the bars for the three sites within the region. > indicates a significant difference ($P < 0.05$) in the abundance between each region; ns indicates that the mean abundance was not significantly different between the two regions.

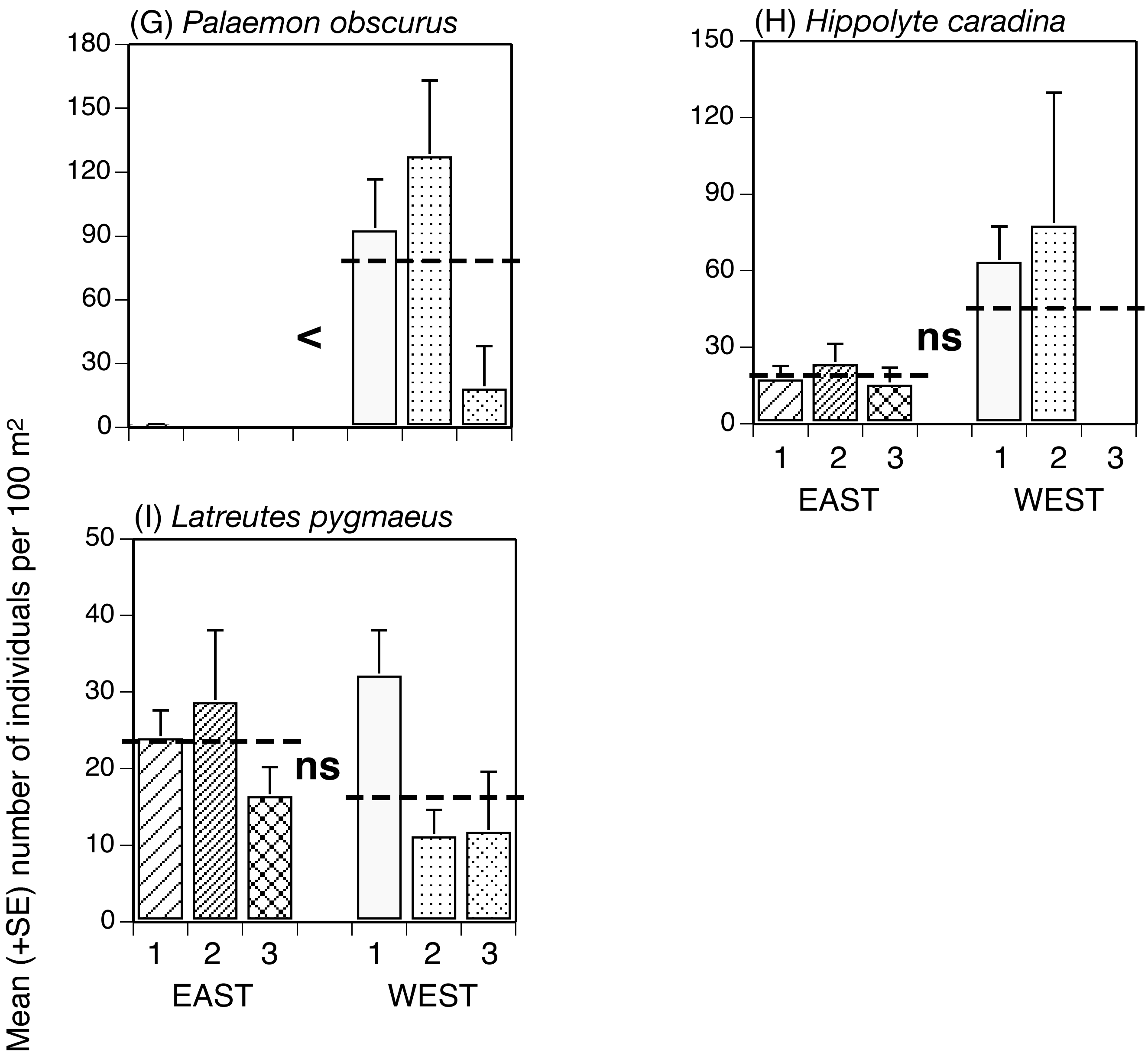
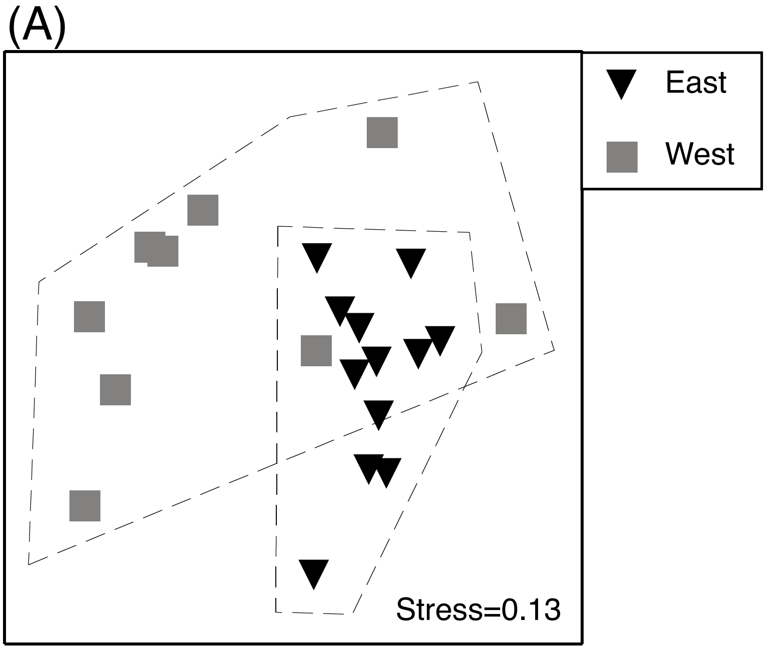


Figure 1.19 continued: Mean (+SE) number of (G) *Latreutes porcinus,* (H) *Hippolyte caradina* and (I) *Latreutes pygmaeus* per 100 m^2 of subtidal seagrass sampled in each of three sites in eastern and western Moreton Bay. Four replicate samples were collected in each site in Spring 2002. See text for further details. Note the differences in the scale on the Y-axis.

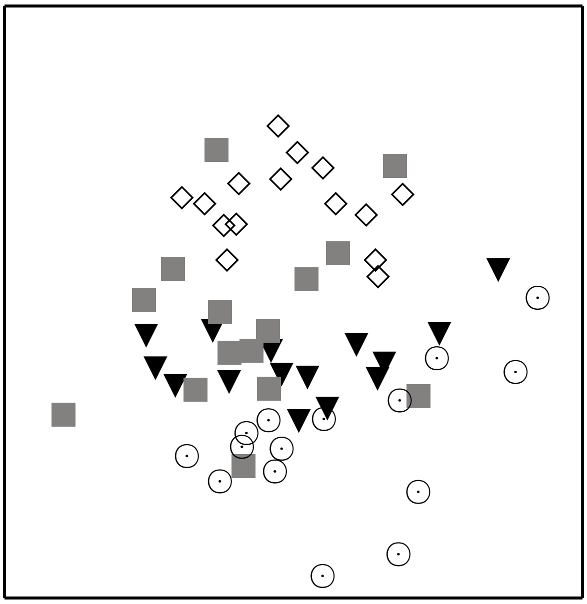


(B)

	Penaeus plebejus	23 %
	Chlorotocella gracilis	15 %
East MB	*Pontophilus* sp. 1	12 %
vs	Gobiidae	08 %
West MB	*Pontophilus* sp. 3	08 %
	Mysidacea	07 %
	Thyrssa aesturia	07 %

Figure 1.20: (A) nMDS ordination of the composition of the nekton assemblage utilising subtidal unvegetated sediments in western and eastern Moreton Bay during Spring 2002. Data are number of animals caught in a seine net hauled across 100 m^2 of habitat (n=4 hauls completed at each of three sites in each region. (B) Results of SIMPER analysis, showing those species contributing more than 5 % to the separation of the two regions from each other.

(A) All samples



(B) Site Centroids

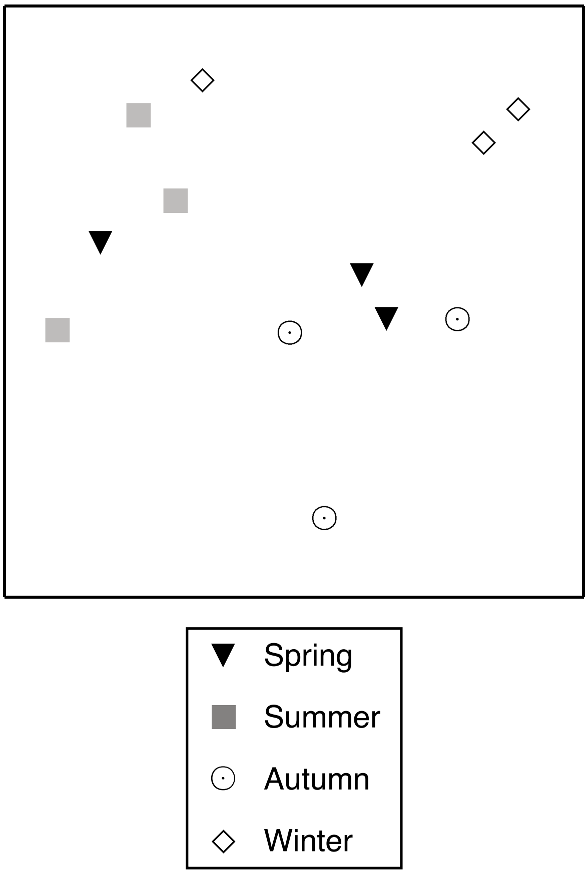


Figure 1.21: nMDS ordination of the composition of the nekton assemblage utilising mangrove forests in western Moreton Bay between Spring 2002 and Winter 2003. Data are catches standardised for a 2 hour soak time for fyke nets set ~ 5 m into the mangrove forest (n = 5 fyke nets deployed at each of 3 sites (Fisherman Islands North, Fisherman Islands South and Thorneside). (A) All data plotted; (B) Site centroids plotted.

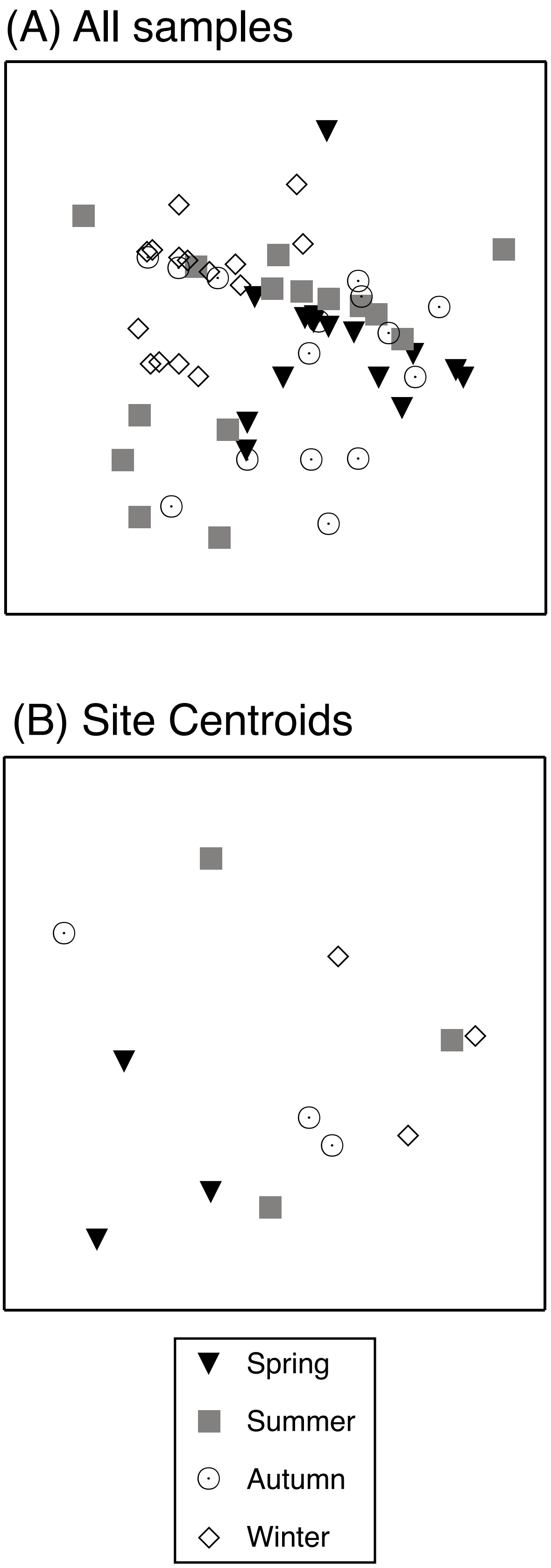


Figure 1.22: nMDS ordination of the composition of the nekton assemblage utilising mangrove forests in eastern Moreton Bay between Spring 2002 and Winter 2003. Data are catches standardised for a 2 hour soak time for fyke nets set ~ 5 m into the mangrove forest (n = 5 fyke nets deployed at each of 3 sites (Myora Springs, One Mile North and Wallen Pools South). (A) All data plotted; (B) Site centroids plotted.

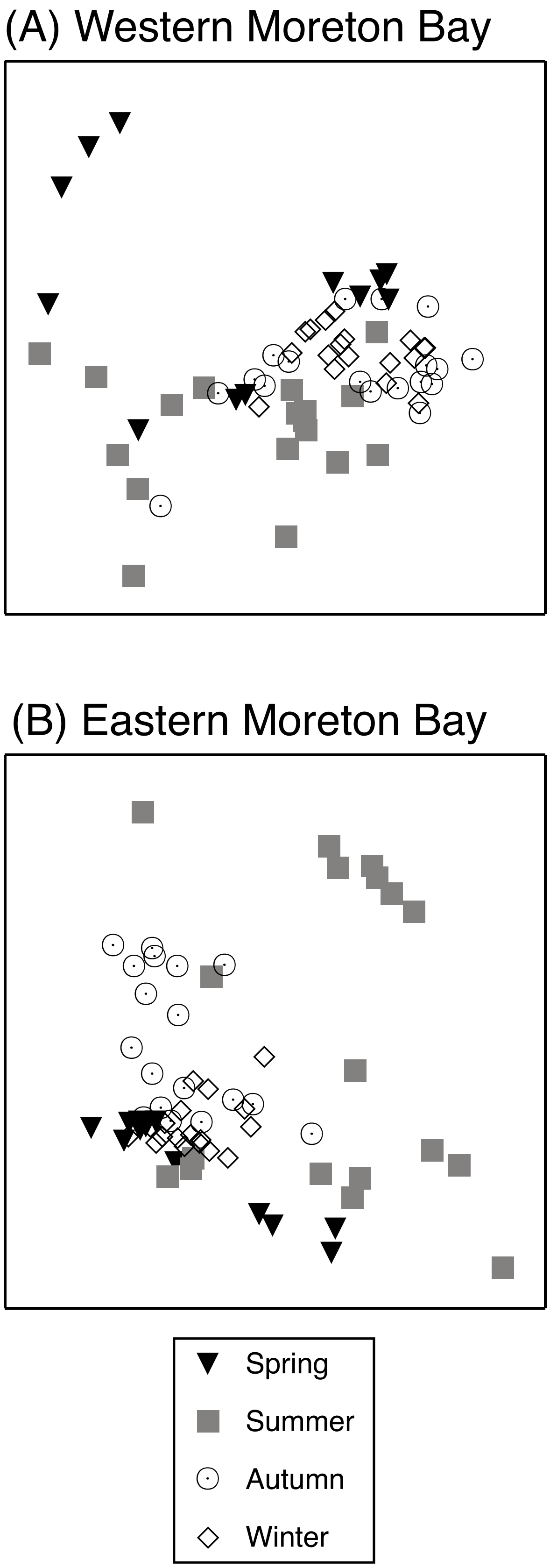
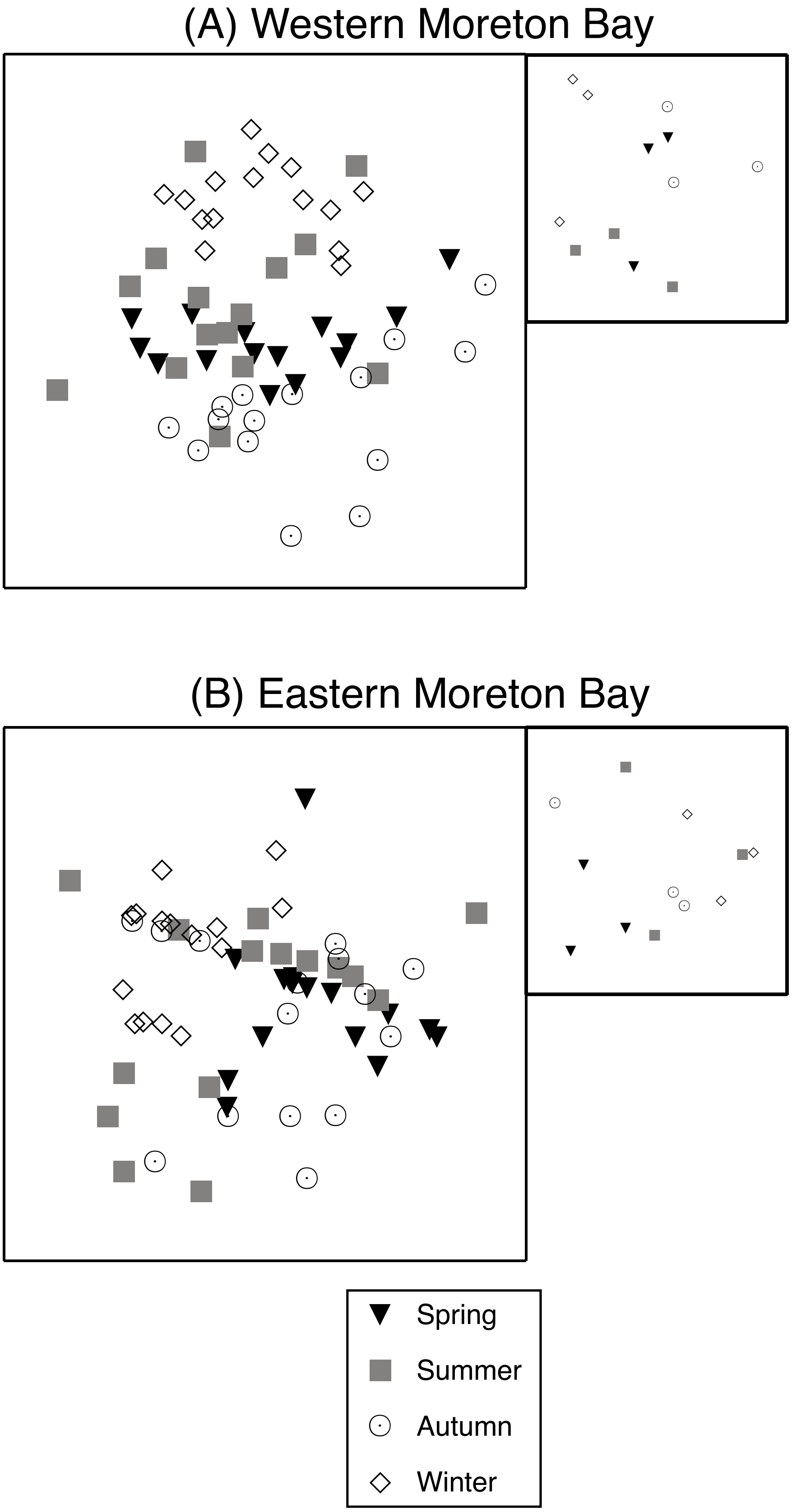
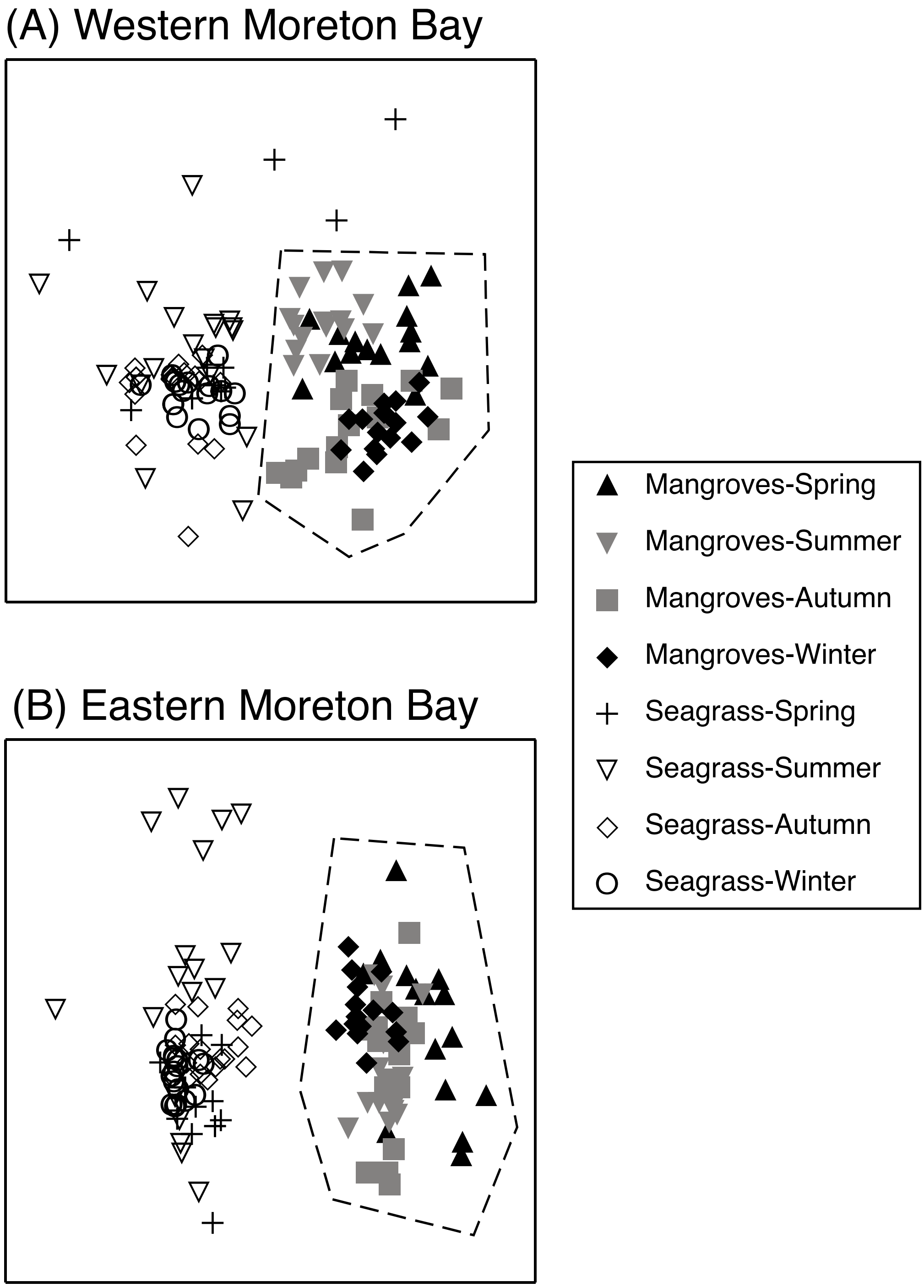


Figure 1.23: nMDS ordination of the composition of the nekton assemblage utilising intertidal seagrass beds in (A) western and (B) eastern Moreton Bay between Spring 2002 and Winter 2003. Data are the number of animals caught in 100 m^2 of seagrass. (n= 4 replicate hauls in Spring 2002, n = 6 replicate hauls in the other seasons at each of three sites in western and eastern Moreton Bay.


(A) Western Moreton Bay

(B) Eastern Moreton Bay

Figure 1.24: nMDS ordination of the composition of the nekton assemblage utilising intertidal mudflats in (A) western and (B) eastern Moreton Bay between Spring 2002 and Winter 2003. Smaller inset shows the site centroids for each season. Data are the number of animals caught within 100 m^2 of substratum. (n= 4 replicate hauls in Spring 2002, n = 6 replicate hauls in the other seasons at each of three sites in western and eastern Moreton Bay.


(A) Western Moreton Bay

(B) Eastern Moreton Bay

Figure 1.25: nMDS ordinations of the composition of the nekton assemblage utilising mangrove forests and intertidal seagrass beds in (A) western and (B) eastern Moreton Bay between Spring 2002 and Winter 2003. Data are presence/absence of species in either fyke nets (mangroves) or seine nets (seagrass) sampled in each of 3 sites in western and eastern Moreton Bay on each occasion.

Section 2: Conceptual Framework for Examining the Effects of Spatial Arrangement of Habitats on Fisheries

2.1. Summary

Effective management of estuarine and coastal fisheries resources requires detailed information on the relationships between the habitats being protected and the fisheries dependent on them. Past research of nekton has focused on comparisons of abundance and species composition between single habitats (e.g. mangroves versus seagrass or vegetated versus unvegetated habitats). These studies have provided valuable insights into the role of coastal habitats for sustaining fisheries and biodiversity but have not considered the importance of adjacent habitats to the overall value of an area. For example, fish are only able to occupy mangrove forests for a restricted amount of any high tide period. The nature of the habitats lower down the shore may be crucial to the overall value of any patch of mangrove for supporting fisheries. In this study, a new approach was used in assessing the value of estuarine habitats for fisheries and biodiversity, taking account of the spatial arrangement of different habitats within an area – or the "mosaic" of habitats within the area. The scale of the area for study is defined by the life history and biology of the species of interest. In addition to estimating the abundance, biomass and community structure of nekton (e.g. fish, crustaceans, molluscs), future work would examine the functioning of mosaics by estimating growth rates and describing the food webs in different mosaics and the characteristics of the mosaics that influence the abundance, diversity and success of the animals using different mosaics. This approach has the potential to be extended to allow much better criteria to be developed for the selection of marine reserves by managers.

2.2. Introduction

Estuarine systems comprise a large number of different types of shallow-water habitats, including seagrasses, mangroves, saltmarshes, sand and mudflats and rubble banks, that support diverse communities of plants and animals (e.g. Hatcher et al., 1989). The importance of these nearshore estuarine habitats for the survival and maintenance of biodiversity (Hockey and Branch, 1997; Brailovskaya, 1998), fisheries resources (e.g. Roberts, 1995; Kaufman and Dayton, 1997; Castilla and Fernandez, 1998; Hastings and Botsford, 1999) and ecosystem services (e.g. Costanza et al., 1997; Peterson and Lubchenco, 1997) has led to an increasing focus on the need to design and establish marine reserves and aquatic protected areas as a tool for conservation and resource management (e.g. Allison et al., 2002).

From a fisheries perspective, most research has concentrated on evaluating the relative importance of vegetated habitats such as mangroves (e.g. Bell et al., 1984; Hatcher et al., 1989; Robertson and Blaber, 1992; Laegdsgaard and Johnson, 1995, 2001), seagrasses (Orth et al., 1984; Bell and Pollard, 1989; Heck and Crowder, 1991; Edgar and Shaw, 1995) and saltmarsh (Odum et al., 1988; Orth and van Montfrans, 1990; Heck and Crowder 1991; Minello and Zimmerman 1992; Thomas and Connolly, 2001). Other habitats dominated by structural and topographical relief, including woody debris (Harmon et al., 1986; Robertson et al., 1991; Everett and Ruiz, 1993), rock and oyster reef (Lenihan and Peterson, 1998; Harding and Mann, 1999; Micheli and Peterson, 1999; Lenihan et al., 2001) and rubble (Dumbauld et al., 1993; Eggleston and Armstrong, 1995; Feldman et al., 1997; Gotceitas et al., 1997) are also known to play an important role in the recruitment and survival of commercially important species. Unvegetated habitats, although receiving less attention from a conservation and management perspective (Hoss and Thayer, 1993), also support diverse assemblages of finfish and decapod crustaceans (Lasiak, 1986; Brown and McLachlan, 1990; Kailola et al., 1993; Morrison et al., 2002).

The characteristics of vegetated habitats that are thought to contribute to their value in supporting and maintaining fisheries stocks include the provision of enhanced food supply (often associated with large levels of primary production), enhanced survival due to the provision of refuges from predation and/or enhanced food supply, and reduced physical harshness and less turbulence than in other habitats. These issues have all been well reviewed elsewhere (e.g. Orth et al., 1984; Bell and Pollard, 1989; Heck and Crowder, 1991; Butler and Jernakoff, 1999; Jackson et al., 2001) and will not be examined in detail here. Our focus is to draw attention to the need for a shift in focus in estuarine fisheries research from an approach that concentrates on the fauna of individual habitat types and makes comparisons between single habitats to one that considers the habitat as part of a mosaic of interconnected patches within a broader landscape (or seascape) made up of many different types of habitat. At present, there is almost no information about the importance of the particular arrangement of the different patches of habitat within land/seascapes on the abundance and diversity of finfish and crustacean communities.

We review the reasons for such a paradigmatic shift and propose an approach that takes into account the potential interactions that occur between different patches of habitat and their use by biota. In reviewing the extensive literature that has examined issues of the relationships between fisheries and estuarine habitats, we focus mostly on those studies that provide a mechanistic understanding of these linkages, rather than those that are primarily descriptive. It is these mechanistic studies that provide key insights into the reasons why finfish and decapod crustaceans use key estuarine habitats and therefore how they might be affected by changes in the spatial arrangement of the patches within a mosaic. These studies also provide a basis for determining the variables that might be considered as measures of differing levels of habitat quality for different mosaics.

2.3. Movement and migration among different patches of habitat

Many of the species using estuarine habitats are highly mobile and move readily between multiple habitat types regularly over a tidal cycle or during the course of their life cycle; however, surprisingly few studies have attempted to quantify the specific patterns of movements among the different patches (Beck et al., 2001; Morrison et al., 2002). Access to intertidal estuarine habitats, such as mangroves, saltmarsh and seagrass, by nekton is a function of the geomorphological and tidal characteristics at each site (Kneib, 1997b) and only occurs during a portion of each tidal cycle: many species move into intertidal areas during the flood tide, but retreat to the shallow subtidal during the ebb flow (Rozas and Odum, 1987; Hettler, 1989; Kneib and Wagner, 1994; Lin and Shao, 1999; Thomas and Connolly, 2001; but see Kneib, 1977a). For example, juvenile prawns (Penaeus merguiensis) move into mangrove forests on high tide, but use the adjacent banks downshore during the low-tide period (Robertson, 1988; Vance et al., 1996, 2002). Over longer time periods, some species are found in different parts of an estuary at different ontogenetic stages (e.g. Chubb et al., 1981; Middleton et al., 1992; Worthington et al., 1992; Gillanders, 1997), potentially exposing the animals to a variety of types of mosaics during their lifetime if the distribution of habitat types varies along estuarine gradients (e.g. Hutchings and Saenger, 1987).

Movement between different habitat types on a daily basis, or during the course of its life cycle, provides an opportunity for an animal to use different resources, such as food or shelter, found in different parts of the mosaic (e.g. Weisberg et al., 1981; Minello and Zimmerman, 1983; Boesch and Turner, 1984; Hansson et al., 1995), but it also potentially exposes it to different predators and other threats (Saunders et al., 1991). It is likely that the value of an intertidal habitat to a species will be at least partially a function of the nature of the subtidal habitat into which it must retreat during low tide. A mosaic comprising an intertidal area adjacent to a subtidal habitat that provides a high-quality refuge (e.g. Rozas and Odum, 1987; Sogard and Able, 1991; Everett and Ruiz, 1993) may be of greater overall value than a mosaic where animals leaving the intertidal with the falling tide are forced to enter an area that offers no protection from predators, such as an unvegetated mudflat. In an elegant study, Irlandi and Crawford (1997) showed that the common pinfish, *Lagodon rhomboides*, was found in greater numbers and grew faster in intertidal saltmarsh adjacent to subtidal seagrass than in saltmarsh adjacent to unvegetated mudflat. The value of the saltmarsh habitat was therefore enhanced by the location of the subtidal high-quality seagrass. Micheli and Peterson (1999) found that the proximity of saltmarsh and oyster reefs affected the survival of benthic clams on the reefs; survival of benthic clams was lower on reefs closer to saltmarsh because of the greater abundance of the predatory blue crabs (*Callinectes sapidus*) that are found in saltmarsh habitats. In both cases, the survival of prey organisms within a mosaic was affected by the spatial arrangement of the patches of habitat. The generality of such responses needs to be investigated, given the mobility of many groups using estuarine habitats and the potential for them to interact with a broad range of habitat types varying greatly in their relative quality and value.

2.4. Use of different habitats by finfish and decapod crustaceans

A major focus of past research has been on comparisons of different types of estuarine habitats in terms of their relative importance to finfish and decapod crustaceans. These studies generally fall into two broad categories: contrasts between vegetated and unvegetated habitats (e.g. mangroves v. mudflats) and contrasts between different types of vegetated habitats (e.g. seagrass v. mangroves, or seagrass beds of different species).

Numerous descriptive and experimental studies have demonstrated that vegetated habitats support a greater diversity and abundance of nekton (fish and decapod crustaceans – sensu Kneib, 1997b), and this has been the basis for the focus on protection and conservation of such areas within estuaries. This general pattern is usually explained by reference to the importance of structural complexity in mediating predator–prey interactions. As the structural complexity of the habitat increases, the intensity and success of predation generally declines (e.g. saltmarsh – Vince et al., 1976; Minello and Zimmerman, 1983; seagrass – Coen et al., 1981; Heck and Thoman, 1981; Stoner, 1982; Summerson and Peterson, 1984; Leber, 1985; Kenyon et al., 1995). Some studies have not supported this general paradigm though, suggesting that more detailed understanding of the specific links between the habitats and the biota is needed. For example, Thomas and Connolly (2001) found no clear difference in the assemblage of fish using patches of saltmarsh and adjacent unvegetated sediments, and Edgar and Shaw (1995) found that for many commercial species, seagrass beds were not more important nursery areas than nearby unvegetated areas. Importantly, there are some clear indications that the use of one habitat is affected by the proximity to another. Ferrell and Bell (1991) and Jenkins and Hamer (2001) found that the number of fish that occurred in unvegetated areas was tightly linked to the proximity of those sites to nearby patches of seagrass, suggesting that factors affecting one part of a mosaic would also influence the dynamics in the other patches (see also Heck and Thoman, 1984; Shaw and Jenkins, 1992).

Fewer studies have specifically contrasted different types of vegetated habitat (reviewed by Jackson et al., 2001). Robertson and Duke (1987) and Laegdsgaard and Johnson (1995) compared the abundance of nekton in mangrove and seagrass habitats, and in general found that the mangroves supported greater densities of fish than seagrass. Similarly, Sogard and Able (1991) compared the abundance of nekton in seagrass and saltmarsh creeks and found similar results. Irrespective of whether such patterns are true across a broader range of geographic areas and times, an important unaddressed question relates to how use of intertidal mangrove (or saltmarsh) areas is affected by the nature of the adjacent habitats into which nekton must migrate at low tide (Laegdsgaard and Johnson, 1995). Areas of high-quality mangrove, available for only a small proportion of any tidal cycle, may vary in their value as a nursery (sensu Beck et al., 2001) depending on the nature of the subtidal habitats in which the animals spend the majority of their time (Irlandi and Crawford, 1997; Jenkins et al., 1997).

Within the broad category of studies contrasting different types of vegetated habitats, important information on the features that determine the relative value of an estuarine habitat has also been obtained through comparisons of seagrass beds composed of species with different morphological characteristics. Factors such as leaf length (canopy height above the substratum), blade width and blade density have all been shown to influence the composition of the nekton community that uses seagrass beds (e.g. Stoner and Lewis, 1985; Bell and Westoby, 1986a, b, c; Middleton et al., 1984; Worthington et al. 1992; Kenyon et al. 1995; Gotceitas et al. 1997; Loneragan et al., 1998, 2001). Features providing structural complexity within mangroves, such as the density of pneumatophores and prop roots, have also been linked with differences in community composition of nekton (e.g. Thayer et al., 1987; Blaber et al., 1995; Laegdsgaard and Johnson, 2001). Again, these patterns have mostly been explained in relation to the role of structural complexity and the effects on predator–prey interactions (see references above), although there is some debate as to whether the role of predation is a proximal or indirect control on abundance (see Bell and Westoby, 1986a). Given that different seagrass beds consist of a mosaic of patches of different sizes and shapes, interspersed with unvegetated corridors (Irlandi, 1994, 1996), variation in these structural characteristics of the seagrass would suggest that the overall quality of a habitat mosaic that included mangroves and seagrass could vary considerably at different spatial scales. Experimental studies, manipulating levels of structural complexity with associated effects on other measures of habitat quality, have confirmed that these factors strongly influence the value of a patch for supporting nekton communities but these studies have all focussed on within-habitat type comparisons. No studies in marine or estuarine environments have examined the interactions between habitats or how the composition and spatial arrangement of different types of patch affect the way mosaics are used by organisms. The evidence suggests strongly that the presence of different types of patch in an estuarine mosaic will change the overall value of that mosaic because of the different resources that are provided.

2.5. Spatial arrangement of patches in a mosaic

The size and spatial arrangement of a patch of habitat may also influence its value to the animals that are using it. Irlandi et al. (1995) showed that survival of juvenile bay scallops (*Argopecten irradians*) declined in beds of seagrass that were very patchy (22% vegetation) compared with patchy (70% cover) or continuous (97%) cover, and that these effects were not due to variation in characteristics of the vegetation such as density, blade length or biomass. They attributed these results to greater access of predators to prey in very patchy areas because of increased edge-to-interior ratios compared with the more continuous beds. The unvegetated areas within the seagrass bed essentially act as corridors for movement of predators, enhancing their effectiveness at locating and acquiring prey (see also Micheli and Peterson, 1999). Similarly, growth and survival of another commercial bivalve, *Mercenaria mercenaria*, was also significantly affected by the size of seagrass patch (Irlandi, 1996, 1997). Bowden et al. (2001) found that patch size significantly affected the composition of infaunal assemblages in seagrass, although spatial variation at the regional level was relatively more important in determining the differences among seagrass beds. These novel approaches need to be applied in studies on more mobile fauna, such as the nekton that use estuarine mosaics.

2.6. A new approach – evaluating habitat mosaics for fisheries and diversity

2.6.1. Stage 1: Large-scale GIS mapping of mosaics

The consequences to fisheries from the large-scale loss of and damage to estuarine habitats (e.g. Naylor et al., 2000; Jackson et al., 2001) is now well recognised and has focused attention on the need for the establishment of marine protected areas and reserves (Margules and Nichols, 1988; McNeill, 1994; Kelleher et al., 1995). In many cases, specific types of habitats (e.g. mangroves) are protected from development and/or loss (Valiela et al., 2001) but this does not take into account deterioration of adjacent patches of habitat that may not receive the same level of protection. The ecological significance of the spatial arrangement of the different patches within a mosaic and the interactions across boundaries between patches has been well explored in terrestrial environments (e.g. Wiens et al., 1985; Hansson and de Castri, 1992) but is only now being investigated for marine and estuarine systems (Irlandi, 1994, 1996; Robbins and Bell, 1994; Irlandi and Crawford, 1997; Brooks and Bell, 2001).

The basis of our approach here is to incorporate information on the spatial arrangement, structure and condition of the patches of different habitat within a mosaic, rather than focusing just on individual types of habitats. This allows us to address the issue of whether deteriorating quality of any particular patch of habitat affects the value to fisheries of adjacent elements within the mosaic. Using this approach, we are able to ask whether the loss of or damage to a subtidal seagrass bed may have consequences for the value of a patch of intertidal mangrove, even when the latter is protected within a reserve and/or is relatively undisturbed. Answers to such questions will allow a more focused approach to deciding which combinations of habitat types are best protected within a region, given that the total area to be included within a reserve system will be limited.

Our approach is to measure and quantify the spatial extent and arrangement of the different habitats within an estuarine area, drawing on techniques and methods developed for terrestrial landscape ecology (e.g. Forman and Godron, 1986; Turner, 1989; Turner and Gardner, 1991). Spatial-pattern metrics are used to describe the characteristics of the patches of different habitat based on their extent and configuration within the mosaics. The metrics being used include area metrics (e.g. total area of habitat patch), edge metrics (e.g. patch perimeter) and connectivity metrics (e.g. nearest neighbour, proximity and fragmentation). Data on wetland distribution in south-east Queensland are being obtained from a variety of sources. Detailed methodology on the analysis and interpretation of the data can found in Manson et al. (2003).

An important component of the analysis of the spatial mapping information is the change-detection analysis on the distribution and arrangement of different mosaics through time. These analyses provide us with a measure of how much the distribution of a particular type of mosaic has changed through time and, more importantly, which mosaics have been interchanged in any area. This then provides a basis for considering the implications of any differences in the relative value to fisheries and biodiversity of the different mosaics and also provides a means of evaluating the effects of large-scale habitat fragmentation and loss within estuarine systems.

2.6.2. Stage 2 - Measures of structural complexity for habitat mosaics

Given the demonstrated importance of characteristics of habitats that provide structural complexity (see above), the differentiation and categorisation of different mosaics is based on the quantitative analysis of these measures for each of the patches within the mosaic. Detailed mapping and measurement of the physical characteristics of each of the patches or elements within each mosaic (Table 2.1) will be done to define whether each element could be considered as a high-, medium- or low-quality patch. Multivariate analysis of these physical data (e.g. nMDS – Clarke, 1993 and Canonical Correspondence Analysis – ter Braak, 1987) is used to differentiate between patches of differing quality. The core hypotheses being examined are about whether the use of these different patches is affected by the nature of the adjacent elements within the mosaic. Thus, the following three mosaics might be chosen for comparison: high quality for both mangroves and seagrass (i.e. multiple sites of high-quality mangroves with dense seagrass lower down the shore), high-quality mangroves and low-quality seagrass (multiple sites of high-quality mangroves with sparse seagrass lower down the shore) and low-quality mangroves and high-quality seagrass (multiple sites with low-quality mangroves and dense seagrass lower on the shore). A range of potentially suitable sites will be selected from the GIS database, followed by detailed ground-truthing of the physical characteristics of the patches within the mosaics.

2.6.3. Stage 3: Sampling of fish and decapods

Continuing the above example, sampling the nekton in the mangrove component of the mosaic would examine whether use of this habitat type varies as a function of the nature of the downshore habitat (high- or low-quality seagrass). Thus, multiple sites containing mangroves of similar quality would be sampled and compared on the basis of the nature of the adjacent habitats. It is important to note that this approach avoids the problem of trying to make direct comparisons of abundance and community composition between different habitat types (e.g. mangroves v. seagrass) when the methods required to sample within those habitats usually vary (e.g. Robertson and Duke, 1987; Laegdsgaard and Johnson, 1995). The specific comparisons are all, initially at least, based on an examination of whether use varies within a particular patch-type – each patch of mangroves is sampled using the same methods and experimental design. Conversely, using the same data set, we are also able to examine whether use of the seagrass habitat varies as a function of the quality of the upshore mangroves. Choosing a range of mosaics that include patches of habitat along a gradient of relative quality enhances our capacity to determine whether the composition and spatial arrangement of the elements affects use of the mosaic by the nekton.

This approach allows the specific methods and experimental design for sampling the nekton to be optimised for each of the habitat types within the mosaic and, where necessary, multiple methods to be employed in order to obtain the best estimates of community composition using the elements of the mosaic. A combination of methods has been chosen to sample the different elements of the mosaic, including: fyke nets (e.g. Lin and Shao, 1999), stake nets (e.g. Vance et al., 1996, 2002) and pop nets (e.g. Connolly, 1994; Thomas and Connolly, 2001) for within the mangroves; two different sizes of seine nets (e.g. Hindell et al., 2000) and pop nets for intertidal unvegetated and seagrass areas; and seine nets and a small otter trawl (e.g. Peterson and Skilleter, 1994) for subtidal habitats.

The design for the sampling program incorporates multiple spatial scales including comparisons between two regions in Moreton Bay (western – heavily urbanised, eastern – relatively pristine), between two separate coastal embayments in south-east Queensland (Moreton Bay and Hervey Bay) and different proximity to the shoreline (mosaics along the edge of the estuary v. those existing as isolated banks and islands within the embayments). Sampling will be done in spring/summer and winter of two successive years to test whether the different mosaics are used in the same way by different species and different ontogenetic stages of the same species. During different times of the year, depending on when particular species are recruiting, the nekton communities in some mosaics are likely to be dominated by new recruits, whereas at other times of the year the fauna will be dominated by larger individuals, possibly from several different year-classes, or different species (Connolly et al., 1999).

2.7. Functional value of different mosaics

In response to the challenges posed by Beck et al. (2001), we recognise that measures of abundance alone are not a good indication of the relative value of an estuarine habitat, or of patches of habitat within a mosaic. Determining the overall value of the different combinations of habitat therefore requires that measures of the ecological function (O'Neill et al., 1992; Fairweather, 1999) provided by different mosaics are obtained, once the initial descriptive work has been done. Trophic structure and predator–prey interactions represent important attributes of the functional aspects provided by habitats, and these may vary in response to changes in the spatial arrangement and structural complexity of the mosaics and the fauna that are using them. The growth and survival of abundant species in different mosaics would also provide an indication of overall habitat quality and function, and hence could be measured to provide other indices of the ecological functioning of a mosaic.

Table 2.1: Physical characteristics of the different habitats that are used to define patches of habitat in terms of their relative value to nekton. For example, a high-quality seagrass patch would be one with large above-ground biomass of seagrass, with long blades and large shoot density. A high-quality mangrove patch would be one with dense pneumatophores, high biomass of leaf litter and a small density of dead trees. An example of a comparison of two mosaics would contrast an area of high quality mangroves adjacent to low quality seagrass versus high quality mangroves adjacent to high quality seagrass. The focus would be on whether the utilisation by nekton of the mangroves varied as a function of the differences created by the adjacent lower-shore habitat.

Seagrass Characteristics	Mangrove Characteristics	Sediment Characteristics
Seagrass species	Density of pneumatophores	Proportion of rubble/gravel[1]
Above-ground biomass	Biomass of epiphytic algae	Sediment compaction[2]
Below-ground biomass	Biomass of leaf litter	Mean grain size
Percent coverage	Biomass of branches	Organic content of sediment
Blade length (canopy)	Biomass of macroalgae	Biomass of macroalgae
Shoot density	Density of saplings	
Macroalgal species	Density of dead/live trees	
Total biomass	Organic content of soil	
Percent coverage	Mangrove canopy cover[3]	

1=rubble/gravel is defined as particles greater than 2 mm in size
2=sediment compaction is measured using a penetrometer (e.g. Skilleter et al., 2005)
3=mangrove canopy cover measured as foliage projective cover using a crosswire sighting tube

Section 3: Classification of the Spatial Arrangement of Estuarine Habitats in Moreton Bay

3.1. Summary

Coarse-resolution thematic maps derived from remotely sensed data and implemented in GIS play an important role in coastal and marine conservation, research and management. In this section, an approach for fine-resolution mapping of land-cover types using aerial photography and ancillary GIS and ground data in a large (100 x 35 km) subtropical estuarine system (Moreton Bay, Queensland, Australia) is presented. We have developed and implemented a classification scheme representing 24 coastal (subtidal, intertidal, mangrove, supratidal and terrestrial) cover types relevant to the ecology of estuarine animals, nekton and shorebirds. The accuracy of classifications of the intertidal and subtidal cover types, as indicated by the agreement between the mapped (predicted) and reference (ground) data, was 77 – 88 %, depending on the zone and level of generalization required. The variability and spatial distribution of habitat mosaics (landscape types) across the mapped environment were assessed using K-means clustering and validated with Classification and Regression Tree models. Seven broad landscape types could be distinguished and ways of incorporating the information on landscape composition into site-specific conservation and field research are discussed. This research illustrates the importance and potential applications of fine-resolution mapping for conservation and management of estuarine habitats and their terrestrial and aquatic wildlife.

3.2. Introduction

Information on land-cover patterns at a range of spatial scales has important applications in conservation. In the context of marine conservation, land-cover and land-use data are necessary for environmental monitoring, change detection, designation of marine protected areas as well as development of multiple zoning schemes (Mumby and Harborne, 1999; Klemas, 2001; Mumby and Edwards, 2002). At the level of individual ecosystems, these data can be applied to investigate the links between biological processes, such as survival and dispersal of individuals, and physical patterns, such as landscape structure, (McGarigal and McComb, 1995). In both instances, to be applied efficiently, land-cover and land-use data need to be available in GIS (Kerr and Ostrovsky, 2003).

Estuarine ecosystems comprise shallow-water, semi-aquatic, intertidal and adjacent terrestrial habitats. Due to their variable physical environment, high habitat diversity and high primary productivity, estuaries support rich resident assemblages of vertebrate and invertebrate organisms providing important economic, conservational and aesthetic value (Hatcher et al., 1989). Furthermore, estuaries are critically important to a number of migratory species (crustaceans, Potter et al., 1991; fish, Heck et al., 2003; shorebirds, Zharikov and Skilleter, 2004; Durell et al. 2005a) at certain stages of their lifecycles. It is clear that estuaries throughout the world are becoming increasingly subjected to anthropogenic impacts that result in changes in land-cover, including dredging, loss of seagrass meadows, mangrove clearing and saltmarsh reclamation (Pye, 1995; Valeila et al., 2001; Kaldy et al., 2004; Durell et al., 2005b). These changes can both disrupt environmental flows in estuarine systems and affect organisms at various stages of their life cycles and thus compromise ecosystem functions and services (Costanza et al., 1997; Duarte, 2002). Detailed thematic maps of estuarine systems are therefore important in facilitating conservation of estuarine landscapes, increasing the understanding of the factors driving distribution of estuarine animals and investigating population-level responses to environmental changes that occur at local (within estuary) scales (Durell et al., 2005b).

Although national mapping programmes of coastal and estuarine environments exist (e.g., C-CAP, Dobson et al., 1995), they are designed to cover large spatial extents (100s to 1000s of km) of coastline and thus may not provide enough detail on landscape structure for individual estuarine sites (Klemas, 2001). Detailed, fine-resolution thematic mapping of estuarine environments is however, presently uncommon (Visser et al., 2002; Manson et al., 2003; Higinbotham et al., 2004) and it has focused on the supratidal and terrestrial zones. This limitation in coverage may make the data on landscape composition derived from such maps insufficient for the conservation and management of the species, both terrestrial and aquatic, utilising intertidal areas. Maps covering the full tidal range of estuarine cover types would be required in such cases. By comparison shallow-water marine ecosystems, coral reefs in particular, are mapped at fine resolutions much more frequently (Sheppard et al., 1995; Chauvaud et al., 1998; Mumby and Harborne, 1999; Cuevas-Jiménez and Ardisson, 2002; Mumby and Edwards, 2002). The cited reasons for the lack of detailed mapping of estuarine sites include the paucity of input data (aerial imagery, Higinbotham et al., 2004) and inflexibility in timing of data acquisition relative to tidal fluctuations (satellite imagery, Thompson et al., 2003). Also, when only satellite imagery is available, its resolution (e.g., 20 m, SPOT; 30-120 m, Landsat TM), may be too coarse for site-specific mapping of estuarine environments due to their pronounced zonation and high degree of intermingling of different cover types. Thus, in coastal environments satellite imagery is more appropriate for mapping general land-cover types (e.g., vegetated versus unvegetated, reef versus non-reef) when the information on fine-scale habitat heterogeneity is relatively unimportant (Mumby and Harborne, 1999; Klemas, 2001). Conservation, research and management conducted at the level of individual estuarine systems (extent < 100 km), will require more detailed information on the identity of specific land-cover types and their distribution. For such studies, aerial photographs are a more appropriate source of data (Higinbotham et al., 2004, but see Wang et al., 2004).

The aims of this study were to (1) develop and implement a reliable and accurate thematic mapping technique and classification scheme for a subtropical estuarine system comprising shallow subtidal, intertidal, supratidal and adjacent terrestrial zones, (2) use a clustering approach to describe the spatial distribution of habitat mosaics (landscapes types) at a scale relevant to mobile estuarine organisms, and (3) investigate ways of incorporating information on landscape composition into estuarine conservation, research and management.

3.3. Methods

3.3.1. Study Area

The study was carried out in Moreton Bay, a large embayment with semidiurnal tides (range 2.7 m) and a diversity of coastal and shallow-water habitats on the east coast of Australia (Figure 3.1). The western side of the Bay is dominated by inputs from four river systems, while the eastern side experiences more oceanic conditions (Abal et al., 1998). Moreton Bay supports several commercial and recreational fisheries and prawn aquaculture, is extensively used for outdoor recreation and has a number of wildlife conservation zones. It is also listed as an internationally important Ramsar site for migratory shorebirds (order Charadrii) (Asia-Pacific Migratory Waterbird Conservation Committee, 2001).

The area mapped included the coastal perimeter of Moreton Bay from the north-western corner of North Stradbroke Island (27.24° S, 153.26° E) to the southern extremity of North Stradbroke Island (27.43° S, 153.26° E), across to the coastal mainland (27.44° S, 153.21° E) and north to Toorbul Point in the north-western part of the Bay (27.04° S, 153.08° E) (Figure 3.1).

3.3.2. Data and methods

3.3.2.1. Remotely sensed data

Two sets of aerial photographs were used in this study: (1) fine resolution (pixel size 1.2 m), commercially processed (georeferenced, ortho-rectified and mosaiced) true-colour aerial images acquired in 2002 (MapView®) and (2) raw, scanned, true-colour aerial images taken in 1999 and 2000 (GeoScape®). Both sets were obtained from the Queensland Government Department of Natural Resources and Mines.

3.3.2.2. Image georeferencing and spatial uncertainty

GeoScape imagery was pre-processed in ArcGIS 8.3 (ESRI Inc.). Data were projected in Geocentric Datum of Australia 1994, zone 56. The MapView images did not require pre-processing and, as the more recent dataset, were the preferred source of data. Not all of the MapView images were taken within two hours of the mid-low tide, when the intertidal zone was exposed, precluding their use for mapping of inter- and shallow subtidal habitats. Therefore, a selection of GeoScape images taken at mid-low tide was georeferenced to the corresponding MapView images using 8 to 10 evenly distributed reference points. The stated horizontal root mean square error of MapView images was ± 2 m. We allowed for a horizontal error between the MapView and GeoScape imagery of ± 8 m. The cumulative horizontal error of the georeferenced GeoScape images was therefore ± 10 m. Ten GeoScape images covering islands in the central part of Moreton Bay (4% of the total area mapped) could not be georeferenced to the MapView imagery as no coverage for these areas was available. Instead this subset was georeferenced to a separate coastal vegetation database obtained from the Queensland Government Department of Natural Resources and Mines with a stated horizontal uncertainty of ± 10 m (Dowling and Stephens, 1999). In these cases, the cumulative horizontal error was ± 18 m. The horizontal accuracy in this study was within the range (1 - 22 m) reported in similar mapping endeavours (Barrette et al., 2000; Fletcher et al., 2003; Higinbotham et al., 2004).

The coastal strip in the study area is relatively flat, therefore first order polynomial transformation and nearest neighbour re-sampling algorithms were applied to the GeoScape imagery in the process of ortho-rectification. The resolution of the processed GeoScape imagery was 2.5 m, which allowed for effective mapping of objects with the smallest dimension of ≥ 7.5 m (Lillesand et al., 2004).

3.3.2.3. Intertidal habitat digitising

Automated supervised and unsupervised image processing is frequently used to create thematic maps from remotely sensed imagery (Lillesand et al., 2004). However, due to image specific idiosyncrasies (medium conditions, angle and type of the sensor, position of the sun) application of these methods requires separate classification routines to be developed for each individual image (scene). This is appropriate when a few individual scenes are to be processed (Cuevas-Jiménez and Ardisson, 2002; Wang et al., 2004). In our case, 141 aerial photographs taken under a range of conditions were required to cover the perimeter of Moreton Bay and therefore automated classification of individual images was not feasible. We interpreted and digitised the aerial photographs on-screen using the Habitat Digitizer 3.1 (National Oceanic and Atmospheric Administration, 2002) extension to ArcView 3.3 (ESRI Inc.). The Habitat Digitizer uses a hierarchical classification scheme to delineate predefined cover types. In this study the classification scheme was designed to represent ecologically meaningful, i.e. known to be utilised for a specific purpose, habitats of two broad functional groups of mobile estuarine animals, nekton (sensu Kneib, 1997) and shorebirds (e.g., Durell et al., 2005a, b) after an extensive literature review (Table 3.1). Thus, we mapped all vegetated and unvegetated intertidal cover types and the adjoining subtidal, mangrove, supratidal and terrestrial cover types approximately from 250 m down-shore from the low-tide water line to 250 m inland past the high-tide line. The size of the minimum mapping unit, i.e. an object identifiable on the ground and represented in the GIS database, was set to 500 m2 to minimize the impact of horizontal uncertainty in the data on the accuracy assessment (see below). All images were interpreted at the scale of 1:5000.

Imagery interpretation was done by one individual with extensive field experience in the system. Areas of uncertain identity were mapped as such and then specifically visited in the field. The earlier versions of the GIS database were revisited at the end of the mapping exercise to confirm that imagery interpretation was consistent across time. To further facilitate accurate imagery interpretation, 329 intertidal ground control points were surveyed in 2003 - 2004 throughout the Bay. At each point a circular plot with the radius of 15 m (0.07 ha) was visually categorized into one of the pre-defined cover types (Table 3.1). The position of each plot was recorded using a Global Positioning System (Garmin eTrex®, horizontal accuracy of ± 4 m), imported into a GIS and plotted over the aerial images. Mangroves and supratidal cover types were classified by overlaying individual images with a previously developed coastal vegetation database (Dowling and Stephens, 1999). Subtidal areas were interpreted as either bare or vegetated based on our knowledge of the area due to our inability to survey subtidal sites in this study directly.

3.3.2.4. Accuracy assessment

3.3.2.4.1. Reference Points

Terrestrial, mangrove and supratidal cover types are typically distinct and have abrupt boundaries, and therefore were unambiguously identified on the imagery (Manson et al., 2001, 2003). In addition, mangrove and supratidal cover types in the study region have been independently ground-truthed by Dowling and Stephens (1999). The accuracy assessment was thus focused on the intertidal and shallow subtidal zones.

Classification accuracy of intertidal cover types was assessed in May - August 2004 along a 45 km (intertidal area = 3,125 ha) stretch of the mainland coastline. The area contained all the mapped intertidal cover types and was assumed to be representative of the entire Bay (Figure 3.1 inset). A stratified probability sampling design (Stehman and Czaplewski, 1998) was followed whereby 30 random reference points were assigned in a GIS to the seven intertidal cover types (Table 3.1, river-bed type was ignored due to its small total area). Reference points, separated by ≥100 m, were placed > 10 m from the mapped edges of habitat patches to minimize the impact of horizontal uncertainty in the data on the accuracy assessment. Reference points were accessed on foot at low tide and surveyed without specific knowledge of their mapped identity. At each point a 15 m radius circle (0.07 ha) was characterised following the same criteria as were used for ground control points (Table 3.1). The only additional information recorded was the dominant species of seagrass (*Zostera capricorni* or *Halophila ovalis*, no other species were found) to allow for insights into possible causes of errors in image classification. Due to time limitations, 82% (173 out of 210) of reference points were accessed. Fourteen points were not accessed for sand (the least ambiguous cover type), eight for 'sporadic' seagrass, six for mud and rubble, and three for the 'mud/sand' type. To assess the accuracy of classification of the subtidal cover types we used data from a bay-wide underwater seagrass survey obtained from the Queensland Environmental Protection Agency. The survey recorded the percent cover of subtidal seagrass in 2001 at randomly selected sites using visual assessment of 5 or 20 x 2 m transect swims. A total of 168 subtidal sites intersected our mapped areas (Figure 3.1 inset). We defined the survey sites with 0 to 5 % seagrass cover as 'unvegetated subtidal' and those with > 5 % seagrass cover as 'vegetated subtidal'.

3.3.2.4.2. Classification Accuracy

A confusion matrix (Stehman, 1997; Stehman and Czaplewski, 1998; Foody, 2002) was used to provide a variety of estimates of the classification accuracy of the thematic map (Figure 3.2). For a stratified random sample of reference locations, where mapped cover types represent strata, p_{ij}, the proportion of area in mapped type *i* and reference type *j* is estimated as:

$$p_{ij} = (n_{ij} / n_{i+}) * (N_{i+} / N) \tag{1}$$

where n_{ij} is the number of reference points classified as mapped type *i* and reference type *j*, n_{i+} and N_{i+} are the sample and population sizes in stratum *i* and *N* is the total size of the area being classified in terms of the population of reference points.

The overall proportion of area correctly classified is then simply the sum of the diagonal of the confusion matrix:

$$P_c = \sum_{k=1}^{q} p_{kk} \qquad (2)$$

User's accuracy for type *i* is the conditional probability of a random point classified as type *i* by the map being classified as type *i* by the reference data:

$$P_{Ui} = p_{ii} / p_{i+} \qquad (3)$$

Producer's accuracy for type *i* is the conditional probability of a random point classified as type *i* by the reference data being classified as type *i* by the map:

$$P_{Pi} = p_{ii} / p_{+i} \qquad (4)$$

The Kappa (κ) index expresses the proportionate reduction in classification error, compared with the error of a random assignment of cover types:

$$\kappa = \frac{P_c - \sum_{k=1}^{q} p_{k+} p_{+k}}{1 - \sum_{k=1}^{q} p_{k+} p_{+k}} \qquad (5)$$

where p_{k+} and p_{+k} are the row (map) and column (reference) sums for type *k* respectively.

3.3.2.5. Modelling and validating coastal landscape types

3.3.2.5.1. Modelling coastal landscape types

Description of a landscape pattern has to be carried out at the scale (grain) relevant to the organism or process under study (McGarigal and McComb, 1995). To describe the variability of coastal landscapes, we placed non-overlapping rectangular random landscape sampling units spanning the area from 250 m above the Highest Astronomical Tide line to 100 m below the Lowest Astronomical Tide line as defined by the contour database obtained from the Queensland Department of Transport throughout the mapped environment. Landscape sampling units were oriented east-west or north-south depending on the angle of the coastline so as to be approximately perpendicular to it. The cross-shore dimension of the units (range 370 – 3,180 m) depended on the width of the intertidal zone, while the long-shore dimension was kept constant at 200 m. This scale approximates the range of daily foraging movements of individual high-tide nektonic predators and grazers (finfish, crustaceans, Vance et al., 1996; Hindell and Jenkins, 2004) and low-tide epibenthic predators, (shorebirds, Zharikov and Skilleter, 2002, 2004; Granadeiro et al., 2004) occurring in estuaries.

The landscape sampling units were categorized by the extent of nine common habitats: vegetated intertidal (all seagrass classes combined, Table 3.1), vegetated subtidal, unvegetated intertidal (all classes combined, Table 3.1), unvegetated subtidal, rubble, *Avicennia marina* mangroves (both height classes combined, Table 3.1), claypan and saltmarsh classes combined, urban/residential, and woodland. Additionally, the total number of cover types (variety) was recorded per landscape sampling unit to provide a measure of habitat richness. Absolute amounts of habitats (ha) as opposed to proportions were used as independent variables because this implicitly included the size of units into the analyses.

K-means clustering was used to separate the landscape sampling units into distinct clusters of habitat mosaics. K-means clustering seeks to partition a dataset into a predefined number of groupings so as to minimize variability within clusters and maximize variability among clusters. The number of clusters (*k*) is set *a priori*. However, generally the number of 'true' clusters in the data is not known. A reasonable approach in this case is to specify a range of *k* values based on the dataset properties and run several trials checking that meaningful output is produced in terms of cluster membership and between-cluster differences (e.g., Arnot et al., 2004). Our hypothesis was that with nine habitats there would be approximately nine clusters (landscape types), each one driven by the dominance of a particular habitat. However, we also restricted the minimum cluster membership to 1% of the dataset to allow for a meaningful interpretation of cluster properties. Six trials of the algorithm were run with *k* set to 5, 6, 7, 8, 9, and 10 on each of two random sets ($n = 250$ and $n = 515$) of landscape sampling units. Increasing the number of clusters above 10 resulted in groupings of less than 1% of the dataset. The initial cluster centres were assigned so as to maximize the between-cluster distances. Calculations were done on Euclidean distance matrices computed from raw data, which allowed the variables with greater ranges to have more impact on the cluster properties. Clustering produced similar results whether 250 or 515 random landscape sampling units were used (Y. Zharikov, unpubl. data). Only the results of runs using the 250-unit dataset are reported here.

2.3.2.5.2. *Validating coastal landscape types*

Each clustering trial was assessed using the Classification and Regression Tree algorithm (Breiman et al., 1984; De'ath and Fabricius, 2000). The algorithm fits a non-parametric hierarchical tree-based model by recursive partitioning of a dataset. The goal is to minimize the variance of the response variable at each node, but also to keep the tree as small (parsimonious) as needed to explain the variance in the data. Partitioning at each consecutive node is defined by a simple rule based on a single predictor variable after considering all possible splits. The division process continues until additional partitioning of the data does not further increase within-node homogeneity or the data become too sparse resulting in an overgrown tree. Therefore, trees are 'pruned' upward until the lowest misclassification rate is achieved using a *v*-fold cross-validation procedure (De'ath and Fabricius, 2000).

We carried out 10-fold cross-validation on the data using the discriminant-based univariate splitting technique (Loh and Shih, 1997). With regards to the K-means clustering results, the expectation is that if the grouping variable (cluster type) does not adequately partition the data, the number of terminal nodes in the best Classification and Regression Tree model will differ from the number of clusters in a given classification. All statistical tests and clustering were carried out in Statistica 6.0 software package using the Cluster Analysis and Classification Trees modules.

3.4. Results

3.4.1. Mapping

Interpretation of 141 aerial images took approximately 35 work-days by a full-time dedicated staff member and resulted in a mapped area of > 48,000 ha (Table 3.2). The subtidal zone was dominated by vegetated sediments, the intertidal by contiguous seagrass and soft unvegetated sediments, mangroves by tall *Avicennia marina*, supratidal by claypans, and terrestrial by urban/residential areas and woodlands (Figure 3.3 inset).

3.4.2. Accuracy assessment of intertidal and subtidal habitats

Accuracy assessment of intertidal areas following the original classification scheme (Table 3.1) produced an overall accuracy of 0.774 and $\kappa = 0.695$ (Table 3.3a). Reference sites that were misclassified occurred significantly closer to the edge of mapped patches (42 ± 17 m versus 81 ± 10 m, ANOVA, $F_{1,171}$, = 4.38, P = 0.038) than correctly classified sites. Confusion most commonly (21 out of 43 cases) occurred between the two vegetated types, 'patchy' and 'contiguous', and between 'mud/sand' and seagrass (Table 3.3a). Of the 11 locations mapped as 'mud/sand' but classified on the ground as 'sporadic' or 'patchy' seagrass, five were vegetated by *Zostera capricorni* and six lay in pure *Halophila ovalis* stands. *Halophila* occurred disproportionately more often (4 times, $\chi^2_1 = 8.42$, $P < 0.004$) at incorrectly mapped sites than *Zostera*. The underlying substratum at these (*Halophila*) sites was mud/sand. Reassignment of the six *Halophila* reference points from seagrass to 'mud/sand' type (Stehman and Czaplewski, 1998), however, only slightly improved the overall map accuracy ($P_c = 0.788$, $\kappa = 0.715$). Some confusion also occurred between 'patchy' seagrass and rubble (five sites). Classification of subtidal areas was 81% accurate (Table 3.3b). There was a more frequent occurrence of unvegetated sediments in areas mapped as 'vegetated', than the converse ($\chi^2_1 = 11.28$, $P < 0.001$), resulting in a relatively lower P_{Pi} for the unvegetated type.

For some applications (e.g., habitat zoning) it may be more costly to confuse distinctly different cover types, such as seagrass and rubble, than more biotically similar types (e.g., 'contiguous' and 'patchy' seagrass). Hence, a more general assessment of classification of cover types may be appropriate. Pooling the three seagrass and three unvegetated types together into two respective categories resulted in a noticeable improvement of the overall and type-specific classification accuracies (Table 3c).

3.4.3. Landscape type classification, assessment and distribution

Validation of the clustering output by Classification and Regression Tree models indicated that the overall landscape pattern in the system is driven by variation in three habitats: intertidal seagrass, *Avicennia* mangroves and urbanized/residential areas. The dataset could be reliably split into 7 - 8 landscape types (Table 3.4). Further increasing the value of k caused uneven predictive performance across individual clusters ($k = 9$) or an increased misclassification rate and a tree that was too large for a meaningful interpretation ($k = 10$). A lower number of clusters produced groupings that were either too coarse ($k = 5$, for example, did not distinguish the ecologically meaningful rubble-dominated cluster) or split individual clusters into more than one node ($k = 6$). Thus, we chose the classification with seven clusters as sufficient to represent landscape composition in Moreton Bay (Figures 3.3, 3.4). The two most abundant habitats defining a given landscape type on average occupied ≥ 42% of area per landscape sampling unit. Cluster 1 represents locations with a broad *Avicennia* belt, bare intertidal and subtidal zones and wooded terrestrial habitats. Cluster 2 represents areas with few mangroves and narrow lower intertidal zone but otherwise similar in composition to Cluster 1. Cluster 3 describes sites with high cover type diversity and overall dominated by oyster and rubble banks. Clusters 4 and 7 correspond to areas with diverse landscape composition dominated by intertidal seagrass, with the major difference being that Cluster 7 sites possess a very broad (2 km) intertidal zone. Finally, Cluster 5 describes heavily urbanized coastline with low cover type diversity while Cluster 6 represents areas with > 1 km-wide unvegetated intertidal flats fringed with mangroves and saltmarsh (Table 3.5).

3.5. Discussion

3.5.1. Classification accuracy

For the three general intertidal cover types (unvegetated, vegetated and rubble), the overall classification accuracy was above the generally accepted target of 85% (Thomlinson et al., 1999). The accuracy was lower for the seven specific intertidal cover types and the two subtidal types. Reduction in classification accuracy with an increased number of cover types was expected (e.g., Stehman, 1997; Foody, 2002; Mumby and Edwards, 2002). However, as long as the information about classification error at different levels of generalization and for different cover types is provided (Table 3.3), end-users can make appropriate decisions when applying map information for their purposes. For example, a user applying the intertidal database detailed in Table 3.3a can be 80% confident of finding a patch mapped as contiguous seagrass on the ground (user accuracy). Conversely, the probability that a rubble bank selected in the field has been correctly mapped and is available in the GIS is 86% (producer accuracy).

Most confusion occurred among vegetated cover types and a soft sediment type, and two vegetated cover types. This can be attributed to short-term temporal dynamics in intertidal vegetative cover (Abal et al., 1998) and misinterpretation of the imagery. The effect of horizontal uncertainty is likely small since we specifically acted to reduce it via methodological means (reference point placement and the minimum mapping unit size).

Of the 10 locations where 'patchy' seagrass was confused with 'contiguous' and vice versa, eight fell into areas subjected to regular disturbance by baitworm diggers (G.A. Skilleter, unpubl. data). During their operations, baitworm-diggers remove seagrass and create up to 85 m2 'pits' that overgrow within 18 months. The pit size is greater than the resolution of aerial photography allowing seagrass patchiness caused by bait collection to be detected. Bait-diggers change their areas of operation frequently. Therefore, condition of sediment surface in the imagery and on the ground was likely decoupled in the areas designated for bait collection. This confusion, however, is of little overall importance because regularly dug areas (375 ha) comprise only 8.5% of the total intertidal seagrass cover in the Bay.

True confusion was likely to have occurred at the 11 sites mapped as 'mud/sand' but in the field classified as 'patchy' or 'sporadic' seagrass. Particularly, a pattern of confusion of stands of *Halophila ovalis* with unvegetated sediment was apparent. *Halophila ovalis* is a short plant (2 - 5 cm), which, unlike *Zostera capricorni*, maintains a small total biomass and readily occupies areas prone to physical disturbance and sedimentation (Duarte et al., 1997). If the shoots of this plant become partially or completely buried, patches of *Halophila* may not be reliably distinguished from the unvegetated sediment using aerial imagery. Misclassification of at least some subtidal vegetated sites as unvegetated could also be caused by their occupation by *Halophila ovalis*, which in Moreton Bay occurs at greater depths than *Zostera capricorni* (*Abal et al., 1998). Thus, ground surveys may be required to map distribution of Halophila ovalis* accurately.

Misclassified reference sites were located 50% closer to mapped patch edges than the sites mapped correctly, suggesting gradual transitions among cover types in the benthic system (cf. Sheppard et al., 1995). It has been suggested that the ecosystems with pronounced ecotonal gradients, such as estuaries, should be mapped using 'fuzzy-edge' techniques, which allow for locations to be mapped in terms of mixed class memberships (Foody, 1996). This information may be particularly important (and the method appropriate) when small patches of one cover type (e.g., seagrass) interspersed within the matrix of another cover type (e.g. mud) may considerably enhance the habitat value of the latter (Arnot et al., 2004). However, whether a 'fuzzy-edge' model of a landscape is more valid than a Boolean model, such as applied here, needs to be determined in each specific case depending of the properties of the environment (e.g. abrupt boundaries present or not) and the nature of the question to be addressed (Arnot et al., 2004).

3.5.2 Thematic maps: level of resolution required

Maps derived from satellite data are particularly useful in conservation research and management over large (> 100 km) spatial extents (Mumby and Harborne, 1999; Kerr and Ostrovsky, 2003). While application of fine resolution satellite imagery (e.g., 1 - 4 m, IKONOS) for mapping coastal environment is currently been developed and evaluated (Mumby and Edwards, 2002; Wang et al., 2004), resolution of the most commonly used platforms (e.g., 30 m, Landsat TM) is not fine enough to map individual estuarine systems (Higinbotham et al., 2004). This is because mapping of individual objects requires their smallest dimensions to be 3 - 4 times greater than the resolution of input imagery (Lillesand et al., 2004). Consequently, long-shore strips of ecologically important habitats, e.g., mangroves, < 100 m-wide cannot be reliably mapped using satellite data (e.g., Manson et al., 2001). At some level of habitat selection, mobile estuarine animals will respond to the variability in landscape structure resolved at the scales of < 100 m (Vance et al., 1996; Irlandi and Crawford, 1997; Zharikov and Skilleter, 2002; Granadeiro et al., 2004). Thus, their habitat associations may be well-expressed at these finer spatial scales but they will not be apparent at more coarse scales of habitat generalization (Fielding and Haworth, 1995). Admittedly, fine-resolution mapped data may be required only for species with narrow distribution ranges or specific habitat requirements (Engler et al., 2004), whereas the distribution of broadly occurring species may be reliably predicted with more coarse-grain data (Seoane et al., 2004). Distributions of many estuarine animals, particularly migratory shorebirds during a nonbreeding period, are restricted to single estuarine sites where different species may display specific habitat requirements (Granadeiro et al., 2004; Shepherd and Lank, 2004). Thus, fine-resolution thematic maps may be required to model their habitat selection patterns along with other variables such as food abundance and sediment properties (Durell et al., 2005a, b).

3.5.3. GIS and field conservation research

Thematic maps of estuaries have been used to catalogue land-cover types or describe their temporal changes (Visser et al., 2002; Manson et al., 2003; Higinbotham et al., 2004). They are yet to be applied broadly in the context of conservation and management of coastal and estuarine habitats for wildlife as frequently done in terrestrial environments (Rushton et al., 2004). Below we will briefly outline potential applications of thematic maps in conservation of estuarine animals and their habitats, using the results of spatial analyses presented here as examples.

Arguably, the most frequent application of thematic maps in conservation biology is development of 'landscape-scale' predictive models of animal distribution (Manly et al., 2002). These studies examine the effect of landscape structure on a response variable, such as abundance or occurrence of individuals across a set of landscapes. Provided spatial data are available in a GIS, this application requires only the data on distribution of the study species to be obtained (McGarigal and McComb, 1995).

In our case, the individual landscape sampling units, designed to correspond to the range of daily foraging movements by mobile estuarine animals, could be sampled for abundance or occurrence of organisms of interest (e.g., juvenile stages of commercially important finfish or migratory shorebirds of conservation concern). Mechanistic links between variables describing landscape structure (McGarigal and McComb, 1995) and the distribution of individuals could then be modelled for the locations sampled and predicted for the locations not directly sampled, thus facilitating conservation and management actions (Manly et al., 2002). For example, a migratory shorebird, the bar-tailed godwit Limosa lapponica may be most abundant in the landscapes comprising vegetated and unvegetated intertidal habitats down-shore (Zharikov and Skilleter, 2002) and mangroves and saltmarsh up-shore.

When the effects of habitat structure on ecology of estuarine animals are investigated (Irlandi, 1994; Feldman et al., 1997) it is assumed implicitly that different sampling sites or treatments do not differ in their overall landscape context. This assumption may or may not be true. Data generated from thematic maps can be used to test this assumption explicitly and place field sampling sites or experimental treatments accordingly. For example, the effect of mangrove forest or saltmarsh structure on nekton abundance (Vance et al., 1996; Kneib, 1997) could be examined without the potentially confounding effect of composition of the intertidal zone on the results.

As a more general case of the above approach, thematic maps can be used to stratify field sampling within landscape settings describing variation in a particular environmental parameter. For example, the level of fragmentation of adjacent terrestrial habitats is known to affect aquatic estuarine biota (Hale et al., 2004). Thus, coastal landscapes in Moreton Bay could be ranked according to the degree of fragmentation of up-shore arboreal vegetation (e.g., Cluster 5 > Cluster 1 > Cluster 2, Table 3.5). Accounting for qualitative or ranked differences in landscape structure among locations, may be especially useful in the systems comprised of both degraded and undisturbed coastal areas (Huxham et al., 2004) as it allows to test for interactions between, and thus relative importance of, landscape structure and habitat degradation on organisms or their assemblages.

Finally, landscape-scale data can be incorporated into hierarchical designs where the effects of environmental parameters on species distribution, abundance and diversity are investigated at several spatial scales simultaneously (George and Zack, 2001). This approach accounts for the fact that habitat selection is an inherently hierarchical process and animals respond to features in their environment at a range of spatial scales, from a foraging patch to a biogeographic province (Johnson, 1980). It underscores the need to incorporate micro- scale environmental data, such as vegetation structure collected on the ground, with the data on the surrounding landscape, generated using GIS and remote sensing, within a single study. In our system it would involve obtaining sediment, seagrass and mangrove characteristics for individual landscape sampling units, and in combination with the data on landscape structure, using them to predict abundance or occurrence of study organisms. For example, a protected migratory shorebird, the eastern curlew *Numenius madagascariensis* may occur predominantly in the landscapes comprising unvegetated intertidal habitats down-shore and mangroves up-shore as long as the structure of the unvegetated sediment is suitable for burrowing decapods, the main prey of this species (Zharikov and Skilleter, 2004). Such combination of scales has been shown to improve markedly the capacity to predict distribution of animals in terrestrial systems (George and Zack, 2001). We suggest application of these principles to advance our understanding of the patterns of organismal distribution in coastal environments (Ray, 1991).

3.6. Conclusions

This study produced an accurate thematic map capturing coastal habitats across the gradient from shallow subtidal to terrestrial zones in a large (100 x 35 km) subtropical estuarine system. Our mapping approach relied only on three ingredients, aerial imagery, knowledge of one's study environment and general competency with modern GIS software. Thus, the approach can be readily applied by managers and researchers of landscape patterns in coastal environments. The 24-cover-type scheme of habitat classification represented the diversity of habitats in the system based on published information. It was also successfully used to characterise the distribution of unique landscape types in the mapped environment. The true relevance of the classification scheme and the data on landscape composition derived from it for estuarine animals will be examined and discussed in subsequent Sections.

Table 3.1. The classification scheme applied to the coastal perimeter of Moreton Bay and verbal descriptions of cover types and their ecological significance. Subtidal cover types are not included into the table as they were not directly surveyed.

COVER TYPE	DESCRIPTION, ECOLOGICAL RELEVANCE
Unvegetated intertidal	***Intertidal areas without visible seagrass cover.***
Mud	Soft, fine sediments with high proportion of silt and without pronounced grainy texture. Occurred mainly in deep embayments without strong tidal currents and along mangrove edges. Important foraging habitat for fish (Hindell and Jenkins, 2004) and shorebirds (Shepherd and Lank, 2004).
Mud/sand	A broad range of sediments containing various proportions of mud/silt and sand, relatively firm, with visible grainy texture. Occurred mostly on the intertidal flats with moderate lateral or frontal tidal currents. Inhabited by active bioturbators (Feldman et al., 1997) and are important foraging habitats for nekton (Hindell and Jenkins, 2004) and shorebirds (Zharikov and Skilleter, 2002).
Sand	Hard-packed sediments with pronounced grainy structure.
River-bed	Sections of the intertidal zone constantly subjected to fresh-water run-off from coastal wetlands. This class does not include deep river channels, which are essentially subtidal. May cause localized concentrations of waterbirds (Ravenscroft and Beardall, 2003) and serve as passage-ways for nekton (Desmond et al., 2000).
Rubble	Areas with extensive cover of dead or live shell and/or coral with coarse, hard surface, structurally complex, support diverse bivalve communities and are used extensively by nekton (Micheli and Peterson, 1998) and some shorebirds (Durell et al., 2005b).
	Intertidal areas with well developed seagrass cover
	Vegetated sediments (seagrass) perform important ecosystem functions (Duarte, 2002). They are important foraging and refuge areas for nekton and shorebirds (Zharikov and Skilleter, 2002; Heck et al., 2003). Three types were distinguished.
Contiguous seagrass	Seagrass areas with >80% seagrass cover.
Patchy seagrass	Seagrass areas with 50-80% seagrass cover.
Sporadic seagrass	Seagrass areas with 5-49% seagrass cover.
	Mangroves as per Dowling and Stephens (1999)
Mangroves	Mangrove forests are used for foraging and predator avoidance by a range of nektonic species during high tide (Vance et al., 1996; Hindell and Jenkins, 2004). Three types were distinguished.
Tall *Avicennia marina*	*A. marina*, *Avicennia*-dominated stands, >5 m canopy height.
Short *Avicennia marina*	*A. marina*, *Avicennia*-dominated stands, ≤ 5 m canopy height.
Other mangroves	Species other than *A. marina* (mostly *Rhizophora stylosa*).

Supratidal	***Habitats flooded only during the highest tides as per Dowling and Stephens (1999)***
	Supratidal habitats are important for overall estuarine ecosystem functioning (Pye, 1995). They are used for foraging by nekton (Kneib, 1997) and shorebirds (Velasquez and Hockey, 1992).
Claypan	Saline pans usually devoid of vegetation, except for occasional very sparse patches of samphire, *Sarcocornia quinqueflora* and *Sporobolus virginicus*.
Saltmarsh/sedge	Areas with short herbaceous emergent and semi-emergent vegetation (*Carex* sp).
Sand	Unvegetated areas located at or above the high tide line composed of sand (sand-spits and beaches).
Terrestrial	***Areas located above the Highest Astronomical Tide line***
	Terrestrial land-use patterns in coastal landscapes can strongly affect functioning of estuarine ecosystems via the outflow of solids, nutrients, as well as industrial, urban and agricultural wastes (Hale et al., 2004).
Agriculture	Areas (e.g., ploughed fields, orchards) with clearly visible signs of agricultural activities.
Aquaculture	Rectangular or circular ponds with artificial canals connecting them to the ocean and visible associated infrastructure (roads, buildings, aeration installations).
Woodland	Well-developed woody vegetation other than mangroves.
Fresh/brackish Water	Water-bodies located in the coastal zone not directly connected to the ocean. As no direct sampling was made they were assumed to contain fresh or brackish water.
Industrial	Sites used for manufacture and delivery of goods/services and the associated infrastructure (e.g., Port of Brisbane).
Pasture/grass	Terrestrial habitat with well-developed grassy cover.
Scrub	Patches of short, usually sparse vegetation that was visibly shorter than trees, i.e. areas falling in the woodland type.
Urban/residential	All built-up areas with variable density of dwellings with the associated infrastructure. It could also include imbedded patches of arboreal, scrub, and pasture/grass habitats.

Table 3.2: A summary of the GIS database of the coastal perimeter of Moreton Bay including the cover types, number of polygons belonging to each type, and the total area (ha) occupied by each type. The general groupings of types are given in bold.

COVER TYPE	MAP CODE	POLYGONS	AREA
Subtidal		**243**	**14484.8**
Unvegetated	60	51	11024.5
Vegetated	111	192	3460.3
Unvegetated intertidal		**683**	**4709.2**
Mud	10	233	2118.3
Mud/sand	20	311	2232.0
Sand	30	123	306.0
Creek-bed	40	16	53.0
Rubble	50	79	1688.7
Vegetated intertidal		**561**	**4430.9**
Contiguous Seagrass	110	166	2800.5
Patchy Seagrass	210	260	1229.2
Sporadic Seagrass	310	135	401.2
Mangroves		**741**	**7047.2**
Tall (>5 m) *Avicennia marina*	410	391	5187.2
Short (≤5 m) *Avicennia marina*	420	234	1106.9
Other mangroves	430	116	753.1
Supratidal		**566**	**2792.6**
Claypan	510	328	1416.3
Saltmarsh/sedge	520	185	1219.6
Dry Sand	530	53	156.7
Terrestrial		**445**	**13093.3**
Agriculture	610	16	314.8
Aquaculture	620	6	117.3
Woodland	630	204	4676.3
Fresh/brackish Water	640	44	56.4
Industrial	650	10	863.3
Pasture/grass	660	82	743.6
Scrub	670	35	243.7
Urban/residential	680	48	6078.0
TOTAL		**3318**	**48246.8**

Table 3.3: Confusion matrices and the associated accuracy coefficients for the (A, C) intertidal and (B) subtidal cover types in Moreton Bay. The intertidal matrices are based on (A) original classification scheme and (C) on three generalized cover types. Shaded cells indicate areas of most confusion between individual types: (a) $P_c = 0.774$, $\kappa = 0.695$, (b) $P_c = 0.810$, $\kappa = 0.597$, (c) $P_c = 0.876$, $\kappa = 0.767$.

Map\Reference	Contiguous	Patchy	Sporadic	Mud	Mud/sand	Sand	Rubble	p_{i+}	P_{Ui}
Contiguous	**0.346**	0.087	0.000	0.000	0.000	0.000	0.000	**0.433**	0.800
Patchy	0.023	**0.125**	0.000	0.006	0.000	0.000	0.017	**0.170**	0.733
Sporadic	0.000	0.003	**0.022**	0.000	0.001	0.000	0.003	**0.028**	0.773
Mud	0.004	0.004	0.008	**0.062**	0.012	0.000	0.004	**0.093**	0.667
Mud/sand	0.005	0.015	0.007	0.000	**0.034**	0.000	0.005	**0.066**	0.519
Sand	0.000	0.000	0.000	0.000	0.000	**0.008**	0.000	**0.008**	1.000
Rubble	0.000	0.017	0.000	0.000	0.008	0.000	**0.177**	**0.202**	0.875
p_{+i}	**0.377**	**0.249**	**0.037**	**0.068**	**0.056**	**0.008**	**0.205**		
P_{Pi}	0.917	0.500	0.592	0.916	0.616	1.000	0.862		

Table 3.3(B): (see full caption above).

Map\Reference	Unvegetated subtidal	Vegetated subtidal	p_{i+}	P_{Ui}
Unveg. subtidal	**0.274**	0.036	**0.310**	0.885
Veg. subtidal	0.155	**0.536**	**0.690**	0.776
p_{+i}	**0.429**	**0.571**		
P_{Pi}	0.639	0.938		

Table 3.3(C): (see full caption above).

Map\Reference	Vegetated intertidal	Unvegetated intertidal	Rubble	p_{i+}	P_{Ui}
Veg. intertidal	**0.577**	0.038	0.015	**0.631**	0.915
Unveg. intertidal	0.017	**0.177**	0.008	**0.202**	0.875
Rubble	0.037	0.007	**0.122**	**0.167**	0.731
p_{+i}	**0.631**	**0.223**	**0.146**		
P_{Pi}	0.914	0.794	0.837		

Table 3.4: Results of fitting Classification and Regression Trees to the six K-means classifications of the landscape composition data where *k* refers to the predefined number of clusters, tree size corresponds to the number of terminal nodes, misclassification rate (± SE) is based on 10-fold cross-validation of the data, and the top variables are the three most influential variables driving the splitting criteria in the tree specified in the Classification and Regression Trees output (AM = *Avicennia marina*, UI = unvegetated intertidal, UR = urban/residential, VI = vegetated intertidal).

k	Tree size	Misclassification rate	Top variables
5	5	0.180±0.024	VI, AM, UI
6	9	0.136±0.022	UR, AM, VI
7	7	0.168±0.022	UR, AM, VI
8	8	0.168±0.023	UR, AM, VI
9	9	0.156±0.023	AM, UR, VI
10	12	0.208±0.026	VI, UR, AM

Table 3.5: Characteristics of landscape types (clusters of landscape sampling units) in the coastal landscape of Moreton Bay based on their composition (here given as proportion of total area occupied by a particular habitat) and variety (mean number of individual cover types per landscape sampling unit). Mean total area (ha) and the length of the intertidal zone (m) of landscape sampling units are given to enhance interpretability of the data. Shaded cells with bold lettering indicate the primary habitat while boxed cells with bold lettering indicate the secondary habitat in terms of their contribution to the total area.

Cluster	**1**	**2**	**3**	**4**	**5**	**6**	**7**
Variety	7.15	6.93	8.11	8.51	5.16	7.18	9.00
Veg. intertidal	0.02	0.08	**0.14**	**0.40**	0.07	0.01	**0.60**
Unveg. intertidal	**0.15**	0.14	0.05	0.08	**0.22**	**0.44**	0.09
Rubble	0.03	0.03	**0.40**	0.02	0.03	0.00	0.00
Avicennia marina	**0.44**	0.13	0.10	0.10	0.03	**0.26**	0.10
Claypan+saltmarsh	0.06	0.07	0.01	0.08	0.00	0.10	0.03
Unveg. subtidal	0.11	**0.16**	0.13	0.04	0.15	0.10	0.01
Veg. subtidal	0.02	0.08	0.01	**0.12**	0.05	0.00	**0.11**
Woodland	0.13	**0.26**	0.12	0.08	0.02	0.05	0.01
Urban/Residential	0.04	0.05	0.03	0.09	**0.43**	0.04	0.04
Number of units	20	67	9	53	75	17	9
Total area	23.45	12.01	34.86	20.69	14.76	31.06	45.80
Intertidal width	823	251	1393	684	388	1203	1940

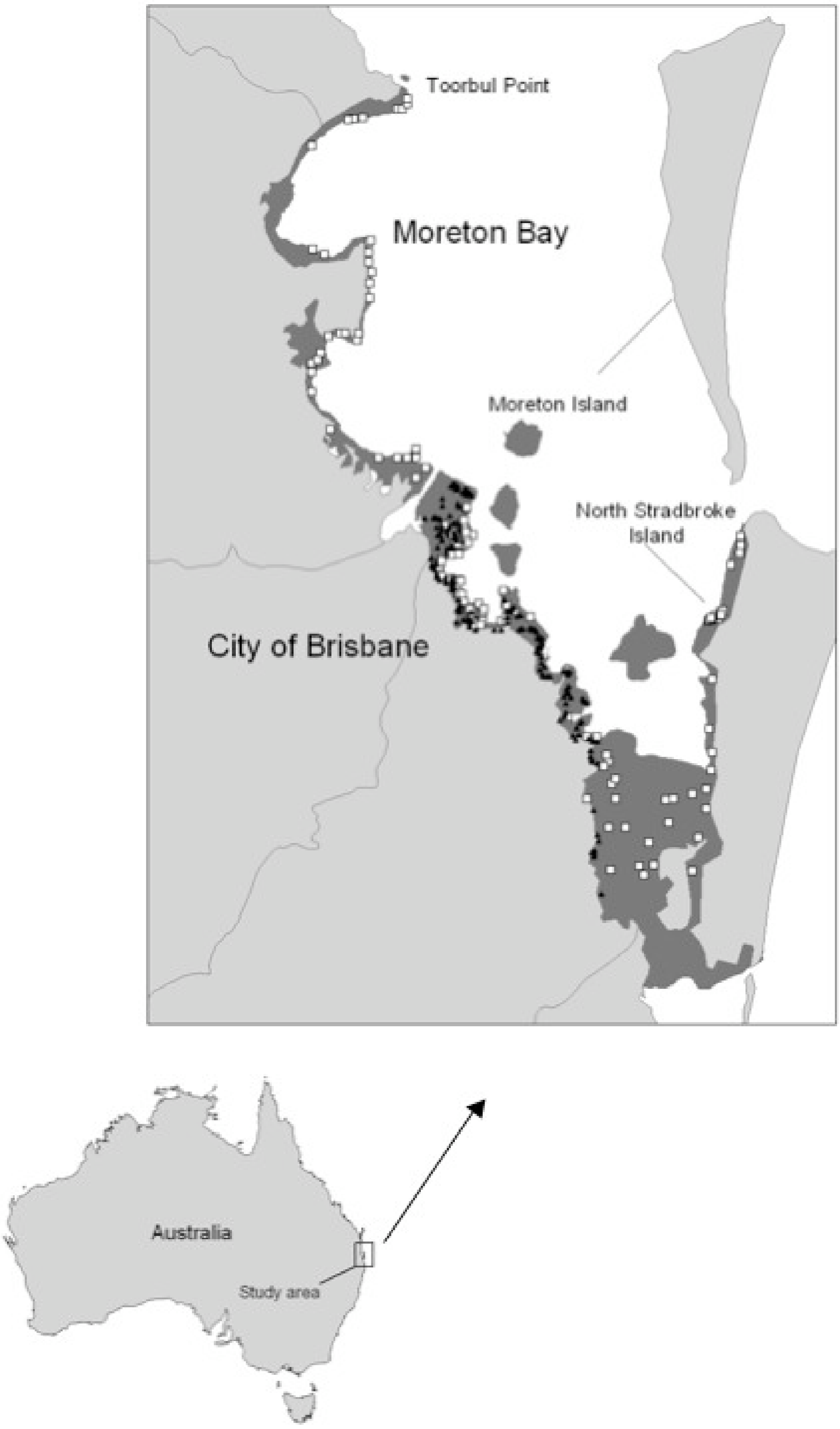
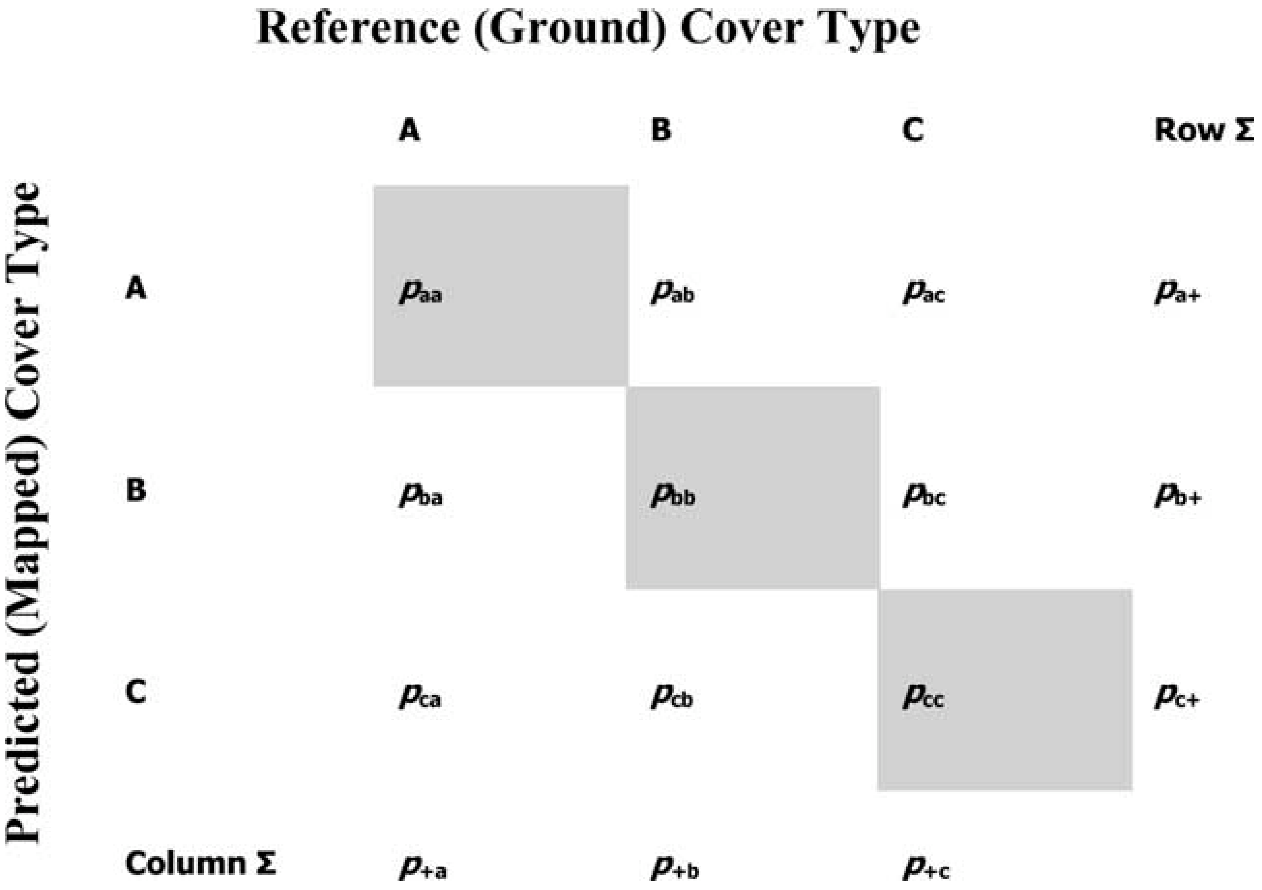


Figure 3.1: Geographic location of Moreton Bay and distribution of reference (accuracy assessment) points (white squares – subtidal zone, black triangles – intertidal zone) The mapped sections of the coastal perimeter of the Bay are given in dark grey.



Reference (Ground) Cover Type

Predicted (Mapped) Cover Type

	A	B	C	Row Σ
A	p_{aa}	p_{ab}	p_{ac}	p_{a+}
B	p_{ba}	p_{bb}	p_{bc}	p_{b+}
C	p_{ca}	p_{cb}	p_{cc}	p_{c+}
Column Σ	p_{+a}	p_{+b}	p_{+c}	

Figure 3.2: A confusion matrix where the shaded diagonal contains the cases of agreement while the cells below and above it represent disagreements between mapped (predicted) and reference (ground) cover types. Note that qij is the proportion of area in mapped cover type i and reference cover type j; qi+ (row sum) is the proportion of area mapped in cover type i; q+i (column sum) is the true proportion of area in cover type i.

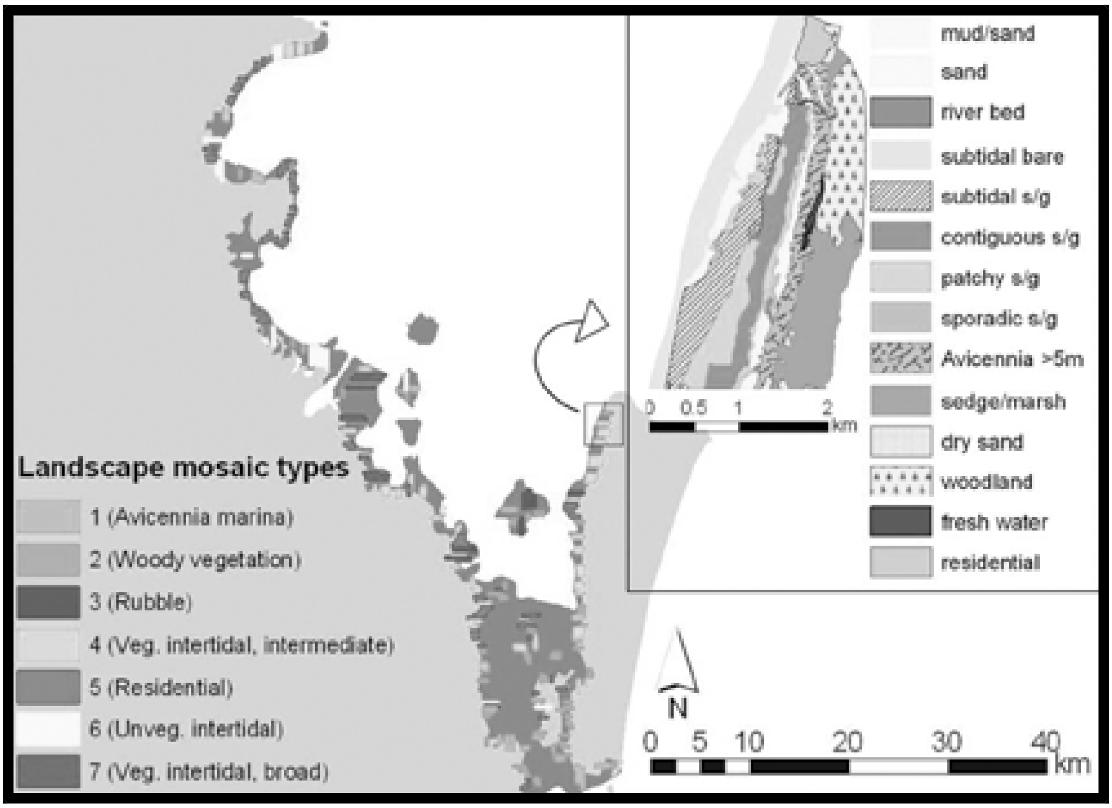


Figure 3.3: Distribution of the seven landscape types in Moreton Bay defined in clustering analyses. Legend reflects the primary cover type for a given landscape type. Inset shows a section of the coastline with cover types as defined in the thematic map (not all 24 cover types are shown).

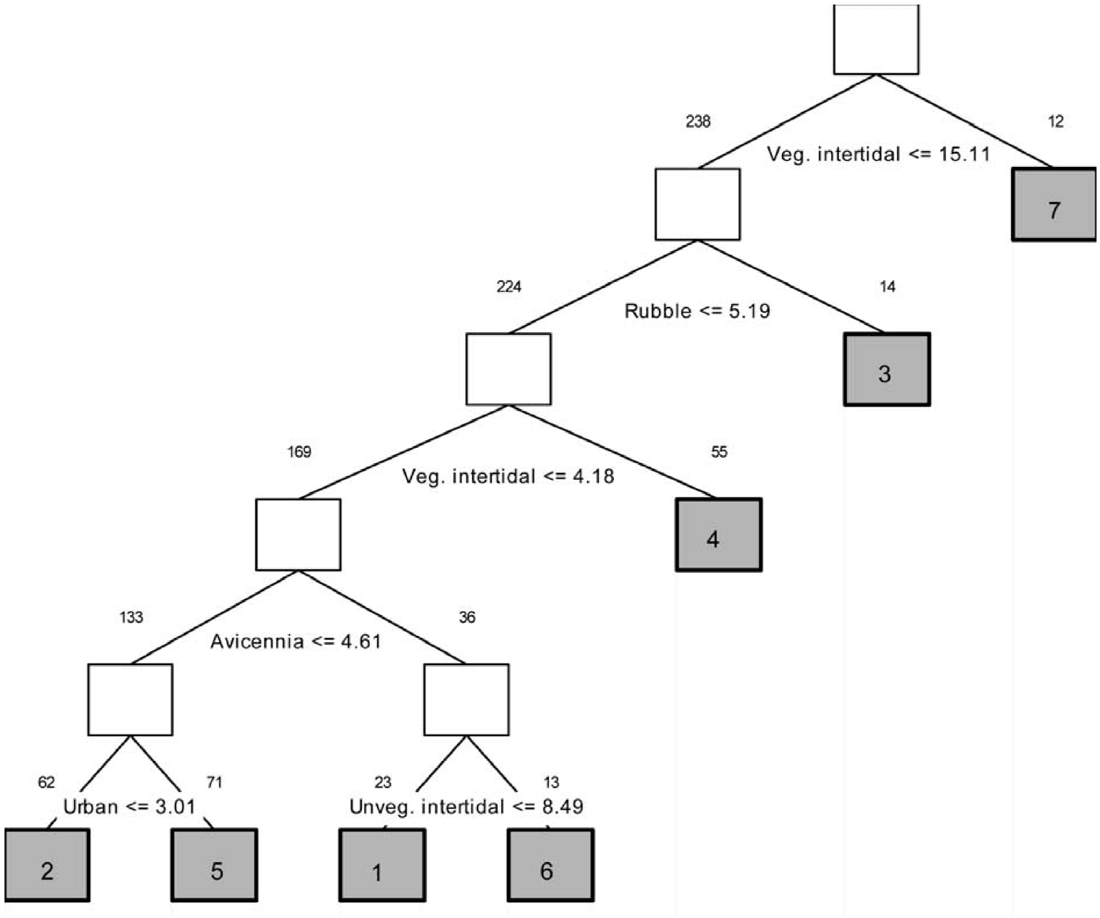


Figure 3.4: The Classification and Regression Tree modelling the cluster (landscape type) as the dependent variable against the areal extents (ha) of nine coastal habitats plus habitat richness as independent variables. Shaded blocks represent the terminal nodes with Cluster numbers they correspond to. The number of assigned cases (landscape sampling units) is given above each node. Split rules are given under the parent nodes. For example: the firsttop-right terminal node corresponds to Cluster 7 and contains all the units with the amount of intertidal seagrass >15.11 ha; the second top-rightterminal node (Cluster 3) contains all the units with the amount of seagrass cover 615.11 ha and the amount of rubble >5.19 ha.

Section 4: Effects of Proximity Between Seagrass and Mangroves on the Abundance of Commercial Prawns

4.1. Summary

Penaeid prawns were sampled with a small seine net to test whether catches of postlarval and juveniles in seagrass were affected by the distance of the seagrass (mainly *Zostera capricorni*) from mangroves and the density of the seagrass in a subtropical marine embayment. Sampling was replicated on the western and eastern sides of Moreton Bay, Queensland, Australia. Information on catches was combined with broad-scale spatial information on the distribution of habitats to estimate the contribution of four different categories of habitat (proximal dense seagrass, distal dense seagrass, proximal sparse seagrass, distal sparse seagrass) to the overall population of small prawns in these regions of Moreton Bay. The abundance of *Penaeus plebejus* and *Metapenaeus bennettae* was significantly and consistently greater in dense seagrass proximal to mangroves than in other types of habitat. Additionally, sparse seagrass close to mangroves supported more of these species than dense seagrass further away, indicating that the role of spatial arrangement of habitats was more important that the effects of structural complexity alone. In contrast, the abundance of *P. esculentus* tended to be greatest in sparse seagrass distal from mangroves compared with the other habitats. The scaling up of the results from different seagrass types suggested that proximal seagrass beds on both sides of Moreton Bay provided by far the greatest contribution of juvenile *M. bennettae* and *P. plebejus* to the overall populations in the Bay.

4.2. Introduction

The spatial arrangement of patches of habitat within a broader landscape is known to influence a range of ecological processes that determine the distribution of local populations and communities (Forman and Godron 1986). Attributes such as the size, complexity, levels of fragmentation and isolation and the relative position of patches within the landscape affect the structure, composition and dynamics of communities and the functional linkages between adjacent patches of habitat (Kareiva 1990; Dunning et al. 1992; Johnson et al. 1992). Although landscape ecological theory has been widely applied in terrestrial systems (Hansen and di Castri 1992; Hanski and Gilpin 1997), the influence of the spatial arrangement of habitats on the dynamics of marine communities remains relatively unexplored (for exceptions see Robbins and Bell 1994; Irlandi and Crawford 1997; Eggleston et al. 1998, 1999; Micheli and Peterson 1999; Hovel and Lipcius 2001, 2002).

Where landscape concepts have been applied in marine systems, specific habitats have largely been examined separately, with studies focussing on the importance of patch size, structural complexity and fragmentation to the population dynamics of key species (e.g. Parish 1989; Bell et al. 1995; Irlandi et al. 1995 1999; Irlandi 1996, 1997; Eggleston et al. 1998, 1999; Acosta 1999; Turner et al. 1999; Bowden et al. 2001; Hovel and Lipcius 2001, 2002). Few studies have examined marine systems as a series of interconnected patches of different habitats, linked actively through the movement of organisms and passively through the waterborne transport of propagules and the outputs of primary production (for exceptions see Rozas and Odum 1987; Irlandi and Crawford 1997; Micheli and Peterson 1999; Raposa and Oviatt 2000; Nagelkerken et al. 2001; Dorenbosch et al., 2004).

The capacity of an organism to utilise resources that occur in adjacent patches of habitat depends on the nature of the boundary and the distance between the patches (Kareiva 1990; Dunning et al. 1992; Johnson et al. 1992; Puth and Wilson 2001). The relative proximity and connectivity between adjacent habitats in terrestrial systems affects the movement and dispersal of populations and the degree of resource exchange between the habitats (Forman and Godron 1986; Kareiva 1990; Dunning et al. 1992; Johnson et al. 1992; Acosta 1999; Puth and Wilson 2001). Animals moving between patches of habitat that are close together traverse shorter distances, are exposed to predators for briefer periods and often experience smaller rates of predation than when patches are more distant from each other (Dunning et al. 1992; Johnson et al. 1992; Puth and Wilson 2001). They may also move more frequently and further into adjacent patches of habitat that are closer together (Kareiva 1990; Johnson et al. 1992) and the relative proximity between adjacent patches may influence the way in which local communities respond to variation in the size of patches, their structural complexity and degree of fragmentation (Kareiva 1990; Dunning et al. 1992; Johnson et al. 1992). The transport of resources between patches by wind or water also depends on the degree of patch separation, with a greater exchange of resources being possible when patches are close together than when they are further apart (Robbins and Bell 1994; McIvor and Rozas 1996; Koch and Madden 2001).

Estuarine landscapes comprise a mosaic of different habitats, including seagrasses, mangroves, saltmarshes, oyster reefs and rubble banks and unvegetated sand and mudflats (Skilleter and Loneragan 2003). Mangroves and saltmarsh, found high in the intertidal, potentially provide resources to communities in adjacent seagrass beds lower down the shore and in the subtidal, directly through the outwelling and demineralization of mangrove detritus (Alongi 1990; Koch and Madden 2001) and indirectly through the movement of nekton between the habitats (Irlandi and Crawford 1997; McIvor and Rozas 1999; Micheli and Peterson 1999). Nekton in seagrass beds close to mangroves or saltmarsh may therefore receive greater amounts of detritus and other food than nekton in beds that are further away (Alongi 1990; McIvor and Rozas 1999). Exported detritus from high-shore habitats also supplies additional physical structure in adjacent seagrass beds providing greater shelter to nekton communities in these areas (Daniel and Robertson 1990).

When inundated, mangroves and saltmarshes provide nekton from seagrass beds lower on the shore with rich and productive areas in which to forage and escape predators (Robertson and Duke 1987; Thayer et al. 1987; Chong et al. 1990; Blaber et al. 1992; Laegdsgaard and Johnson 2001) but these habitats are only available to nekton populations for limited tidal periods. Many nekton species are highly mobile, moving upshore into intertidal mangroves and saltmarsh with the rising tide and withdrawing into seagrass lower on the shore as the tide recedes (Rozas and Odum 1987; Kneib and Wagner 1994; Irlandi and Crawford 1997; Thomas and Connolly 2001; Vance et al. 2002). Movement between these adjacent habitats provides animals with access to different resources and an altered risk of predation (Robbins and Bell 1994; Micheli and Peterson 1999; Beck et al. 2001) but this may have to be traded against the risks from predation and physical stress associated with crossing unvegetated tidal flats to reach the upper shore forests (Irlandi and Crawford 1997; Micheli and Peterson 1999). The proximity of a patch of intertidal seagrass to nearby mangrove forests is likely to be an important factor affecting the dynamics of nekton communities occurring within seagrass beds but there have been no studies that have specifically examined this question (but see Raposa and Oviatt, 2000 who examined this for saltmarsh and seagrass).

The postlarval and juvenile stages of many species of penaeid prawns (=shrimp) occupy markedly different environments to the adults (Dall et al. 1990). The specific habitats utilised by the different stages varies among species. For example, the juveniles of banana prawns (*Penaeus merguiensis* and *P. indicus*) are found almost exclusively in mangrove-lined creeks (Staples et al. 1985; Vance et al. 1998; Rönnbäck et al. 2002), while the juveniles of eastern king (*P. plebejus*) and tiger (*P. esculentus* and *P. semisulcatus*) prawns are most abundant in areas of seagrass (Young and Carpenter 1977; Loneragan et al. 1998; Masel and Smallwood 2002). Detailed work on those species closely linked to seagrass has demonstrated the importance of the structural characteristics, such as density and blade length, in determining the abundance of the postlarval and juvenile stages (e.g. Loneragan et al. 1998; Kenyon et al. 1995, 1997), but the potential influences of adjacent and nearby habitats, such as mangroves, has not been investigated.

Here we describe the patterns of abundance of juvenile prawns found in intertidal seagrass beds positioned at different distances from nearby mangrove forests in Moreton Bay, a large embayment in sub-tropical, eastern Australia. We tested whether the proximity between seagrass and mangroves affected the utilisation of the seagrass by prawns and whether the effects of proximity were influenced by the structural complexity of the seagrass beds. We then scaled up the site-specific estimates of abundance in order to estimate the habitat-specific standing stocks of post-larval and juvenile prawns and the proportional contribution of different types of seagrass bed to these total standing stocks at a much larger scale.

4.3. Methods

4.3.1. Study sites

This study was done in Moreton Bay, Queensland, a large subtropical embayment on the east coast of Australia (Lat. 27° S, Long. 153° E, Figure 4.1). The bay has extensive seagrass coverage (Hyland et al. 1989), with meadows dominated by *Zostera capricorni* Ascherson. Moreton Bay has a semi-diurnal tidal pattern with a range of 1.5 to 2.0 m during a full monthly tidal cycle (Dennison and Abal 1999). Salinity ranges from 28-34 ‰, with little seasonal variation on the eastern side of the embayment (Gabric et al. 1998), but salinities on the western side are more variable because of the influence of several major river systems (Young 1978; O'Brien 1994). Moreton Bay contributes a significant proportion to the total commercial catch on the east coast of Australia for the eastern king prawn (*Penaeus plebejus*) (Trainor 1990, 1991) and provides critical habitat for postlarval and juvenile stages of *P. plebejus*, *P. esculentus* (brown tiger prawn) and *Metapenaeus bennettae* (bay or greasyback prawn) (Young 1975, 1978, Young and

Carpenter 1977; Williams 1991). Postlarvae and juveniles of *Penaeus plebejus* (Young 1975; Young and Carpenter 1977; Coles and Greenwood 1983), *Penaeus esculentus* (Young 1975; Young and Carpenter 1977; O'Brien 1994) and *Metapenaeus bennettae* (Young 1978) are abundant in shallow, nearshore waters of Moreton Bay in spring and summer (September-February).

4.3.2. Spatial variation

To determine whether the abundance of prawns varied with the proximity of seagrass beds to adjacent mangrove forests and/or as a function of the structural complexity of the seagrass, four different categories of seagrass bed were examined:

(1) dense seagrass beds proximal to mangroves,
(2) dense seagrass beds distal to mangroves,
(3) sparse seagrass beds proximal to mangroves and
(4) sparse seagrass beds distal to mangroves.

Three sites in each category of seagrass bed were surveyed in both the eastern and western regions of Moreton Bay (a total of 24 sites). Sites from each region were included in the sampling design to determine whether any differences in utilisation of the different seagrass beds were consistent across the bay. The western region of Moreton Bay is affected by urban and industrial development, whereas the eastern region is relatively undeveloped (Dennison and Abal 1999) so the quality of the environment varies considerably on either side of the Bay. The distribution of the four categories of seagrass bed also varied considerably in different regions of Moreton Bay and the selection of sampling sites had an added criterion that beds belonging to the different categories should be spatially interspersed as much as possible (Figure 4.1) to reduce any spatial confounding of comparisons among the treatments (e.g. Oviatt and Raposa 2000). The measures of proximity and density therefore varied between the two regions. Proximal patches were categorized as those from 0-20 m from adjacent mangrove forests in the western bay and 0-40 m in the eastern bay. Distal beds were separated from adjacent mangroves by 90-150 m of unvegetated sediment in the western bay and 150-350 m in the eastern bay. Sparse sites had 30-40% seagrass coverage in the western bay and 20-30% cover in the eastern bay. Dense sites had 70-90% seagrass coverage in the western bay and 60-70% cover in the eastern bay.

Sampling of the different sites was randomised through time, except that the sites in western Moreton Bay (September-October 2002) were all sampled before those in eastern Moreton Bay (October-November 2002). Two sites could be sampled on any day, so a set of 12 sites (3 sites in each of the 4 categories of seagrass) required 6 days of sampling. Over the required period, sampling of the 12 sites was organised so that any effect of short-term (daily) variation in abundance was randomised across all treatments, avoiding confounding of any comparisons among these treatments.

Four replicate samples were collected at each site with a 6 m seine net (1 mm mesh with 2 m drop) hauled for 25 m parallel to the adjacent mangrove fringe, with a constant mouth width of 4 m, sampling a total area of 100 m^2 for each haul. Replicate hauls were positioned on a transect along the shoreline, each haul separated by a distance of 10 m. Sampling was done within two hours either side of the daytime high tide during the six days spanning either the full or new moon between September and November and at a relatively constant water depth (0.7 to 1.0 m). Although many species of prawns bury in the substratum during the day

(Dall et al. 1990), this behaviour is not evident in the postlarvae or small juveniles of some species (Kenyon et al. 1995, Liu and Loneragan 1997) and the catch rates of these individuals did not differ between day and night sampling in Moreton Bay (Guest et al. 2003). Samples were frozen until they could be examined in the laboratory where individuals were identified to species, counted and measured (carapace length) to the nearest mm.

Due to bad weather, sampling at one site in the dense, distal seagrass category in western Moreton Bay was disrupted. Catches at this site were dominated by large amounts of seagrass and algal debris and it was not clear whether the efficiency of the seine net was similar to that at the other sites. Data from this site were excluded from analyses, leaving an unbalanced design for the western Bay. Data from the western Bay were analysed with three factor ANOVAs, with Proximity and Density treated as fixed factors and Sites as a random factor (nested within Proximity x Density). Two sites, selected at random, from each category were included in these analyses. Data from the eastern Bay were analysed with the same ANOVA design, except all three sites in each category of seagrass bed were included in the analyses. Data were log transformed to meet the assumptions of homoscedasticity of variances after Cochran's test (Underwood 1981). In all cases, transformation removed heteroscedasticity of variances.

4.3.3. Temporal variation

Two sites from each of the four categories of seagrass bed in the western region of Moreton Bay were sampled more intensively to determine whether the utilisation of seagrass beds by prawns was consistent through time. These eight sites were sampled in two blocks between the 30th January and 2nd February 2003 and then again between the 7th to 10th of February 2003 on the full and new moon. Samples were collected by seine nets as described above, except six replicate samples were collected at each site compared with four replicates previously. Two sites, selected at random, were sampled on each day in each sampling block. Samples were frozen until they could be examined in the laboratory where individuals were identified to species, counted and measured. Data (log transformed) were analysed with four factor ANOVAs, with Time (a=2, fixed), Proximity (b=2, fixed) and Density (c=2, fixed) as fixed factors and Sites as a random factor (random, d=2).

4.3.4. Characterising habitat

To describe the physical characteristics of the vegetation and sediments within each category of seagrass bed in western Moreton Bay, samples were collected along the transect where seining had been done. Four randomly positioned 1 x 1 m quadrats were examined in two blocks, 70 and 140 metres respectively along the transect. The percentage cover of macrophytes was recorded for all species of seagrass and algae within each quadrat using the point-intersect method. Ten measures of sediment compaction were taken around each quadrat using a hand-held penetrometer, constructed from a 0.50 m (279.1 g) solid metal probe. The penetrometer was dropped from a constant height of 1.3 m within a loose piping and the depth of penetration into the substratum was recorded. Two 15 cm diameter by 15 cm deep cores of seagrass were collected from within each replicate quadrat and frozen for examination in the laboratory.

The total number of shoots for each seagrass species, the number of blades per shoot (up to 10 shoots) for *Zostera capricorni* and the length of the longest blade per shoot (up to 10 shoots) for each species was recorded in each core. Vegetation was separated into above and below-ground components for each species of seagrass and algae. These components were oven dried at 75 °C for 72 h and then weighed to determine above and below ground biomass for each species of seagrass and algae. An 8 cm diameter by 7 cm deep core of sediment was collected from within each replicate 1 x 1 m quadrat and frozen for characterisation of the substratum at each site. Organic material was digested from the sediment cores with a solution of 42 g/L Sodium Hypochlorite (NaOCl) prior to the sediments being sieved. Samples were oven dried at 75 °C for 72 h and weighed, then wet sieved across a 2.0 mm sieve to determine the gravel component in the sediments. Only gravel content was quantified because this provides an important form of physical structure for epibenthic animals (e.g. Arnold 1984; Dumbauld et al. 1993). Data on the physical characteristics of the habitat at each site were analysed by principal components analysis on normalized data using the Euclidean distance measure and shown as a two-dimensional ordination.

4.3.5. Calculating habitat-specific standing stocks

To assess the broader relevance and importance of the differences in utilisation of the different types of seagrass beds by prawns (see Results), we calculated the total area for each of these patch types and scaled up the site specific estimates of abundance of prawns (= average density of prawns x estimated area for each seagrass type) for two geographic sub-regions of Moreton Bay, covering a total combined coastline of 160 km. The area in western Moreton Bay extended from the mouth of the Brisbane River (27° 22′S and 153° 09′E) to the mouth of the Logan River (27° 42′S and 153° 19′E) (Figure 4.1). Eastern Moreton Bay extended from Amity Point (27° 24′S and 153° 26′E) to the southern tip of North Stradbroke Island (27° 44′S and 153° 25′E). These sub-regions covered the areas in which sampling had been done. We also calculated the total area of each of the seagrass patch types and the scaled estimates of abundance of prawns in the southern and north-western regions of Moreton Bay, as a first approximation of the overall value to prawns of the different types of habitat for the entire Moreton Bay region.

The total area of seagrass was estimated by developing a GIS database (1:5000) of coastal habitats for Moreton Bay using 1999-2002 high-resolution aerial imagery (Zharikov et al. in preparation). In this database, habitats were classified into the following categories: mangrove forests (*Avicennia marina* and other mangroves), unvegetated intertidal habitats, and seagrass habitats - continuous (≥ 80% cover), patchy and sparse (< 80% cover). The accuracy of the classification of seagrass patches in the GIS was tested by comparing the GIS classification with the data from each of the sampling sites in the study, which gave > 82% agreement between the GIS and field classifications of seagrass.

4.4. Results

4.4.1. Spatial variation

Four species of prawns, *Metapenaeus bennettae, M. ensis*. *Penaeus plebejus* and *P. esculentus*, were sampled from the western and eastern regions of Moreton Bay, but *M. ensis* and *P. esculentus* were found only in small numbers in this part of the study so were not analysed further. The sizes of *M. bennettae* ranged between 2-9 mm carapace length (CL) with a mean (± 1 SE) of 5.06 ± 0.09 mm CL (n = 526). The sizes of *Penaeus plebejus* in the western Bay ranged between 2-10 mm CL, with a mean of 3.75 ± 0.01 mm CL (n = 281), while in the eastern Bay they ranged from 2-11 mm CL, with a mean of 4.01 ± 0.06 mm CL (n = 1109).

The catches of both *Metapenaeus bennettae* (Fig. 2A) and *Penaeus plebejus* (Figure 4.2B) in western Moreton Bay, differed significantly with position of seagrass bed (Table 4.1). Catches of *M. bennettae* were largest in proximal dense seagrass beds, followed by proximal sparse and distal dense beds (Figure 4.2A). Catches of *P. plebejus* in western Moreton Bay were significantly larger in the proximal than distal beds (ANOVA, main effect Proximity $P < 0.05$; Figure 4.2B) but there was no significant effect of seagrass density or significant interaction between Proximity and Density. In eastern Moreton Bay, catches of *Penaeus plebejus* were significantly larger in the proximal than the distal beds (ANOVA, main effect Proximity $P < 0.04$) and in the dense than the sparse beds (ANOVA, main effect Density $P < 0.03$) (Figure 4.2C), but there was no significant interaction between Proximity and Density (Table 4.1).

4.4.2. Temporal variation

There was a significant three-way interaction between Proximity, Density and Sampling Period (ANOVA, 3-way interaction, $P < 0.04$, Table 4.2) in the catches of *Metapenaeus bennettae*. The effect of proximity was more important that density though, with proximal sparse beds supporting more *M. bennettae* than distal dense beds, while distal sparse beds supported the smallest densities (Figure 4.3A). These effects were consistent through time, with the contribution of Sampling Period in the significant 3-way interaction arising from changes in whether there were more prawns caught at time 1 or 2 for the different bed types (Figure 4.3A).

Although mean catches of *P. plebejus* in western Moreton Bay were generally small (< 12 per 100 m^2), catches also differed significantly among the different bed types (ANOVA, 3-way interaction, $P < 0.002$, Table 4.2). However, the pattern was more complicated than for *M. bennettae*. At Time 1, there was significantly more *P. plebejus* in the proximal dense beds than in the other three bed types but at Time 2, both the proximal bed types (i.e. dense and sparse) supported significantly more *P. plebejus* than the distal beds (Figure 4.3B).

Although no significant differences were found in the catches of *Metapenaeus ensis* for any of the ANOVA terms involving the main factors of Proximity or Density, the 3-way interaction (Period x Proximity x Density) was close to significant ($P = 0.06$, Table 4.3). Catches of *M. ensis* appeared to be greater in the proximal dense beds than the other categories at Time 1, but not at Time 2 (Figure

4.4A). Catches of *M. ensis* varied considerably between the sites in each category of bed on each occasion (Period x Sites interaction, P < 0.001).

Catches of *Penaeus esculentus* in the four categories of seagrass bed generally showed the opposite pattern to the other three species, with more *P. esculentus* present in the distal than the proximal beds (ANOVA, 3-way interaction, P < 0.02, Table 4.2, Figure 4.4B). At Time 1, these differences were not significant, but at Time 2, there was significantly more *P. esculentus* in the distal sparse beds than the distal dense beds, both of which supported more of this species than the proximal beds (Figure 4.4B).

4.4.3. Habitat characteristics

The physical characteristics of the different seagrass beds in the western bay separated the dense from the sparse beds on the basis of the specific features of the seagrasses and the sediments, specifically the gravel content and degree of compaction (Figure 4.5, Table 4.3). There was considerable overlap in the characteristics of the seagrass beds in both the dense (i.e. proximal dense vs. distal dense) and sparse categories, indicating that there was less variation due to proximity than seagrass density *per se*.

4.4.4. Regional estimates of prawn abundance

In western Moreton Bay, proximal dense seagrass was the most abundant of the four types of habitat examined (Table 4.4), with over 736 hectares present, compared with < 250 ha for each of the other categories of habitat. The estimated potential contribution of proximal dense seagrass to postlarval and juvenile populations of *M. bennettae* and *P. plebejus* from western Moreton Bay was at least 8-12 times greater than the other habitats (Table 4.4). Likewise, the proximal sparse habitat could potentially contribute several times greater numbers to the populations of these species in Moreton Bay than either of the distal seagrass habitats, based on the scaled estimates of abundance for each habitat (Table 4.4).

In eastern Moreton Bay, despite there being considerably less proximal dense seagrass than in western Moreton Bay (166 ha compared with 737 ha), the estimated potential contribution to stocks of *Penaeus plebejus* was similar in both regions because the average density of *P. plebejus* was about six times higher in the east than the west in this type of seagrass (Table 4.4). For the proximal sparse seagrass, the relatively large available area of this habitat in the eastern region, combined with the greater densities of *P. plebejus*, meant that the estimated potential contribution of this habitat to stocks of *P. plebejus* was considerably greater in the eastern region than the western.

4.5. Discussion

This study has clearly shown that the spatial arrangement of intertidal seagrass meadows and mangrove forests influenced the abundance of populations of juvenile prawns using the seagrass. The relative proximity between seagrass and nearby mangroves interacted with the structural complexity (density) of the seagrass beds so that significantly more *Metapenaeus bennettae* and *Penaeus plebejus* were found in patches of dense seagrass close to mangroves than in other patch types. Importantly, patches of sparse seagrass close to mangroves supported greater numbers of these species than dense patches of seagrass further away from the mangroves, in contrast to previous studies that highlighted the importance of the structural complexity of seagrass alone (Young 1978; Loneragan et al. 1998**).** Hovel and Lipcius (2001) previously showed that the spatial configuration of seagrass beds was important to the dynamics of juvenile blue crabs (*Callinectes sapidus*), with seagrass density and patch size interacting to determine rates of survival of the crabs (see also Hovel and Lipcius, 2002). In a detailed study over large-scales, Nagelkerken et al. (2001) found that the species richness and abundance of fish assemblages was significantly greater in seagrass beds with adjacent mangroves than in areas without the mangroves. Similarly, Mumby et al. (2004) showed that the presence of mangroves had significant positive effects on the biomass and community composition of fish using nearby coral reefs. Interestingly, Eggleston et al. (1999) found that the abundance and species richness of macrofauna in patches of seagrass adjacent to patches of oyster rubble was not greater than in isolated patches of seagrass but noted that the species list for each of these habitat types was very similar so the lack of effect of proximity between those habitats is perhaps not surprising. The community composition of the nekton in the intertidal seagrass and mangroves in our study region are very different from each other (unpublished data) suggesting that the two habitats provide a different range of resources to the nekton and possibly accounting for the marked differences we observed between proximal and distal seagrass beds. Clearly, more work is needed on the role of spatial arrangement of habitats for different combinations of habitats in a range of different systems.

There are a number of ways that the proximity between seagrass and mangroves could enhance the value of the seagrass to prawns. Mangroves forests produce large amounts of organic matter via litterfall and much of this detritus is exported into adjacent habitats (Gong et al. 1984; Robertson et al. 1991). Seagrass beds that are closer to adjacent mangroves are likely to receive greater amounts of exported detritus than those beds further away. This enhanced detrital supply could then support more extensive nearshore bacterial and detritivore communities (Alongi et al. 1990), in turn providing more food for juvenile prawns (e.g. Ruello 1973; Wassenberg and Hill 1987). Exported mangrove material may also provide additional physical structure in adjacent habitats (Daniel and Robertson 1990; Robertson and Blaber 1992), so the larger number of prawns in seagrass beds proximal to mangroves than those further away may result from the combined effect of additional food and shelter provided by the exported mangrove detritus (see also Yañez-Arancibia et al. 1993).

It is also possible that the M. *bennettae* and *P. plebejus* in the seagrass are able to take advantage of the additional shelter and food provided by mangroves (Laegdsgaard and Johnson 1995; Manson et al. 2005b), migrating upshore during high tide, as has been suggested for some species of prawns (Robertson 1988; Vance et al. 1996; Bishop and Khan 1999), lobsters (Acosta 1999) and fish (Rozas and Odum 1987; Laegdsgaard and Johnson 1995; Laegdsgaard 1996; Raposa and Oviatt 2000; Sheaves and Molony 2000). The patches of seagrass close to the mangroves would be more accessible than those further away, allowing the prawns to reach this refuge more readily and return to the seagrass as the tide recedes. However, this is unlikely for several reasons. First, although *Metapenaeus bennettae* is abundant in *Avicennia* forests in the western region during spring and summer in Moreton Bay, they were only found in the intervening mudflats (between the seagrass and mangroves) during spring and not in summer. *Penaeus plebejus*, on the other hand, was abundant in the mudflats in spring and summer, but only occurred in very small numbers in the mangroves during summer (Skilleter and Loneragan, unpublished data). If the animals were regularly moving between the seagrass and mangroves, the expectation would be that they would also have occurred more regularly and in larger numbers in both the mangroves and intervening mudflats during both spring and summer when they were abundant in the seagrasses.

Second, juveniles of many species of prawns are less mobile than adults, staying close to the substratum or partially buried within it in, possibly in order to avoid predators (Dall 1958; Joshi et al. 1979; Dall et al. 1990). It is likely that they stay within the habitats in which they settle until moving into deeper water as sub-adults (Lucas, 1974; Coles and Lee Long 1985; Staples et al. 1985; Dall et al. 1990), not risking the exposure that would come from the transition from the seagrass to the mangroves (see Acosta 1999 for an example with spiny lobsters). Studies on the behaviour of juvenile *P. esculentus* (Hill and Wassenberg 1993) showed that they had a strong preference for staying within or close to seagrass during night and day.

Rates of predation on prawns may also vary between proximal and distal seagrass beds, accounting for the observed differences in abundance. The spatial configuration (fragmentation) of seagrass beds has been shown to be an important determinant of the abundance of juvenile blue crabs (Eggleston et al. 1998; Hovel and Lipcius 2001) and grass shrimp (Eggleston et al. 1998) by modifying the distribution and/or behaviour of their predators. Micheli and Peterson (1999) found that the proximity of saltmarsh and oyster reefs affected the survival of benthic clams on the reefs; survival of benthic clams was lower on reefs closer to saltmarsh because of the greater abundance of the predatory blue crabs (*Callinectes sapidus*) that are found in saltmarsh habitats. We currently have no specific detailed information on the predators of prawns in our system, although toadfish (*Tetractenos hamiltoni*) are abundant in intertidal seagrass beds during high tide throughout the year and consume prawns (Skilleter, unpublished data). This potential explanation for our results warrants further investigation.

In contrast to the patterns observed for *M. bennettae* and *P. plebejus*, *P. esculentus* was more abundant in seagrass that was further away from the mangroves, with the largest number in the sparse distal beds. It is also not clear why more *P. esculentus* occurred in the sparse than dense seagrass beds when these beds were distal from the mangroves. Previous studies had found that *P. esculentus* were more abundant in seagrass with the greatest structural complexity and biomass (Loneragan et al. 1994, 1998; Kenyon et al. 1997), albeit in northern tropical waters, although Haywood et al. (1995) found that the abundance of juvenile *P. esculentus* was greatest at a site with the smallest biomass of seagrass. Juvenile *P. esculentus* also suffer greater rates of mortality from predation as the complexity and biomass of seagrass declines (Kenyon et al. 1995). It seems unlikely that the differences in abundance between sparse and dense seagrass in the present study could be explained on the basis of the differences in sediments: patches of sparse seagrass tended to have a greater gravel content and levels of compaction than the dense seagrass. Adult *P. esculentus* have a preference for sandy (coarser) sediments (Somers 1987), but these preferences do not appear to come into play into animals are larger than ~10 mm CL, a size when they are less dependent on the physical structure of seagrass. Further work is needed to explain this particular pattern.

The seagrass beds closer to the mangroves occurred higher on the shore that those further away, so the differences in height on the shore and length of tidal inundation may also account for the observed differences in abundance of the prawns. However, this is unlikely because the intertidal gradient on shores in Moreton Bay is relatively small and there was only a 10-15 minute interval between the time the sites low on the shore were inundated, compared with those closer to the mangroves. All sampling was done during spring tidal periods, when the rate at which the intertidal is flooded is greatest, further reducing the difference in times that the different heights were inundated. Detailed sampling of seagrass beds that extended from the top of the shore to low on the shore in the absence of adjacent mangroves would provide a test of this, but such situations are lacking in western and eastern Moreton Bay.

The abundance of the different species of prawns varied considerably between the western and eastern regions of Moreton Bay, with *Metapenaeus bennettae* more abundant in the west and *Penaeus plebejus* more abundant in the east. These overall differences are at least partly due to variations in the sediments (Williams 1958; Branford 1981; Rulifson 1981; Somers 1987) and salinity (Dall 1958, 1981; Gunter et al. 1964; Mair 1980) and support previous studies showing the same patterns (Young 1975, 1978).

An important component of this study was the experimental design, aimed at maximising the spatial interspersion of sites for each of the different treatments to avoid confounding of the results (see also Nagelkerken et al. 2001). Previously, Raposa and Oviatt (2000) had contended that spatial confounding of their sites was unlikely to explain patterns of distribution and abundance of fish in relation to the proximity of seagrass to saltmarsh. Our results lend support to their arguments, that the proximity of the seagrass to the saltmarsh habitats was the driving force behind the patterns they observed. Variation in rates of advection of larvae to the different patch types (e.g. Stockhausen and Lipcius 2003) is unlikely to explain our results, given the interspersion of the sites belonging to the different treatments. Despite the considerable and often significant variation we observed in the abundance of prawns among sites within the different categories of seagrass, the differences between treatments (proximity and density) were still significant, indicating that variation in these factors was an important contributor towards explaining overall variation in the abundance of prawns across scales of 100-1000's of metres.

The interspersion of the sites that were sampled also allowed us to scale up our estimates of abundance for the different species of prawns at local scales (10-100's m) to provide estimates of the potential contribution of different spatial arrangements (mosaics) of seagrass and mangroves to offshore prawn fisheries. Moreton Bay contributes a significant proportion to the total commercial catch on the east coast of Australia for the eastern king prawn (*Penaeus plebejus*) (Lucas 1974; Trainor 1990, 1991) and our estimates indicate that seagrass located in close proximity to mangroves on both sides of the bay potentially provide the greatest source of juveniles to the regional fishery.

Our estimates should only be viewed as a potential contribution of juveniles to subsequent adult stocks though, because in the absence of detailed information on the relative growth and survival of juveniles in different mosaics and subsequent survival during the migration to the adult habitats, it is not yet possible to determine how many of these juveniles would survive to recruit to the fishery (Beck et al 2001; Halpern, 2004). In addition to this point our estimates are limited in time and the recruitment of prawns to inshore nurseries takes place over extended periods (see e.g. Young and Carpenter 1977; Loneragan et al. 1994; Vance et al. 1998). An experimental examination of the carrying capacity of different mosaics using enclosures stocked with different densities of prawns (e.g. Loneragan et al. 2001) and tagging studies focussed on assessing survival of juveniles and small adults during migration (e.g. Somers and Kirkwood 1984; Montgomery 1990) would enable this information to be incorporated into a model predicting the relative importance of different mosaics in supporting commercial catches of different species. Nevertheless, the potential importance of the variation in spatial arrangement of habitats and, in particular, the relative proximity between different types of habitats on the size of adult populations of nekton should not be underestimated (Nagelkerken et al. 2001).

Table 4.1: Results of analyses of variance comparing the abundance (per $100m^2$) of *Metapenaeus bennettae* and *Penaeus plebejus* sampled in seagrass beds of different complexity (dense versus sparse) and proximity to mangroves (proximal versus distal). Data were transformed to $\log_e(x+1)$. Western Moreton Bay: n=4 from each of two sites per category of seagrass. Eastern Moreton Bay: n=4 from each of three sites per category of seagrass.

Species and Source of Variation	df	Mean Square	F, *P*
(a) *Metapenaeus bennettae* – western bay			
Proximity: P	1	29.04	105.69, 0.001
Density: D	1	14.45	52.59, 0.002
P x D	1	5.29	19.24, 0.012
Sites (P x D)	4	0.27	0.38, 0.819
(b) *Penaeus plebejus* – western Bay			
Proximity: P	1	18.24	7.99, 0.047
Density: D	1	8.16	3.58, 0.132
P x D	1	0.27	0.12, 0.748
Sites (P x D)	4	2.28	5.81, 0.002
(c) *Penaeus plebejus* – eastern Bay			
Proximity: P	1	17.24	6.69, 0.032
Density: D	1	19.30	7.49, 0.026
P x D	1	0.81	0.32, 0.589
Sites (P x D)	8	2.58	2.57, 0.001

Table 4.2: Results of analyses of variance comparing the abundance (per $100m^2$) of *Metapenaeus bennettae, M. ensis, Penaeus plebejus* and *P. esculentus* sampled in seagrass beds of different complexity (dense versus sparse) and proximity to mangroves (proximal versus distal) in western Moreton Bay on two separate occasions. Data were transformed to $\log_e(x+1)$. n=6 from each of two sites per category of seagrass.

Source of Variation	df	Mean Square (MS), F and *P*			
		MS	F and P	MS	F and P
			M. bennettae		*M. ensis*
Period: T	1	0.10	0.02, 0.923	0.32	0.02, 0.908
Proximity: P	1	85.82	36.09, 0.004	2.10	0.50, 0.520
Density: D	1	25.80	10.85, 0.030	12.01	2.85, 0.167
T x P	1	0.06	0.46, 0.502	2.23	3.30, 0.073
T x D	1	0.01	0.04, 0.847	0.70	1.04, 0.312
P x D	1	0.88	0.37, 0.576	2.14	0.51, 0.516
T x P x D	1	0.53	4.37, 0.040	2.46	3.65, 0.060
T x Sites (P x D)	1	6.95	57.10, 0.001	14.97	22.18, 0.001
Sites (P x D)	4	2.38	19.53, 0.001	4.22	6.25, 0.001
			P. plebejus		*P. esculentus*
Period: T	1	0.71	0.11, 0.794	8.65	2.17, 0.380
Proximity: P	1	24.69	56.27, 0.002	33.92	4.62, 0.098
Density: D	1	7.00	15.94, 0.016	0.36	0.05, 0.836
T x P	1	0.00	0.01, 0.979	0.54	0.99, 0.323
T x D	1	0.81	1.52, 0.222	0.06	0.12, 0.733
P x D	1	0.04	0.08, 0.787	0.09	0.01, 0.918
T x P x D	1	5.23	9.77, 0.002	1.67	3.07, 0.083
T x Sites (P x D)	1	6.30	11.77, 0.001	3.98	7.34, 0.008
Sites (P x D)	4	0.44	0.82, 0.516	7.34	13.52, 0.001

Table 4.3: Summary of the key physical characteristics for each of the four seagrass bed types (proximal dense, distal dense, proximal sparse and distal sparse) in western Moreton Bay. Values are means (±SE) based on n=16 samples for each variable. AG biomass = above ground biomass (grams dry weight per m^2), BG biomass = below ground biomass (grams dry weight per m^2). Compaction = cm penetration into sediment by standard probe (larger values mean less compacted sediment).

Species and characteristic	Type of seagrass bed			
	Proximal Dense	Distal Dense	Proximal Sparse	Distal Sparse
Zostera				
Length (mm)	17.16 ± 0.41	13.28 ± 0.93	10.91 ± 0.74	7.29 ± 0.92
Blades.shoot^{-1}	9.09 ± 0.20	9.00 ± 0.28	8.25 ± 0.39	6.39 ± 0.71
AG Biomass (g)	157.28 ± 11.93	172.30 ± 8.39	47.27 ± 5.74	27.83 ± 4.42
BG Biomass (g)	142.70 ± 21.21	153.74 ± 5.74	38.44 ± 4.42	17.67 ± 3.09
Halophila				
AG Biomass (g)	12.37 ± 11.31	7.07 ± 1.41	1.77 ± 0.71	15.90 ± 4.24
BG Biomass (g)	1.77 ± 0.71	8.84 ± 1.24	1.77 ± 0.53	15.90 ± 4.42
Compaction (cm)	21.35 ± 1.12	16.28 ± 0.81	11.99 ± 1.05	8.80 ± 0.33
% gravel content	3.56 ± 0.85	9.17 ± 2.92	23.01 ± 5.53	21.43 ± 1.95

Table 4.4: Estimates of the abundance of postlarval and juvenile *Metapenaeus bennettae* (western Moreton Bay) and *Penaeus plebejus* (western and eastern Moreton Bay) in each of the four different types of seagrass bed (proximal and distal dense and proximal and distal sparse), scaled to the absolute area of these patch types. Scaled abundance is shown as the range based on 95% confidence limits around the mean.

Region, species and variable	Type of seagrass bed			
	Proximal - Dense	Proximal - Sparse	Distal - Dense	Distal - Sparse
Western Moreton Bay				
Area (ha)	736.6	152.3	240.3	119.7
M. bennettae				
Density (100m^2)	34.08	12.92	2.00	0.50
Scaled Mean	2,510,576	196,721	48,060	5,985
Range: ± 95%CL	1,601,213 - 3,419,944	12,866 – 380,576	3,785 – 92,335	0 – 16,982
P. plebejus				
Density (100m^2)	41.08	23.50	9.63	3.75
Scaled Mean	3,026,196	357,905	231,289	44,888
Range: ± 95%CL	1,124,862 – 4,927,535	210,832 – 504,978	95,485 – 367,082	6,637 – 83,138
Eastern Moreton Bay				
Area (ha)	165.8	221.0	30.3	21.3
P. plebejus				
Density (100m^2)	184.75	39.08	39.83	8.50
Scaled Mean	3,063,155	863,741	120,695	18,105
Range: ± 95%CL	1,524,472 – 4,601,839	642,009 – 1,085,475	109,062 – 132,328	12,895 – 23,315

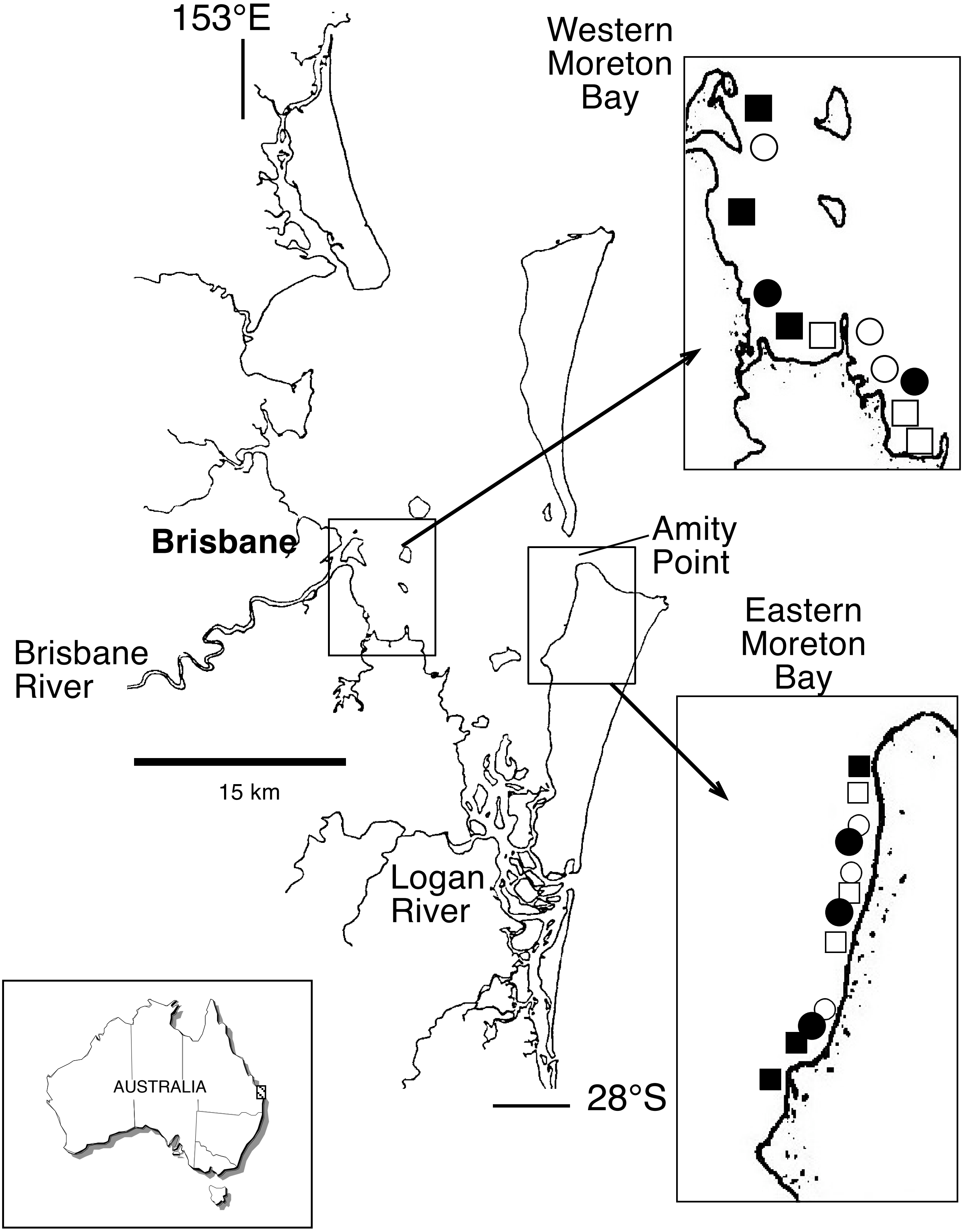


Figure 4.1: Map of Australia and the Moreton Bay region in South-east Queensland, showing the position of the study sites in western and eastern Moreton Bay. Four different categories of seagrass bed were sampled: proximal dense beds (filled squares), proximal sparse beds (clear squares), distal dense beds (filled circles) and distal sparse beds (clear circles). The position of beds of each category were interspersed as much as possible along the shoreline (see text for further details).

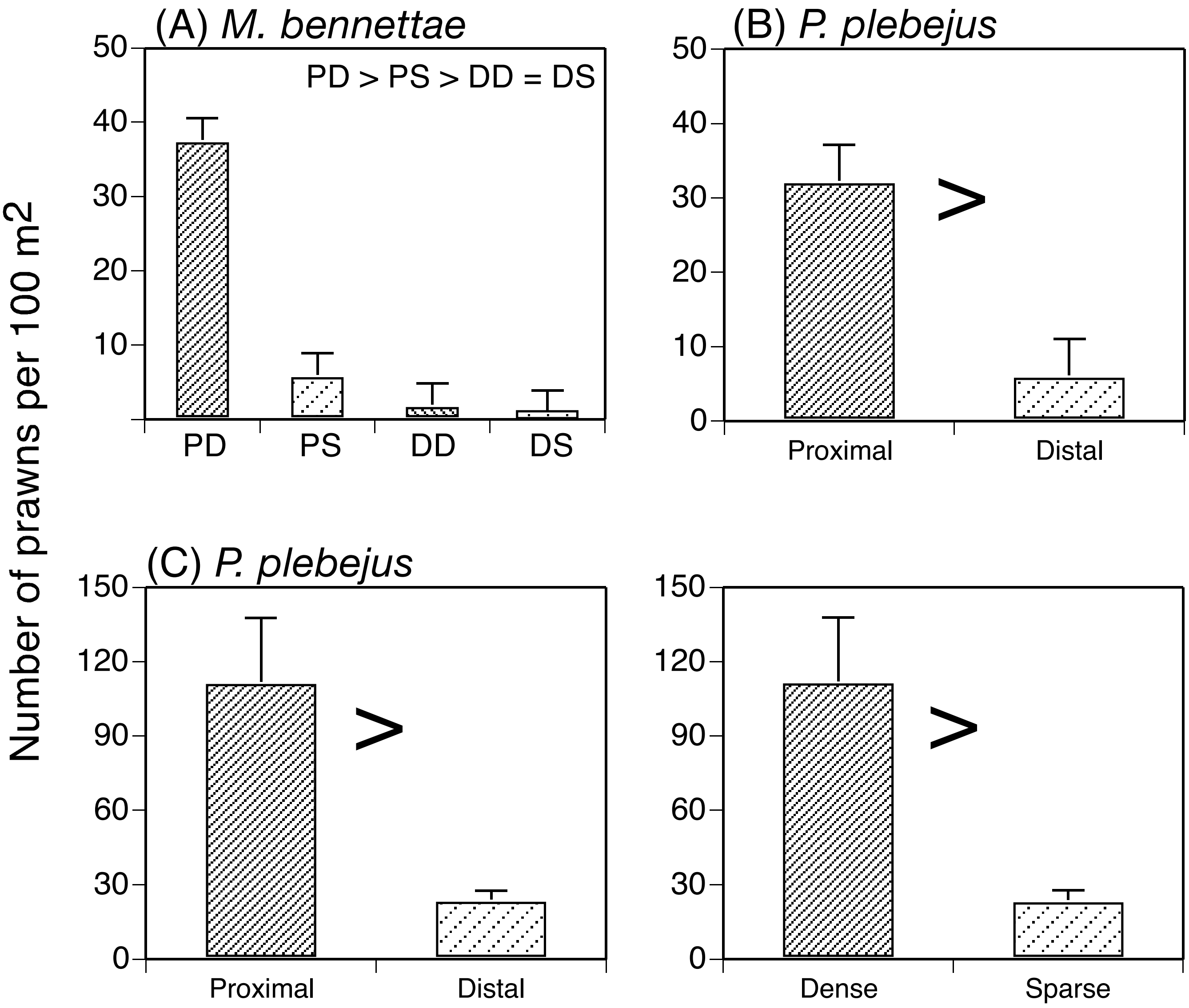


Figure 4.2: Mean (+SE) abundance of (A) *Metapenaeus bennettae* in western Moreton Bay, (B) *Penaeus plebejus* in western and (C) *Penaeus plebejus* in eastern Moreton Bay per 100 m^2 of intertidal seagrass in four categories: proximal dense (PD), proximal sparse (PS), distal dense (DD) and distal sparse (DS) in western Moreton Bay. Results of SNK tests are shown for differences among interactions (A & B) or main effects (C).

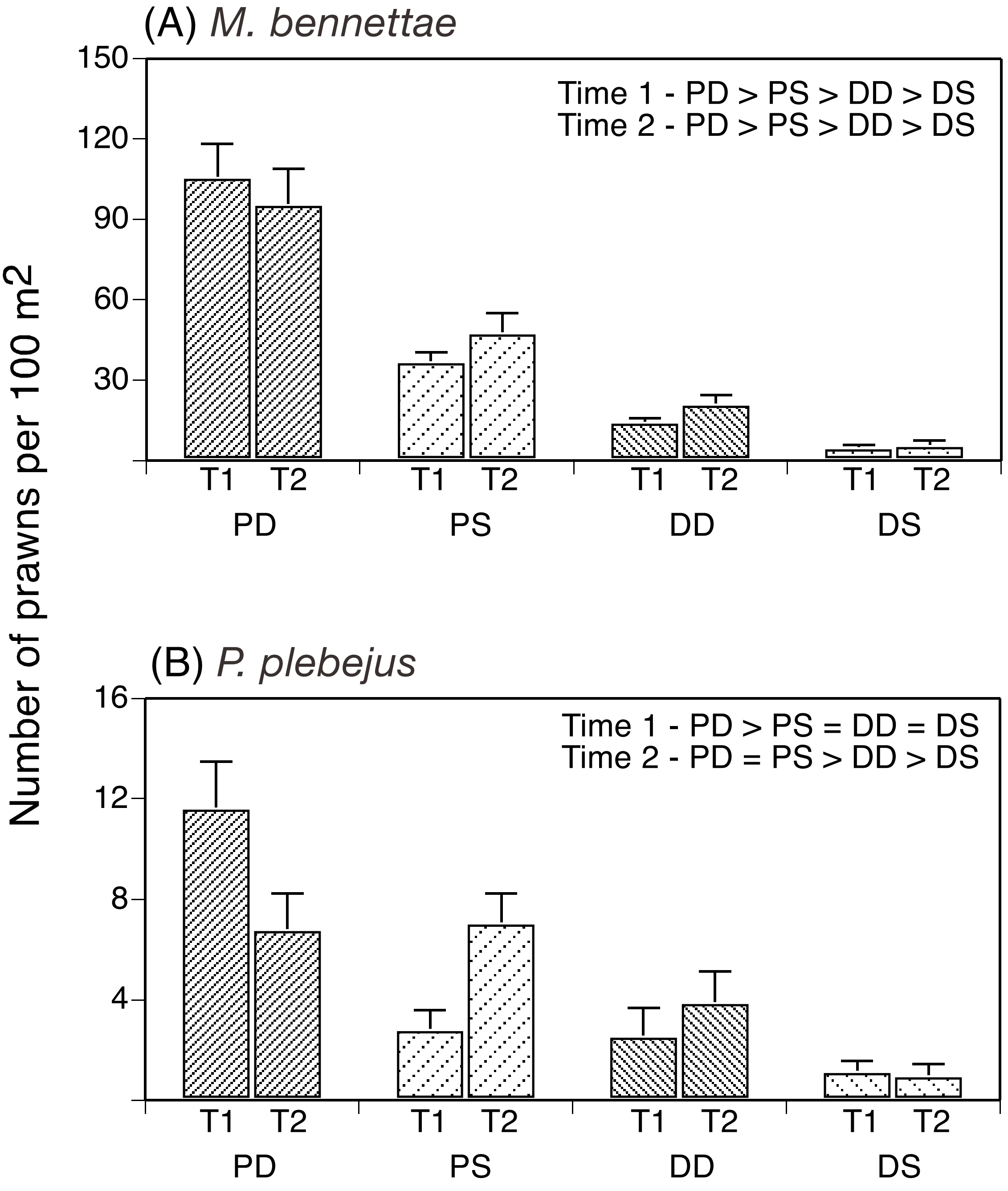


Figure 4.3: Mean (+SE) abundance of (A) *Metapenaeus bennettae* and (B) *Penaeus plebejus* in western Moreton Bay per 100 m^2 of intertidal seagrass in four categories: proximal dense (PD), proximal sparse (PS) distal dense (DD) and distal sparse (DS) sampled on two separate occasions. Results of SNK tests are shown for differences among treatment in 3-way interaction (n=12 pooled across 6 replicate seines in each of 2 sites per time).

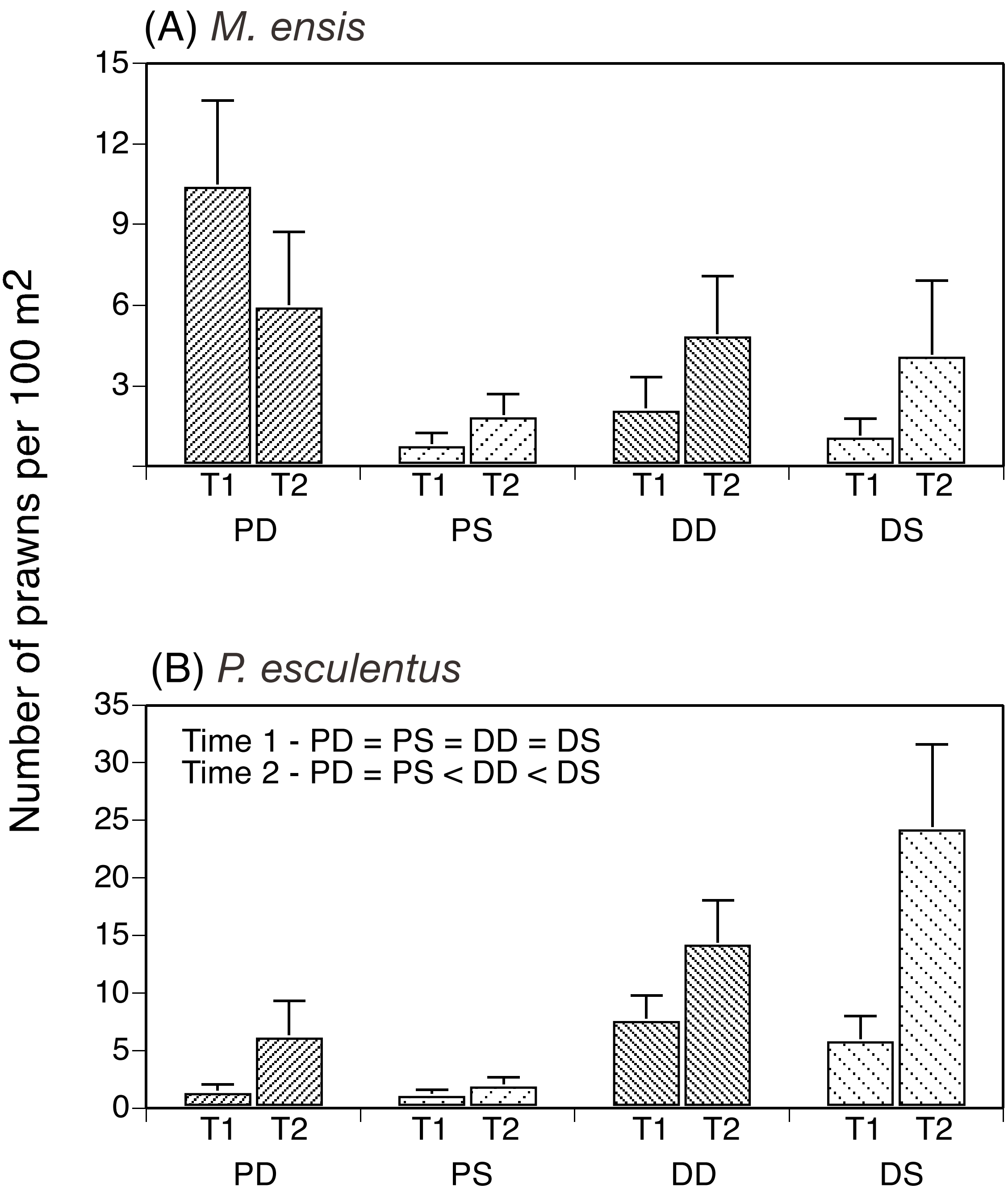


Figure 4.4: Mean (+SE) abundance of (A) *Metapenaeus ensis* and (B) *Penaeus esculentus* in western Moreton Bay per 100 m^2 of intertidal seagrass in four categories: proximal dense (PD), proximal sparse (PS) distal dense (DD) and distal sparse (DS) sampled on two separate occasions. (n=12, pooled across 6 replicate seines in each of 2 sites per time).

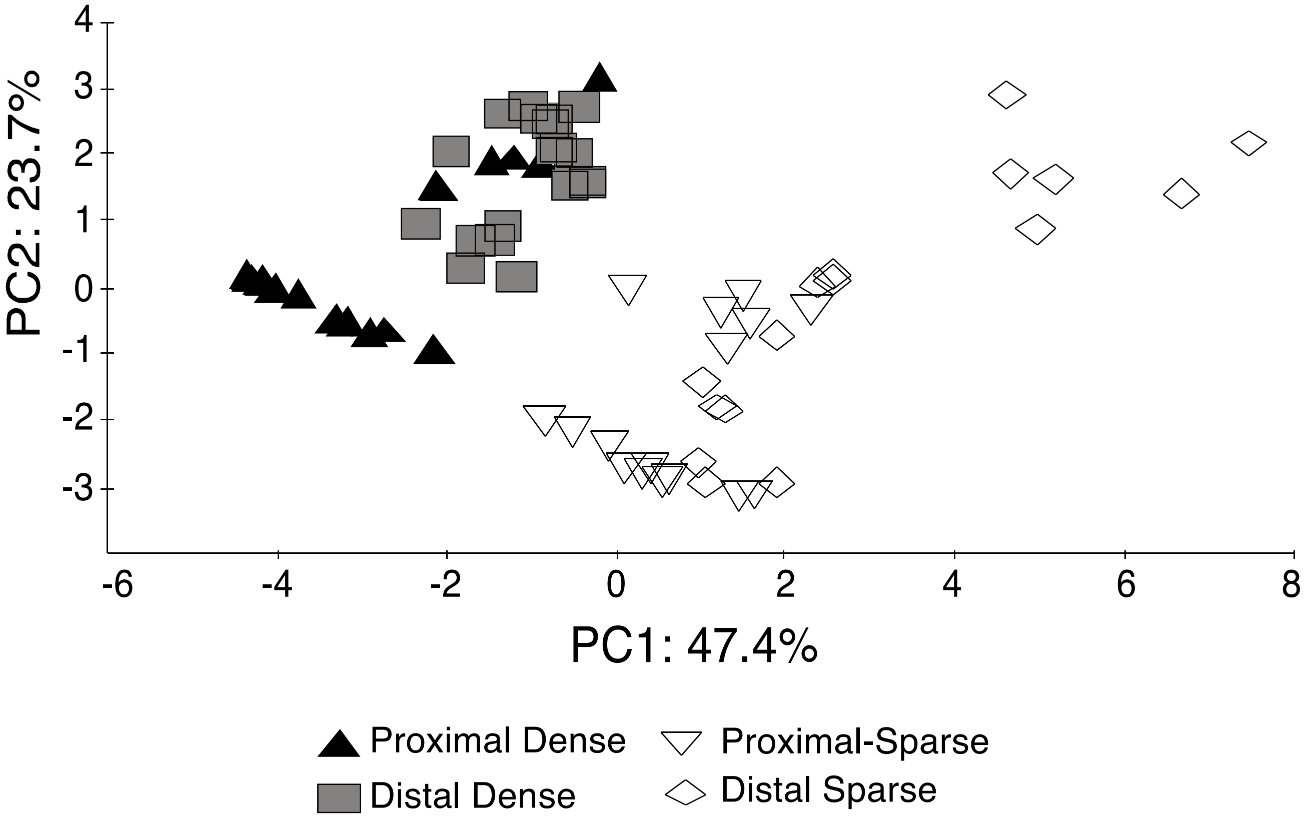


Figure 4.5: Principal components analysis ordination on normalised data using the Euclidean distance measure for physical characteristics (see Table 3) of each of the categories of seagrass bed in western Moreton Bay. The two PC axes shown account for 71% of the variation in the data. PC1 reflects variation in characteristics of *Zostera capricorni*, while PC2 reflects variation in sediment characteristics (% gravel and compaction) and characteristics of *Halophila ovalis*.

Section 5: Effects of Proximity Between Seagrass and Mangroves on Nekton Assemblages

5.1. Introduction

Loss of and changes to habitat quality and extent have had marked impacts on estuarine and coastal systems globally (Lotze et al., 2006) with implications for fisheries, biodiversity and ecosystem services (Jackson et al., 2001; Anger et al., 2005; Danielson et al., 2005). Fragmentation and disturbance have changed the structure and configuration of coastal landscapes (Sala et al., 2000; Valiela et al., 2001, 2002; Manson et al., 2003; Hale et al., 2004; Adger et al., 2005), potentially decreasing the extent to which they can support plants and animals (Saunders et al., 1991; Fahrig, 2003).

Fragmentation in landscapes leads to a decrease in patch size and/or an increase in the distance between different types of patch, both leading to a decrease in the connectivity within the landscape (Goodwin and Fahrig, 2002). A loss of connectivity among different types of habitat is likely to affect the capacity for animals to move through the landscape, taking advantage of different resources (Taylor et al., 1993; Goodwin and Fahrig, 2002). Maximising connectivity among different patches of habitat has now been recognized as an important criterion for incorporation into conservation planning for the maintenance of biodiversity (e.g. Pyke, 2005) because of the influence on the persistence of populations and assemblages. Proximity between different patches of habitat has been shown to influence the dispersal of animals between sources and sinks, with occupancy of sink patches greater when they are closer to larger areas providing a source of individuals (e.g. Nol et al., 2005).

Estuarine tidal banks and wetlands (mangroves and saltmarshes) support diverse and abundant assemblages of nekton, a function of abundant food and refuges from predation (reviewed by Kneib, 1997; Manson et al., 2005b). Access to these areas by nekton is, however, often restricted to periods of high tide, with the extent and mix of habitats that are available during these periods dependent on the physical characteristics of a site, such as tidal range and gradient, but also varying temporally through the lunar cycle and as a consequence of local climatic conditions (e.g. low pressure systems nearby increase the tidal excursion through the intertidal). During high tide though, the amount of shallow water habitat is greatly increased, especially for small fish (McIvor and Odum, 1988). During high tide, nekton move from subtidal refuge habitats into mangroves and seagrass areas (Rozas and Odum, 1987; Kneib and Wagner, 1994; Irlandi and Crawford, 1997; Vance et al., 1996, 2002), but in doing so may have to traverse patches of habitat, such as bare sediment or sparse seagrass (Skilleter et al., 2005; Zharikov et al. 2005), that provide less protection from predators (Coen et al., 1981; Heck and Thoman, 1981; Stoner, 1982; Summerson and Peterson, 1984; Skilleter, 1994). An increase in the distance between patches of habitat under these circumstances will decrease connectivity, either through increased rates of predation (Dunning et al., 1992; Johnson et al., 1992; Puth and Wilson, 2001) or greater avoidance of the riskier habitats within the landscape matrix (Taylor et al., 1993; Keyser et al., 1998; Olden et al., 2004). Using a combination of simulations and empirical field tests, Goodwin and Fahrig (2002) showed that inter-patch distance had the strongest and most consistent (negative) effect on landscape connectivity in terrestrial systems.

The value of specific types of estuarine intertidal nursery habitats in supporting nekton assemblages may be adversely affected by declining connectivity between different types of habitat. Direct measures of connectivity often require detailed measurements of either individual movements or indirect measures of immigration/emigration rates (Goodwin and Fahrig, 2002). There is, however, a paucity of detailed information on movements by nekton in estuarine systems (see Beck et al. 2001 and Gillanders et al., 2003). A first step in acquiring information on connectivity is to identify a range of species that respond differently to variation in landscape composition and structure. Such species are then good candidates for more detailed studies testing directly hypotheses about connectivity. This is an appropriate approach for an estuarine landscape involving use of intertidal habitat patches by nekton: these patches are only available for a short period of time during high tide so the presence of nekton within the patch indicates dispersal from a subtidal habitat.

Here, we examined the extent to which the proximity between seagrass beds and mangrove forests affected the utilisation of these seagrass beds by nekton (i.e. landscape complementation *sensu* Dunning et al., 1992). This is one important measure likely to influence connectivity among patches of habitat within a landscape (Dunning et al., 1992; Taylor et al, 1993). Clearly, connectivity between habitats such as mangroves and seagrass plays an important role determining the use of estuarine habitats for groups such as prawns (Skilleter et al., 2005). In Section 6, we examine another important component of connectivity, the structural characteristics of the matrix that separates the different patches of habitat being utilized by the nekton at different stages of the tide. The composition and arrangement of the different patches will affect connectivity among patches (Goodwin and Fahrig, 2002), so we simultaneously examined the effects of proximity at two levels of habitat complexity for the seagrass beds. Dense seagrass generally supports greater numbers of species and individuals than sparse patches (references above), so we expected an interaction between the effects of proximity and seagrass density on the utilisation of the seagrass by nekton. Variation in landscape context (Wiens, 2002 and references therein) has also been shown to influence the connectivity of landscapes, so we examined the effects of proximity (and density) in two very different regions of a subtropical estuarine embayment.

5.2. Methods

5.2.1. Study sites

Sampling was done in Moreton Bay, Queensland, a large subtropical embayment on the east coast of Australia (Lat. 27° S, Long. 153° E, Fig. 1) with extensive seagrass coverage (Hyland et al. 1989) dominated by *Zostera capricorni* Ascherson. Moreton Bay has a semi-diurnal tidal pattern with a range of 1.5 to 2.0m during a full monthly tidal cycle (Dennison and Abal 1999). Salinity ranges from 28-34‰, with little seasonal variation on the eastern side of the embayment (Gabric et al. 1998), but salinities on the western side are more variable because of the influence of several major river systems (Young 1978; O'Brien 1994).

5.2.2. Experimental Design and Sampling Methods

To determine whether the abundance and composition of the nekton assemblage varied with the proximity of seagrass beds to adjacent mangrove forests and/or as a function of the structural complexity of the seagrass, four different categories of seagrass bed were examined:

(1) dense seagrass beds proximal to mangroves,
(2) dense seagrass beds distal to mangroves,
(3) sparse seagrass beds proximal to mangroves and
(4) sparse seagrass beds distal to mangroves.

Initially, three sites in each category of seagrass bed were surveyed in both the eastern and western regions of Moreton Bay (a total of 24 sites). Sites from each region were included in the sampling design to determine whether any differences in utilisation of the different seagrass beds were consistent across the bay. The western region of Moreton Bay is affected by urban and industrial development, whereas the eastern region is relatively undeveloped (Dennison and Abal 1999) so the quality of the environment varies considerably on either side of the Bay. The distribution of the four categories of seagrass bed also varied considerably in different regions of Moreton Bay and the selection of sampling sites had an added criterion that beds belonging to the different categories should be spatially interspersed as much as possible (Figure 5.1) to reduce any spatial confounding of comparisons among the treatments (e.g. Raposa and Oviatt, 2000; Jelbart et al., 2007). The measures of proximity and density therefore varied between the two regions. Proximal patches were categorized as those from 0-20 m from adjacent mangrove forests in the western bay and 0-40 m in the eastern bay. Distal beds were separated from adjacent mangroves by 90-150 m of unvegetated sediment in the western bay and 150-350 m in the eastern bay. Sparse sites had 30-40% seagrass coverage in the western bay and 20-30% cover in the eastern bay. Dense sites had 70-90% seagrass coverage in the western bay and 60-70% cover in the eastern bay.

Sampling of the different sites was randomised through time, except that the sites in western Moreton Bay (Sept-Oct. 2002) were all sampled before those in eastern Moreton Bay (Oct.-Nov. 2002). Two sites could be sampled on any day, so a set of 12 sites (3 sites in each of the 4 categories of seagrass) required 6 days of sampling. Over the required period, sampling of the 12 sites was organised so that any effect of short-term (daily) variation in abundance was randomised across all treatments, avoiding confounding of any comparisons among these treatments.

Sampling was done using a small seine net, hauled through the water at high tide by two people, along a pre-designated path marked by buoys. The net, 6 m long (1 mm mesh with 2 m drop), was hauled for 25 m parallel to the adjacent mangrove fringe, with a constant mouth width of 4 m, sampling a total area of 100 m^2 for each haul. At each site, four replicate hauls were positioned on a transect along the shoreline, each haul separated by a distance of 10 m. Sampling was done within two hours either side of the daytime high tide during the six days spanning either the full or new moon and at a relatively constant water depth (0.7 to 1.0 m). Samples were frozen until they could be examined in the laboratory where individuals were identified to species and counted.

Analysis of the data from this first set of samples indicated marked effects of both proximity and density on the abundance of different groups of nekton, so two sites from each of the four categories of seagrass bed in the western region of Moreton Bay were sampled more intensively to determine whether these patterns were consistent through time. These eight sites were sampled in two blocks between the 30th January and 2nd February 2003 and then again between the 7th to 10th of February 2003. Samples were collected by seine nets as described above, except six replicate samples were collected at each site compared with four replicates previously. Two sites, selected at random, were sampled on each day in each sampling block. Samples were frozen until they could be examined in the laboratory, where individuals were identified to species and counted.

The physical characteristics of the vegetation and sediments within each category of seagrass bed have been previously described (Skilleter et al., 2005; Section 4, this report).

5.2.3. Statistical Analyses

The composition of the nekton assemblage in seagrass beds in western and eastern Moreton Bay was compared using Analysis of Similarities (ANOSIM) on untransformed data using the Bray-Curtis similarity measure (Clarke, 1993), indicating highly significant differences between the two regions (see Section 5.3 - Results). On this basis, subsequent analyses comparing the composition and abundance of nekton in the different categories of seagrass bed were done separately for western and eastern Moreton Bay.

In each of the regions, the four categories of seagrass bed were analysed with a one-way analysis, given the limitations in the analysis of complex experimental designs using ANOSIM (it is not possible to test formally for interactions; Clarke, 1993). Pairwise tests within ANOSIM, comparing all possible combinations of the four categories of seagrass bed, were done to determine the nature of any differences among the bed types in the composition of the nekton assemblage. Examination of the values for the R-statistic from these pair-wise tests and the ordination plots (see below) was used to infer the presence of any interactions between the effects of proximity and density of seagrass bed. Differences in composition of the nekton assemblage were also examined graphically using non-metric multidimensional scaling (ordination) using the Bray-Curtis similarity measure on untransformed data (Clarke, 1993). The contributions of different taxa to the variation in the composition of the nekton among the four categories of seagrass bed were examined using SIMPER.

The species selected by SIMPER analyses were analysed with three factor, univariate ANOVAs, with Proximity and Density treated as fixed factors and Sites as a random factor (nested within Proximity x Density). Due to bad weather, sampling at one site in the dense, distal seagrass category in western Moreton Bay was disrupted. Catches at this site were dominated by large amounts of seagrass and algal debris and it was not clear whether the efficiency of the seine net was similar to that at the other sites. Data from this site were excluded from analyses, leaving an unbalanced design for the western Bay. Data from the eastern Bay were analysed with the same ANOVA design, except analyses were completely balanced. Data were log transformed to meet the assumptions of homoscedasticity of variances after Cochran's test (Underwood 1981).

Data from the more intensive sampling in western Moreton Bay, done on two additional occasions, were analysed in the same manner except the statistical model used for the univariate analyses of variance included an additional factor of Period. Period was considered a fixed factor in these analyses for several reasons. First, sampling on both these occasions was specifically done during summer to allow valid comparisons with the previous sampling which was also done during the Austral spring-summer period. Second, the two periods were separated by only a short amount of time to increase the likelihood that the same populations of nekton would be sampled as they accessed the intertidal seagrass beds from subtidal refuge areas, without substantial recruitment from the plankton.

5.3. Results

5.3.1. Community Composition – Regional Comparisons

Different nekton assemblages utilised the intertidal seagrass beds in the eastern and western regions of Moreton Bay (ANOSIM, $P < 0.001$). The ordination indicated there was some overlap of samples collected from each region (Figure 5.2) but generally the two assemblages were distinct. Seven species of invertebrates and five species of fish (Table 5.1) contributed 80 % to the separation of the two assemblages, with one species, the eastern king prawn, *Penaeus plebejus*, contributing > 18% to the distinction between the two regions. The large differences in the overall composition of the assemblages in eastern and western Moreton Bay and the marked differences in the abundance of individual species suggested that it would be appropriate to consider any effects of proximity (between seagrass and mangroves) and density (of seagrass) separately for the two regions.

5.3.2. Community Composition – Effects of Proximity and Seagrass Density

The four different categories of seagrass bed (Proximal-Dense, Proximal-Sparse, Distal-Dense and Distal-Sparse) in eastern Moreton Bay supported quite distinct nekton assemblages (ANOSIM, $P < 0.001$; pairwise comparisons: $P < 0.001$ to 0.003). There was some overlap in the samples from the distal and proximal beds in dense seagrass, but none between the proximal and distal beds in sparse seagrass, suggesting an interaction between the effects of proximity and density (Figure 5.3). This was supported by the differences in the value of the R-statistic for the pairwise comparisons, with a larger value for the comparison in sparse seagrass than in dense seagrass (Proximal vs. Distal: $R_{sparse}= 0.52$; $R_{dense}= 0.39$).

In the dense seagrass beds, 8 species contributed ~80 % to the separation of the nekton assemblages in the proximal and distal beds (Table 5.2: Proximity x Density), with eastern king prawns, *Penaeus plebejus*, contributing over 40 %, with much greater abundances in beds close to mangroves than in those further away. The other species (5 species of fish and 2 species of invertebrates) contributed much less to the distinction between the proximal and distal beds. The fish were generally more abundant in the proximal beds, except for *Pelates sexlineatus*, the eastern striped trumpeter, where abundance were slightly greater in the distal beds. Both of these species of invertebrates, the carid shrimp *Palaemon debilis* and the southern pygmy squid, *Idiosepius notoides*, were more abundant in distal seagrass beds.

There was some overlap in the species that contributed to the separation of the proximal and distal beds in sparse seagrass compared with the dense beds (Table 5.2) with *Penaeus plebejus* again making the largest contribution to the separation of the assemblages (> 24 %). Five species each contributed more than 10 % to the separation of the proximal and distal beds in sparse seagrass, compared with only 2 species in the dense seagrass.

In western Moreton Bay, there was again complete separation of the assemblages in the four categories of seagrass bed (ANOSIM, $P < 0.001$; pairwise comparisons: $P < 0.001$ to 0.003), but in this region there was much greater distinction in the composition of the nekton assemblage from proximal and distal beds in the dense or the sparse seagrass beds (Figure 5.4). There was no indication of an interaction between the effects of proximity and density, the values of the R-statistic for the pairwise comparisons were similar in dense and sparse grass (Proximal vs. Distal: $R_{sparse}= 0.48$; $R_{dense}= 0.49$).

A larger number of species contributed to the separation of the proximal and distal beds in both dense and sparse seagrass (11 species – dense, 9 species – sparse) than was evident for eastern Moreton Bay, although there was considerable overlap in which species were responsible for these differences. In the dense seagrass, the half-bridled goby (*Arenigobius frenatus*) contributed the greatest amount to the separation of the assemblages in proximal and distal beds, with 100 times more animals occurring in the dense beds close to the mangroves compared with those further away. In the dense seagrass beds, 8 of the 11 species contributing to the differences in the nekton assemblage between proximal and distal beds were more abundant in the former category (Table 5.3). In the sparse seagrass beds 6 of 9 species were more abundant in the proximal beds than those further away (Table 5.3).

5.3.3. Population Sizes - Effects of Proximity and Seagrass Density

5.3.3.1. Eastern Moreton Bay

The number of fish using sparse seagrass did not differ between the proximal and distal beds, but there were almost twice as many fish caught in the dense seagrass close to the mangroves than in seagrass further away (ANOVA; significant interaction – Proximity x Density – Table 5.4; Figure 5.5A). The number of species of fish, however, was only marginally greater in the dense than sparse beds and there was no significant interaction between proximity and density, nor any effect of proximity as a main effect (Table 5.4; Figure 5.5). Nine species were sufficiently abundant to analyse individually. Three species were significantly more abundant in dense seagrass beds close to mangroves that those further away (Figure 5.5E - *Arenogobius leftwichi*, Figure 5.5F - *Favonigobius lentiginosus*, Figure 5.5G - *Tetractenos hamiltoni*), but there was no significant different between proximal and distal sparse beds (significant interaction, Table 5.4; Figure 5.5). *Centropogon australis* showed a similar pattern,, with a 3-fold increase in abundance in the dense proximal beds over those further away, but this was not significant (Table 5.4; Figure 5.5D). In contrast, *Palaemon serenus* was more abundant in sparse seagrass close to mangroves than in the sparse beds further away, but there was no difference in the numbers using the dense beds (Figure 5.5I). *Palaemon debilis* was also more abundant in the sparse beds close to the mangroves than those further away, but in the dense beds the pattern was reversed with more shrimp in the beds further away from the mangroves (Figure 5.5H). The six-lined trumpeter, *Pelates sexlineatus*, showed a similar pattern (Figure 5.5C), but these differences were not significant (Table 5.4). *Hippolyte* sp. was more abundant in sparse seagrass away from the mangroves (Figure 5.5J) but this difference was not significant. Finally, the squid *Idiosepius notoides* was significantly more abundant in distal than proximal grass beds and in dense compared with sparse beds (Figure 5.5K), but there was no significant interaction between these main effects.

5.3.3.2. Western Moreton Bay

In western Moreton Bay, over 4 times as many fish individuals were caught in the dense beds close to the mangroves than in any of the other patch types (Figure 5.6A). Slightly more species of fish were caught in the proximal than distal beds, but there was no difference in species richness between densities of seagrass and there was no interaction between proximity and density (Figure 5.6B; Table 5.4). Five species were more abundant in proximal than distal beds: *Arenogobius frenatus* – Figure 5.5E (dense and sparse beds), *Centropogon australis* – Figure 5.6D, *Tetractenos hamiltoni* – Figure 5.6G, *Acanthopagrus australis* – Figure 5.6H (dense beds only) and *Urocampus carinirostris* – Figure 5.6I (proximal beds irrespective of density) (Table 5.4). *Pelates sexlineatus* was also more abundant in dense proximal beds than in patches further away, but this difference was not significant (Figure 5.6C; Table 5.4). In dense seagrass, *Latreutes pygmaeus* was more abundant in beds close to the mangroves than those further away, but the opposite was true for sparse seagrass, with more shrimp in distal than proximal beds, although neither of these differences was significant despite the large effects sizes (Figure 5.6J; Table 5.4). *Favonigobius exquisitus* (exquisite goby) was more abundant in dense than sparse seagrass but there was no effect of proximity on its abundance (Figure 5.6F; Table 5.4). Finally, *Latreutes porcinus* was more abundant in dense than sparse seagrass, but there was no effect of proximity as a main effect or an interaction with seagrass density (Figure 5.6K; Table 5.4).

5.3.3.3. Temporal Sampling - Western Moreton Bay

When sampling in western Moreton Bay was repeated on another two occasions, many of the patterns become clearer, with more pronounced differences in the abundance of different species between proximal and distal seagrass beds. More species of fish occurred in the proximal than distal beds, but only in dense seagrass (Figure 5.7A; Table 5.5). In contrast, total species richness (fish and invertebrates) was greater in dense beds than sparse but there were no significant effects of proximity, either as a main effect or as an interaction with any other term (Table 5.5). The number of individuals of fish was again significantly greater in the dense beds close to mangroves than those further away (Figure 5.7B), with no effect of proximity on the number of fish in the sparse beds. This same pattern was seen for *Arenigobius frenatus* (Figure 5.7C), *Favonigobius exquisitus* (Figure 5.7E), *Mugilogobius stigmaticus* (Figure 5.7F), *Centropogon australis* (Figure 5.7G) and *Tetractenos hamiltoni* (Figure 5.7M). In contrast, there were marginally more silver biddy, *Gerres subfasciatus (*Figure 5.7H), in proximal than distal beds for both dense and sparse seagrass, but there differences were not significant, presumably because of significant small-scale spatial variation in the abundance of this species (Table 5.5). For other species of fish, there were significantly more caught in proximal beds but only on one of the two occasions (e.g. *Pelates sexlineatus* – Figure

5.7I; *Pseudogobius* sp. – Figure 5.7K, *Urocampus carinirostris* – Figure 5.7L). Finally, the brown saber toothed blenny, *Petroscirtes lupus* was always more abundant in the distal seagrass beds (Figure 5.7J), although the magnitude of that difference varied between dense and sparse seagrass and between the two periods sampling was done (3-way interaction, Table 5.5).

In contrast to the patterns for the majority of species of fish, epibenthic invertebrates were more abundant in the seagrass beds further away from the mangroves, than those that were closer. There was a 10-fold increase in the abundance of *Palaemon debilis* (Figure 5.7N) in the distal beds compared with the proximal ones, with no effect of seagrass density either as a main effect or as an interaction (Table 5.5). *Latreutes pygmaeus* was more abundant in distal than proximal beds for both dense and sparse seagrass, although only the former was significant (Figure 5.7P). *Latreutes porcinus* was also significantly more abundant in distal than proximal dense seagrass beds but only on one occasion. There was no effect of proximity in the sparse seagrass (Figure 5.7O). Finally, *Idiosepius notoides* was generally more abundant in distal than proximal beds, but this was only significant in dense seagrass sampled on one occasion.

5.3.3.4. Overall Effects of Proximity

Overall, the above sampling programme potentially provided, for each species or variable that was analysed, up to four independent tests of whether proximity had a significant effect on the abundance of epibenthic nekton: one sampling period in eastern Moreton Bay and three in western Moreton Bay (Table 5.6). For many species though, regional variation in abundance (a species only occurred in either western or eastern Moreton Bay) or temporal variation in abundance (a species was not sampled on all occasions in western Moreton Bay) reduced the number of cases where data were available for all four periods (Table 5.6). As discussed above, on some occasions the abundance of a species was greater in the proximal (or distal) beds, even though those differences were not detected as being significant in post-hoc tests. We calculated the probability that the abundance of any species was greater in proximal than distal beds more often than would be expected by chance using a binomial test on the results from these four independent sets of data. If the abundance of a species was greater in the proximal than distal beds, then the probability of obtaining that result is 0.0613 ($\text{Prob(r)} = \frac{n!}{r!(n-r)!} p^r (1-p)^{(n-r)}$) = ($\text{Prob(r)} = \frac{4!}{0!(4)!} 0.5^0 (1-0.5)^{(4)}$) = 0.0613). Given there are only four tests that are possible, a probability of P=0.06 was interpreted as indicating a significant departure from the chance occurrence of a greater abundance of animals occurring in proximal than distal beds. We determined the probability of obtaining this result separately for dense and sparse seagrass, given the large number of cases where there was a significant interaction between the effects of seagrass density and proximity when examining the abundance of individual taxa on each occasion (see above results).

There were eighteen (18) cases where there were sufficient data (i.e. information was available from all four independent tests) to calculate this probability (Table 5.6). For six (6) of these cases there was a significant departure from chance, in that for all four tests, there was a greater number of animals or species caught in proximal than distal seagrass beds (number of fish – dense beds; species richness of fish – dense beds; number of *Tetractenos hamiltoni* and *Centropogon australis* – dense and sparse beds). For four (4) other variables, there were more individuals or species caught in proximal than distal beds in 3 of the 4 tests (species richness of fish – sparse seagrass; number of invertebrates – dense and sparse seagrass; number of *Urocampus carinirostris* – dense seagrass). Additionally, there were five (5) cases where the number of individuals or species was greater in distal than proximal beds (number of fish – sparse beds; species richness of invertebrates – dense and sparse beds; number of *Palaemon debilis* – dense and sparse beds). Two species of goby, *Arenigobius frenatus* and *Favonigobius frenatus*, only occurred in eastern Moreton Bay, so it was only possible to obtain three independent tests of the hypothesis. For both these species in dense seagrass, there were more individuals found in the patches close to the mangroves than those further away on all three occasions (Table 5.6). *Favonigobius frenatus* was also more abundant in proximal than distal sparse seagrass on all three occasions it was sampled.

5.4. Discussion

Changes in the connectivity between patches of seagrass and mangroves had a marked effect on the composition and abundance of the nekton assemblages occupying nearshore seagrass beds. Overwhelmingly, larger numbers of individuals and species were associated with seagrass beds that were close to the mangroves than those further away, an outcome consistent with previous studies on subtropical estuarine prawns (Skilleter et al., 2005) and temperate fishes (Jelbart et al, 2007). For many species, this pattern was repeated in dense and sparse seagrass beds, indicating that for those species the influence of potential connectivity between the two types of habitats (i.e. seagrass and mangroves) had a greater influence on nekton than structural complexity. For other species, there was a pronounced effect of proximity between the seagrass and mangroves in dense seagrass but not in sparse patches, or the pattern was reversed and more animals were present in the distal than proximal patches. Clearly though, the way in which patches of habitat are used by nekton in these subtropical estuarine systems is influenced by the surrounding landscape and individual patches or types of habitat are not independent of the surrounding matrix. Such interactions among and between habitats have been well described in terrestrial systems (e.g. Kareiva, 1990; Dunning et al., 1992; Johnson et al., 1992), but there is considerably less information for marine and estuarine communities (but see Irlandi and Crawford, 1997; Eggleston et al., 1998; Micheli and Peterson, 1999; Hovel and Lipcius, 2001, 2002 for some notable exceptions).

Most of the species that were caught in the intertidal seagrass beds must have moved into that habitat from subtidal areas on the rising tide (Skilleter and Loneragan, 2003; Sheaves, 2005). Very few species remain in the intertidal seagrass during low tide, with the exception of the penaeid prawns and portunid crabs (Vance et al., 1994; Skilleter, unpublished data). The subtidal habitat downshore from each of the areas sampled was primarily seagrass, *Zostera capricorni*, (Zharikov et al., 2005) providing a dense and structurally complex refuge into which nekton could retreat at low tide (Irlandi and Crawford, 1997; Sheaves, 2005; Jelbart et al., 2007). The intertidal seagrass was separated from the mangroves by a band of unvegetated sediments. This intervening matrix, fine muds in western Moreton Bay and coarser sand in the eastern region, provides little in the way of structural refuges for nekton moving to and from the mangroves during high tide (Sheaves, 2005). Although the seagrass beds closer to the mangroves were higher on the shore than those further away, it is unlikely that differences in tidal inundation time for the different treatments could account for the observed patterns. The intertidal gradient on the shores in Moreton Bay is relatively small and there was only a 10-15 minute interval between the time the sites low on the shore were inundated and when the water reached the sites closer to the mangroves. Both proximal and distal seagrass beds were accessible to nekton for similar amounts of time during each high tide, so the variation in use by nekton of these different mosaics is a function of the different connectivity between mangroves and seagrasses.

The relatively high rates of production of organic matter via litterfall within mangroves forests is converted to detritus (Hogarth, 1999) which is then exported to adjacent habitats (Gong et al., 1984; Robertson et al., 1991). Seagrass beds that are closer to adjacent mangrove forests are likely to receive greater amounts of this exported detritus than those beds that are further away, in turn providing enhanced food supplies for detritivores. The distribution of juveniles prawns, especially *Penaeus plebejus* and *Metapenaeus bennettae,* was consistent with this model, with greater numbers in proximal that distal beds (Skilleter et al., 2005; Section 4 this report). The stable isotope signature of prawns caught in mangrove creeks indicates that they could be assimilating some of the carbon from mangrove sources, whereas those further away assimilate little carbon from mangroves (Loneragan et al., 1994). Detritus and small detritivores are an important component of the diet of juvenile prawns (Ruello, 1973; Wassenberg and Hill, 1987), so prawns may have obtained some dietary benefit from their use of seagrass beds close to mangroves. A range of other detritivorous species including flat-tailed mullet, *Liza argentea*, sea mullet *Mugil cephalus*, and silver mullet, *Valamugil georgii* (Miller and Skilleter, 2006) were abundant in nearby mangroves but were only rarely caught in the seagrass beds (Section 1 this report). It seems likely that these species may use the mangroves preferentially during high tide, perhaps because of the increased availability of *in situ* detritus, then move as schools back to subtidal refuges as the tide recedes, without utilising the nearby seagrass for extended periods of time. If this is the case, they would not be expected to gain a benefit from using seagrass close to the mangroves.

During high tide, when mangroves are inundated, nekton may take advantage of the additional shelter provided by the shallow water (McIvor and Odum, 1988; Vance et al., 1996; Paterson and Whitfield, 2000) and/or the increased structural complexity found within mangroves (Rozas and Odum, 1987; Laegdsgaard and Johnson, 1995; Raposa and Oviatt, 2000; Manson et al., 2005b). Mangroves also often support increased densities of macroinvertebrate prey compared with other nearby habitats (Laegdsgaard and Johnson, 1995; Sheridan 1997; Manson et al., 2005b), providing enhanced opportunities for foraging when the forest is flooded. There are also large numbers of small fish sheltering among the roots and pneumatophores within the forest, providing an enhanced food supply for piscivores specialising on small individuals (Sheaves, 2005). Movement by nekton between the mangroves and seagrass beds, whether for increased shelter or food resources, would be enhanced when the two types of habitat are closer together than when they are more distant to each other because of the reduction in time needed to cross the intervening mudflats where there is little protection from predators. The expectation would be that mangrove-associated species (e.g. Sheaves, 2005) would have greater densities in the patches of seagrass close to mangroves than those further away, a pattern previously described in relation to connectivity between seagrass and mangroves in temperate systems (Jelbart et al., 2007). Many of our results also support this model.

Macrobenthic predators, such as *Acanthopagrus australis*, *Tetractenos hamiltoni*, *Gerres subfasciatus* and *Centropogon australis* (Bell et al., 1978; Linke et al., 2001; Miller and Skilleter, 2006) were consistently more abundant in the proximal seagrass beds than in those further away. Another predator, the weeping toado, *Torquigener pleurogramma* (Potter et al., 1988), showed a similar pattern although numbers were too small to analyse. These species were also abundant in the nearshore mangroves (see also Jelbart et al., 2007), suggesting they move among different interconnected, estuarine habitats to forage (Skilleter and Loneragan, 2003).

The mangrove goby, *Mugilogobius stigmaticus*, and half-bridled goby, *Arenigobius frenatus*, were both more abundant in proximal than distal seagrass beds, consistent with results from a temperate estuary in Australia (Jelbart et al., 2007). Both species are closely associated with mangroves (Laegdsgaard and Johnson, 1995; Jelbart et al., 2007; Section 1, this report) and seagrass (York et al., 2006) and *Mugilogobius stigmaticus* also moves further upshore into the saltmarsh and clay pans (Thomas and Connolly, 2001). There is no published information on the diet of these gobies in seagrass or mangroves, but related species consume primarily meiofauna and small macrofauna, foraging within seagrass but also over bare mud (e.g. Robertson, 1980; Coull et al., 1995). *Mugilogobius stigmaticus* consume copepods and crab larvae when foraging in nearby saltmarsh (personal communication – Rod Connolly). Movement (and hence foraging patterns) of small seagrass-associated gobies are heavily influenced by predation pressure from larger piscivores (Robertson, 1982), so increased connectivity between the seagrass and mangroves may provide a combination of access to enhanced food supplies and protection from predators in the mangroves during high tide.

Pelates sexlineatus, the six-lined trumpeter, was also more abundant in proximal seagrass beds, but it is unlikely that this species was taking advantage of enhanced food or refuges in the mangroves. The diet of *P. sexlineatus* comprises mainly macro-crustaceans (amphipods, tanaids, ostracods and copepods), all common and abundant in seagrass beds (Sanchez-Jerez et al., 2002). *P. sexlineatus* has a strong association with seagrass (Edgar and Shaw, 1995; Gray et al., 1996; Sanchez-Jerez et al., 2002) and was only rarely caught in the mangroves in this study (Table 1.2, Section 1), although it has been reported to use mangroves elsewhere (Bell et al., 1984; Laegdsgaard and Johnson, 1995). Jelbart et al. (2007) also found that *P. sexlineatus* was more abundant in seagrass close to mangroves than in beds further away, in a temperate estuary in NSW. It is possible that the abundance of prey items was greater in proximal than distal beds, perhaps in response to elevated amounts of detritus exported from the mangroves supporting greater abundances of seagrass-associated benthic prey (see above), in turn providing a richer foraging habitat for the trumpeter. More specific dietary studies in relation to availability of food in different mosaics are required.

Some species, such as the hairy pipefish, *Urocampus carinirostris* and the carid shrimps, *Latreutes porcinus* and *L. pygmaeus*, showed a strong response to variation in the structural complexity (density) of the seagrass beds with greater numbers in the dense than sparse seagrass, but no consistent response to the connectivity between seagrass and the nearby mangroves. Hairy pipefish are seagrass specialists that have a sit-and-wait feeding strategy targeting small crustaceans, such as amphipods, mysids and copepods (Howard and Koehn, 1985). The effectiveness of this predation strategy is often enhanced in habitats with increased structural complexity (e.g. Heck and Orth, 1980; Coen et al., 1981;

Howard and Koehn, 1985), thus these species would benefit from being in the dense compared with sparse seagrass. The pipefish and the carid shrimps are, however, small and relatively slow moving and may be susceptible to predation when moving between different habitats, so are unlikely to move into nearby mangroves, across the unprotected mudflats. These species were rare or did not occur in samples collected from the adjacent mangrove forests. In contrast, the southern pygmy squid, *Idiosepius notoides*, also a sit-and-wait ambush predator consuming small crustaceans (Kasugai, 2001) was consistently more abundant in dense seagrass beds further away from the mangroves than those close to the forest. This species may avoid areas used extensively by larger predators transiting between the mangroves and adjacent habitats.

The presence or absence of mangroves in the intertidal zone may be a critical feature in maintaining patterns of increased biodiversity and biomass of fish assemblages in other nearshore habitats and this influence of the connectivity may operate over a range of spatial scales. Our results and those of Jelbart et al. (2007) indicate that connectivity is important within a specific estuarine system, over scales of 100's metres to kilometres. At larger scales (10-100's kms), Nagelkerken et al. (2001) found that the species richness and abundance of fish assemblages was markedly greater in seagrass beds with adjacent mangroves than those in areas without mangroves and Mumby et al. (2004) found that the presence of mangroves had significant positive effects on the biomass and community composition of fish utilising nearby coral reefs. These critical linkages between mangroves and other components of estuarine and coastal systems suggest that the loss or degradation of mangroves is therefore likely to have serious implications for the overall function and value of nearby habitats, if there is a reduction in habitat connectivity.

Globally, mangrove ecosystems are impacted by both acute and chronic disturbances, as a result of both natural and human influences. Acute disturbances within mangrove forests include clearing and reclamation for canal estates, aquaculture, storm damage, residential and industrial development and oil spills (Lugo et al., 1981; Hatcher et al., 1989; Levings and Garrity, 1994; Ellison and Farnsworth, 1996; Rönnbäck, 1999). In comparison to acute disturbances, little research or conservation effort has focused on chronic forms of disturbance (e.g. Adam, 1984). Chronic disturbances may impact either directly or indirectly on mangrove ecosystems. For example, mangrove ecosystems may be directly impacted by human activities such as trampling, rubbish disposal and the construction of structures within the forest, such as boardwalks, jetties and pipelines (Hutchings and Saenger, 1987; Skilleter and Warren, 2000). Alternatively, activities within the catchment, such as increased agriculture and urbanisation, may impact indirectly on mangrove ecosystems, by changing freshwater regimes, sedimentation and increasing deterioration of water quality (e.g. Loneragan and Bunn, 1999). Clearly, the potential effects of lost connectivity between mangroves and other habitats, such as seagrass, need to be considered in relation to any understanding of the effects of human activities on estuarine and coastal ecosystems.

Table 5.1: Results of SIMPER (Similarity Percentages) analysis showing the contributions to differences in composition for the nekton assemblage caught in intertidal seagrass beds in the eastern and western Moreton Bay during the day in September-November 2002. The average abundance for each species is shown, plus the % contribution of each species to the separation of the nekton assemblages in each region. Only those species contributing up to a total of ~ 80 %, between the two regions are shown. Data are the number of animals caught in 100 m^2 of seagrass (N = 44 seine net samples from western Moreton Bay and N = 48 seine net samples from eastern Moreton Bay pooled across two sites in each of four categories of seagrass bed (see text for further details).

	Eastern	**Western**	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Penaeus plebejus	65.17	19.48	18.24
Favonigobius lentiginosus	26.96	0.07	9.39
Urocampus carinirostris	18.23	6.57	6.89
Palaemon debilis	9.27	12.34	6.58
Arenigobius frenatus	4.56	50.89	6.15
Pelates sexlineatus	5.19	14.66	5.12
Lucifer hanseni	0.04	9.66	4.71
Centropogon australis	6.27	14.18	4.28
Metapenaeus bennettae	0.00	13.32	4.19
Latreutes porcinus	0.48	10.66	3.53
Palaemon serenus	6.48	1.16	3.01
Latreutes pygmaeus	0.10	9.20	2.80

Table 5.2: Results of SIMPER (Similarity Percentages) analysis showing the contributions to differences in composition for the nekton assemblage caught in four different categories of intertidal seagrass beds in the eastern Moreton Bay during the day in September- November 2002. The average abundance for each species is shown, plus the % contribution of each species to the separation of the nekton assemblages in each category. Only those species contributing up to a total of ~ 80 %, between the two regions are shown. Comparisons are shown for main effects of Proximity, Density and the interaction between Proximity and Density. Data are the number of animals caught in 100 m^2 of seagrass (N = 12 seine net samples pooled across two sites in each of four categories of seagrass bed (see text for further details).

Effect of Proximity	Proximal	Distal	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Penaeus plebejus	19.58	110.75	32.10
Favonigobius lentiginosus	26.63	27.29	14.75
Urocampus carinirostris	14.04	22.42	8.43
Palaemon debilis	6.54	12.00	7.91
Palaemon serenus	0.63	12.33	7.13
Pelates sexlineatus	3.88	6.50	3.68
Idiosepius notoides	7.83	3.83	3.46
Centropogon australis	4.71	7.83	3.27

Effect of Density	Dense	Sparse	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Penaeus plebejus	107.13	23.21	30.73
Favonigobius lentiginosus	38.08	15.83	15.58
Urocampus carinirostris	20.46	16.00	8.40
Palaemon debilis	7.08	11.46	8.06
Palaemon serenus	0.21	12.75	5.85
Idiosepius notoides	9.00	2.67	4.18
Pelates sexlineatus	4.71	5.67	3.62
Centropogon australis	7.04	5.50	3.18

Effect of Proximity x Density	Proximal - Dense	Distal - Dense	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Penaeus plebejus	183.42	30.83	41.86
Favonigobius lentiginosus	46.75	29.42	14.24
Urocampus carinirostris	27.50	13.42	6.91
Palaemon debilis	3.25	10.92	5.45
Arenigobius leftwichi	24.42	2.83	3.90
Idiosepius notoides	5.75	12.25	3.78
Centropogon australis	8.58	5.50	2.44
Pelates sexlineatus	4.67	4.75	2.43

Effect of Proximity x Density	Proximal – Sparse	Distal - Sparse	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Penaeus plebejus	38.08	8.33	24.32
Palaemon serenus	24.25	1.25	15.09
Favonigobius lentiginosus	7.83	23.83	13.96
Palaemon debilis	20.75	2.17	12.20
Urocampus carinirostris	17.33	14.67	10.10
Pelates sexlineatus	8.33	3.00	4.82
Centropogon australis	7.08	3.92	4.29
Hippolyte sp.	2.33	5.17	3.96

Table 5.3: Results of SIMPER (Similarity Percentages) analysis showing the contributions to differences in composition for the nekton assemblage caught in four different categories of intertidal seagrass beds in the western Moreton Bay during the day in September - November 2002. The average abundance for each species is shown, plus the % contribution of each species to the separation of the nekton assemblages in each category. Other details as in Table 5.2.

Effect of Proximity	Proximal	Distal	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Arenigobius frenatus	91.75	1.85	20.88
Penaeus plebejus	30.83	5.85	9.63
Lucifer hanseni	14.25	4.15	8.61
Metapenaeus bennettae	23.50	1.10	8.52
Palaemon debilis	13.38	11.10	6.43
Pelates sexlineatus	18.42	10.15	6.02
Centropogon australis	21.13	5.85	5.78
Latreutes porcinus	11.54	9.60	5.44
Latreutes pygmaeus	7.92	10.75	4.59
Urocampus carinirostris	10.00	2.45	3.10

Effect of Density	Dense	Sparse	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Arenigobius frenatus	100.65	9.42	21.25
Latreutes porcinus	22.20	1.04	7.87
Penaeus plebejus	26.70	13.46	7.56
Metapenaeus bennettae	21.25	6.71	7.16
Lucifer hanseni	0.35	17.42	7.16
Centropogon australis	25.80	4.50	6.64
Pelates sexlineatus	21.35	9.08	6.57
Palaemon debilis	9.70	15.54	5.77
Latreutes pygmaeus	11.60	7.21	4.48
Urocampus carinirostris	11.75	2.25	3.34
Favonigobius exquisitus	7.00	1.88	2.93
Gnatholepis gymnocara	4.00	0.04	1.99

Effect of Proximity x Density	Proximal - Dense	Distal - Dense	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Arenigobius frenatus	166.84	1.38	31.30
Metapenaeus bennettae	34.08	2.00	8.74
Latreutes porcinus	22.00	22.50	7.17
Penaeus plebejus	38.08	9.63	6.79
Centropogon australis	36.25	10.13	6.73
Pelates sexlineatus	26.42	13.75	6.59
Latreutes pygmaeus	15.25	6.13	3.74
Favonigobius exquisitus	6.67	7.50	3.13
Urocampus carinirostris	16.17	5.13	3.10
Gnatholepis gymnocara	3.67	4.50	2.67
Palaemon debilis	14.33	2.75	2.65

Effect of Proximity x Density	Proximal – Sparse	Distal - Sparse	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Lucifer hanseni	28.00	6.83	16.88
Penaeus plebejus	23.58	3.33	14.35
Arenigobius frenatus	16.42	2.17	10.82
Palaemon debilis	12.42	16.67	10.47
Metapenaeus bennettae	12.92	0.50	7.77
Pelates sexlineatus	10.42	7.75	6.03
Latreutes pygmaeus	0.58	13.83	5.84
Centropogon australis	6.00	3.00	4.28
Idiosepius notoides	2.08	6.33	3.22

Table 5.4: Results of analyses of variance comparing the abundance (per 100 m^2)of fish and invertebrates (decapods and cephalopods) sampled in seagrass beds of different complexity (dense versus sparse) and proximity to mangroves (proximal and distal) in eastern and western Moreton Bay during spring (September-November, 2002). Data were transformed to $\log_e(x+1)$ where indicated. N = 4 replicate samples from each of three sites per category of seagrass. P-values are shown. The terms in the analyses that were interpreted and presented graphically are shown in italics.

Eastern Moreton Bay

Variable	Proximity	Density	P x D	Sites (P x D)
Individuals – fish[1]	0.109	0.017	*0.032*	0.005
Species – fish	0.930	0.070	1.000	0.005
Arenogobius leftwichi[1]	0.007	0.001	*0.006*	0.003
Favonigobius lentiginosus[1]	0.186	0.006	*0.010*	0.010
Centropogon australis[1]	0.174	0.244	0.970	0.001
Pelates sexlineatus[1]	0.577	0.364	0.368	0.005
Urocampus carinirostris[2]	0.329	0.599	0.545	0.001
Penaeus plebejus[3]	*0.032*	*0.026*	0.589	0.001
Hippolyte sp.[1]	0.344	0.371	0.559	0.001
Palaemon debilis[1]	0.168	0.126	*0.033*	0.001
Idiosepius notoides	*0.020*	*0.002*	0.108	0.202

Western Moreton Bay

Variable	Proximity	Density	P x D	Sites (P x D)
Individuals – fish[1]	0.004	0.003	*0.008*	0.001
Species – fish	*0.003*	*0.003*	0.194	0.001
Arenogobius frenatus[1]	0.001	0.007	*0.012*	0.013
Favonigobius exquisitus[1]	0.732	0.138	0.786	0.010
Centropogon australis[1]	0.008	0.002	*0.021*	0.016
Pelates sexlineatus[1]	0.922	0.456	0.974	0.001
Urocampus carinirostris[2]	*0.006*	*0.003*	0.474	0.431
Penaeus plebejus[3]	*0.047*	0.132	0.748	0.002
Metapenaeus bennettae[3]	0.001	0.002	*0.012*	0.819
Latreutes pygmaeus[1]	0.417	0.589	0.683	0.001
Latreutes porcinus[1]	0.641	0.019	0.718	0.001

1. Data transformed
2. Figure not shown
3. Results reported in Skilleter et al. (2005).

Table 5.5: Results of analyses of variance comparing the abundance (per 100 m^2) of fish and invertebrates (decapods and cephalopods) sampled in seagrass beds of different complexity (dense versus sparse) and proximity to mangroves (proximal and distal) in western Moreton Bay on two occasions in summer (January-February, 2003). Data were transformed to $\log_e(x+1)$ where indicated. N = 6 replicate samples from each of two sites per category of seagrass. P-values are shown. The terms in the analyses that were interpreted and presented graphically are shown in italics.

Temporal Comparisons

Variable	Period	Proximity	Density	Period x Proximity	Period x Density	Proximity x Density	Period x Proximity x Density	Sites (P x D)	Period x Sites (P x D)
Species (fish + invertebrates)	0.017	0.470	*0.001*^{A}	0.496	0.130	0.513	0.496	0.163	0.152
Species (fish only)	0.002	0.402	0.001	0.452	*0.050*	*0.001*^{A}	1.000	0.275	0.589
Individuals – fish[1]	0.556	0.118	0.001	0.791	0.709	*0.001*^{A}	0.956	0.275	0.204
Arenogobius frenatus[1]	0.148	0.060	0.001	0.583	0.824	*0.001*^{A}	0.769	0.182	0.145
Favonigobius exquisitus	0.710	0.110	0.510	0.280	*0.040*	*0.003*^{A}	0.750	0.120	0.130
Gnatholepis gymnocara[1]	0.220	0.162	0.614	0.847	0.367	0.438	0.159	0.070	0.003
Pseudogobius sp.	0.015	0.028	0.028	0.015	0.015	0.028	*0.015*	0.317	0.205
Centropogon australis[1]	0.304	0.019	0.008	0.242	0.331	*0.010*	0.874	0.282	0.002
Gerres subfasciatus	0.584	0.332	0.897	0.944	0.177	0.682	0.459	0.040	0.538
Mugilogobius stigmaticus[1]	0.308	0.388	0.361	0.469	0.308	0.388	0.469	0.002	0.370
Pelates sexlineatus	0.009	0.381	0.040	0.045	0.117	0.048	*0.015*	0.174	0.840
Petroscirtes lupus[1]	0.250	0.107	0.726	0.445	0.890	0.630	*0.046*	0.071	0.257
Tetractenos hamiltoni[1]	0.709	0.001	0.001	0.902	0.144	*0.001*	0.737	0.757	0.020
Urocampus carinirostris[1]	0.147	0.698	0.409	0.475	0.717	0.077	*0.021*^{B}	0.396	0.000
Penaeus plebejus[2]	0.794	0.002	0.016	0.979	0.222	0.787	*0.002*	0.516	0.001
Penaeus esculentus[2]	0.380	0.098	0.836	0.323	0.733	0.918	*0.023*	0.001	0.008
Metapenaeus bennettae[2]	0.923	0.004	0.030	0.502	0.847	0.576	*0.040*	0.001	0.001
Metapenaeus ensis[2]	0.908	0.520	0.167	0.073	0.312	0.516	*0.060*	0.001	0.001
Latreutes porcinus[1]	0.258	0.001	0.005	0.175	0.301	0.002	*0.001*^{B}	0.105	0.473
Latreutes pygmaeus[1]	0.673	0.217	0.819	0.713	0.400	0.550	0.932	0.025	0.193
Palaemon debilis	0.213	*0.033*	0.153	0.286	0.480	0.148	0.812	0.313	0.003
Idiosepius notoides[1]	0.619	0.004	0.784	0.019	0.010	0.798	*0.002*	0.337	0.356

1. Data transformed
2. Results reported in Skilleter et al. (2005).

A. Test using pooled term for Period x Sites (Proximity x Density) + Sites (Proximity x Density) + Error (with 88 degrees of freedom)
B. Test using pooled term for Sites (Proximity x Density) + Error (with 84 degrees of freedom)

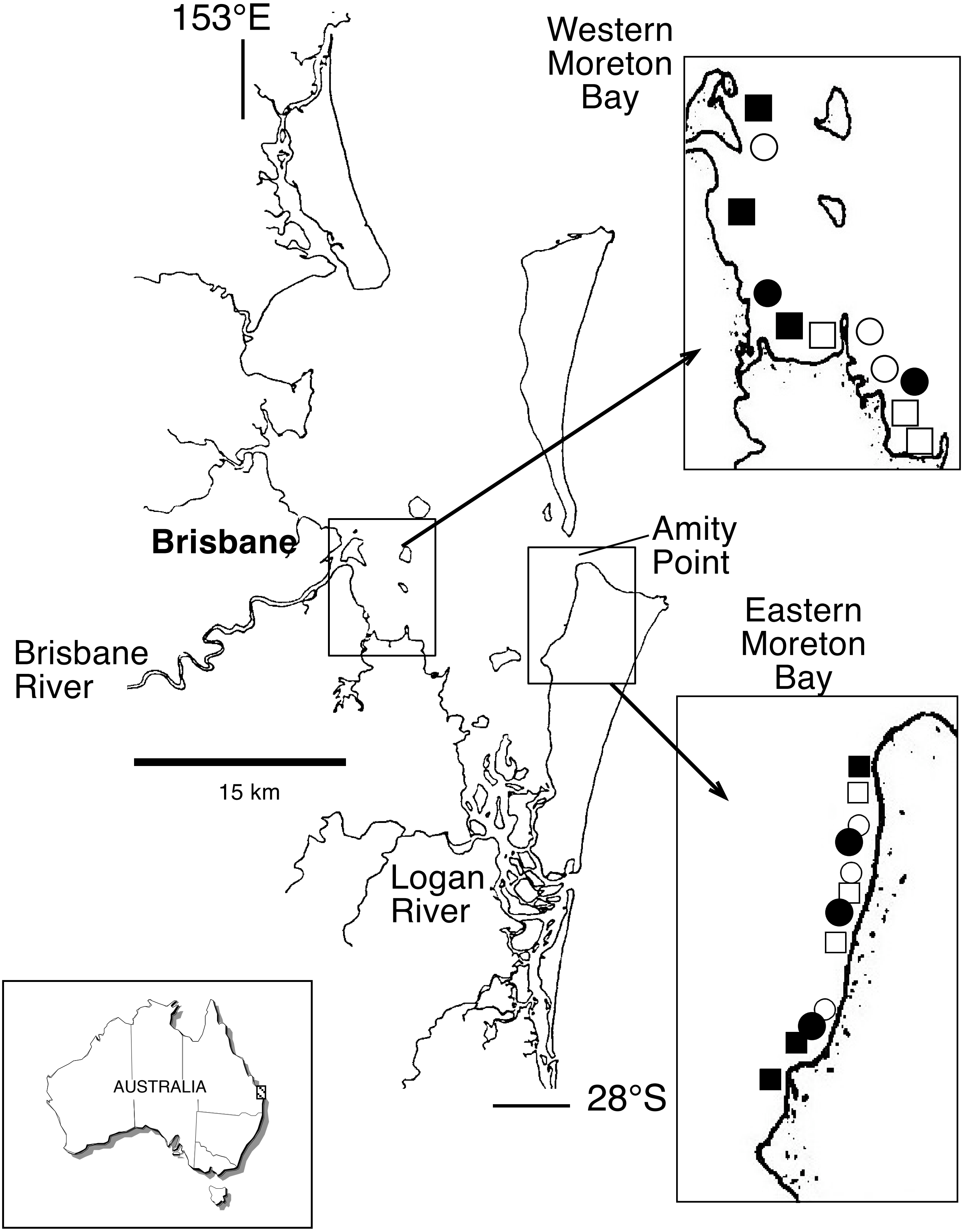


Figure 5.1: Map of Australia and the Moreton Bay region in South-east Queensland, showing the position of the study sites in western and eastern Moreton Bay. Four different categories of seagrass bed were sampled: proximal dense beds (filled squares), proximal sparse beds (clear squares), distal dense beds (filled circles) and distal sparse beds (clear circles). The position of beds of each category were interspersed as much as possible along the shoreline (see text for further details).

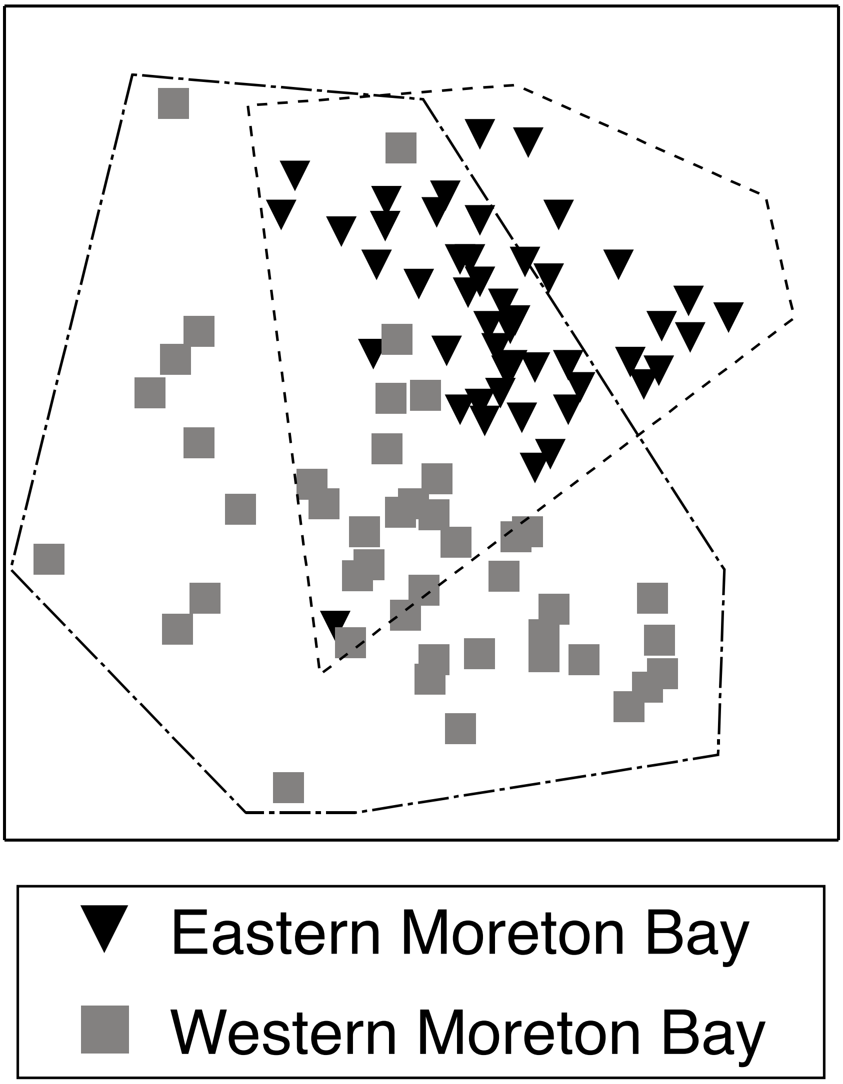


Figure 5.2: nMDS ordination of untransformed data on the composition of the nekton assemblage (fish, decapods and cephalopods) utilising intertidal seagrass beds in eastern and western Moreton Bay. N = 44 samples from western Moreton Bay and N = 48 samples from eastern Moreton Bay, pooled across sites in each of four different categories of seagrass bed (see text for further details). Dashed lines represent significantly different groups in ANOSIM

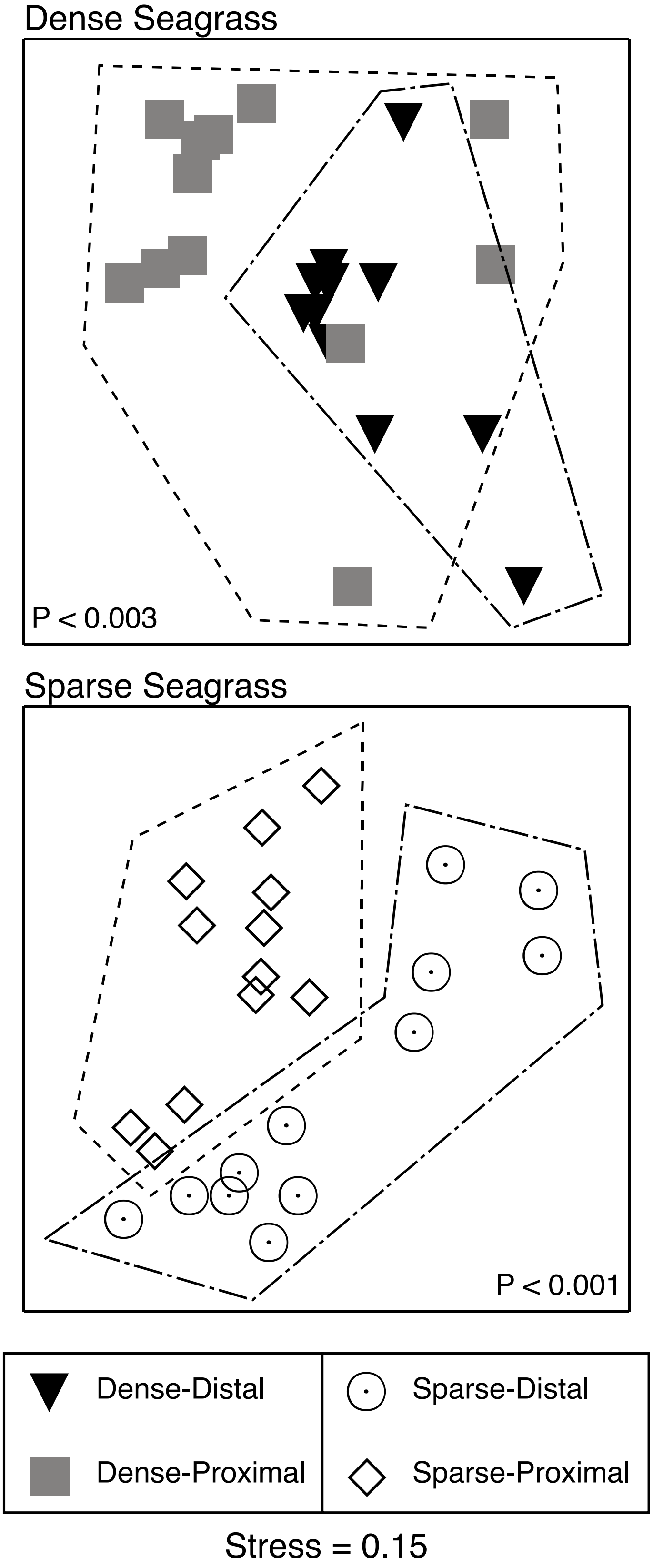


Figure 5.3: nMDS ordinations of untransformed data on the composition of the nekton assemblage (fish, decapods and cephalopods) utilising intertidal seagrass beds in eastern Moreton Bay. Comparisons are shown separately for dense and sparse seagrass, contrasting proximal and distal beds, for ease of viewing but are based on a single analysis. N = 48 samples from eastern Moreton Bay, pooled across sites in each of four different categories of seagrass bed (see text for further details). Dashed lines represent significantly different groups in ANOSIM. P values are from pair-wise tests after ANOSIM.

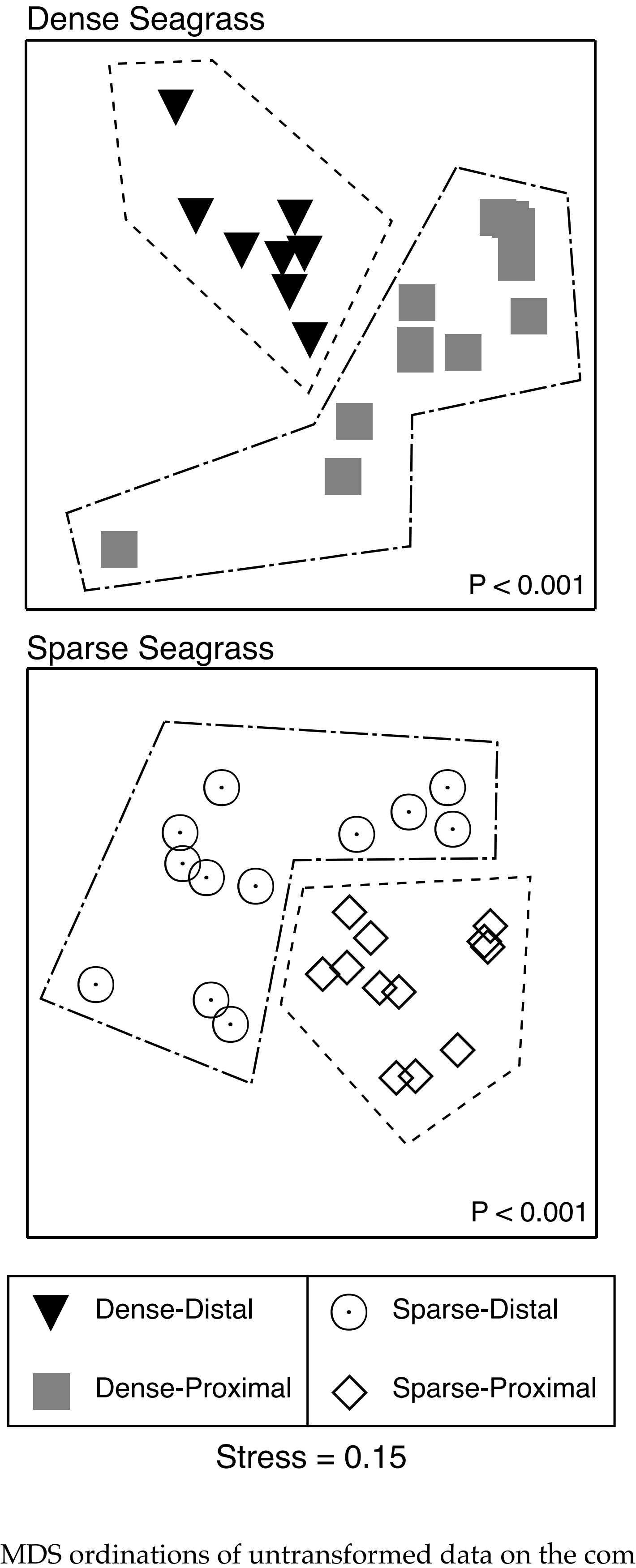


Figure 5.4: nMDS ordinations of untransformed data on the composition of the nekton assemblage (fish, decapods and cephalopods) utilising intertidal seagrass beds in western Moreton Bay. Comparisons are shown separately for dense and sparse seagrass, contrasting proximal and distal beds, for ease of viewing but are based on a single analysis. N = 48 samples from western Moreton Bay, pooled across sites in each of four different categories of seagrass bed (see text for further details). Dashed lines represent significantly different groups in ANOSIM. P values are from pair-wise tests after ANOSIM.

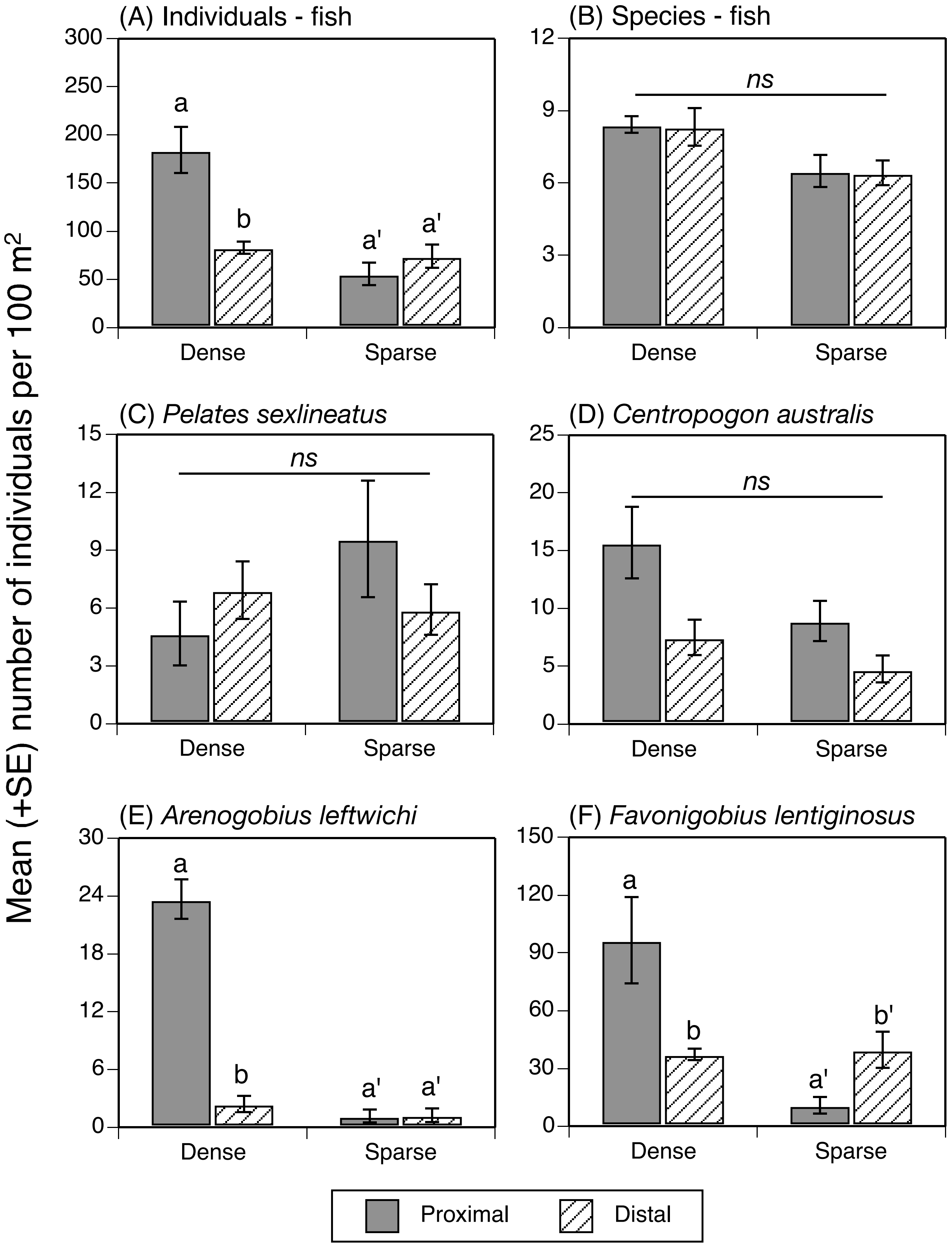


Figure 5.5: Mean (+SE) number of (A) fish individuals, (B) fish species, (C) *Pelates sexlineatus*, (D) *Centropogon australis*, (E) *Arenogobius leftwichi* and (F) *Favonigobius lentiginosus* per 100 m^2 of intertidal seagrass in four categories: proximal dense,proximal sparse, distal dense and distal sparse in eastern Moreton Bay, sampled in October-November, 2002. Results of SNK tests after ANOVA are shown: means topped by the same letter were not significantly different from each other (P > 0.05). A bar topping all means indicates no significant difference among treatments in ANOVA. Note the differences in the scale on the Y-axes.

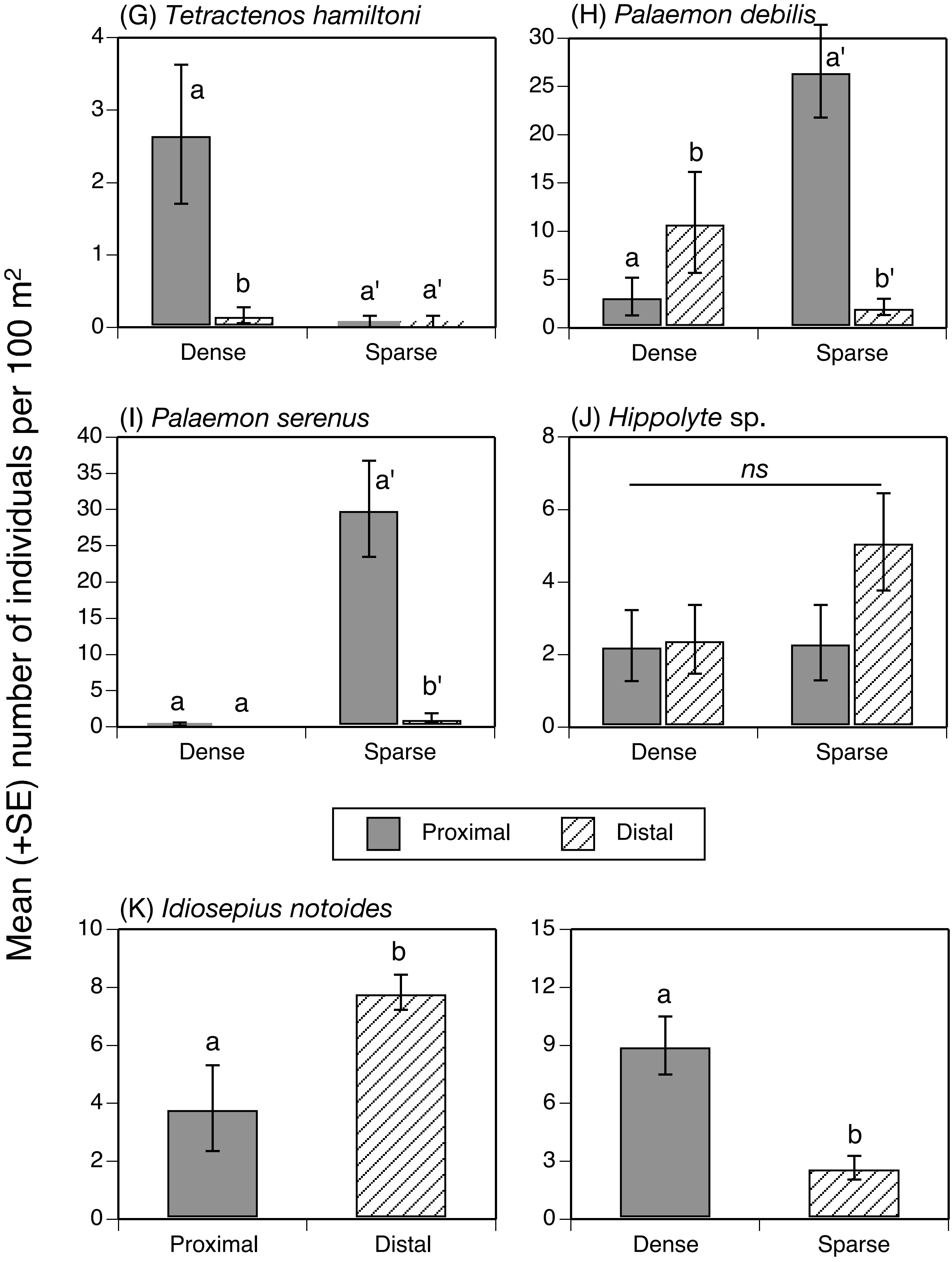


Figure 5.5 continued: Mean (+SE) number of (G) *Tetractenos hamiltoni*, (H) *Palaemon debilis*, (I) *Palaemon serenus*, (J) *Hippolyte* sp. and (K) *Idiosepius notoides* per 100 m^2 of intertidal seagrass in four categories: proximal dense, proximal sparse, distal dense and distal sparse in eastern Moreton Bay, sampled in October-November, 2002. Other details as in Figure 5.5

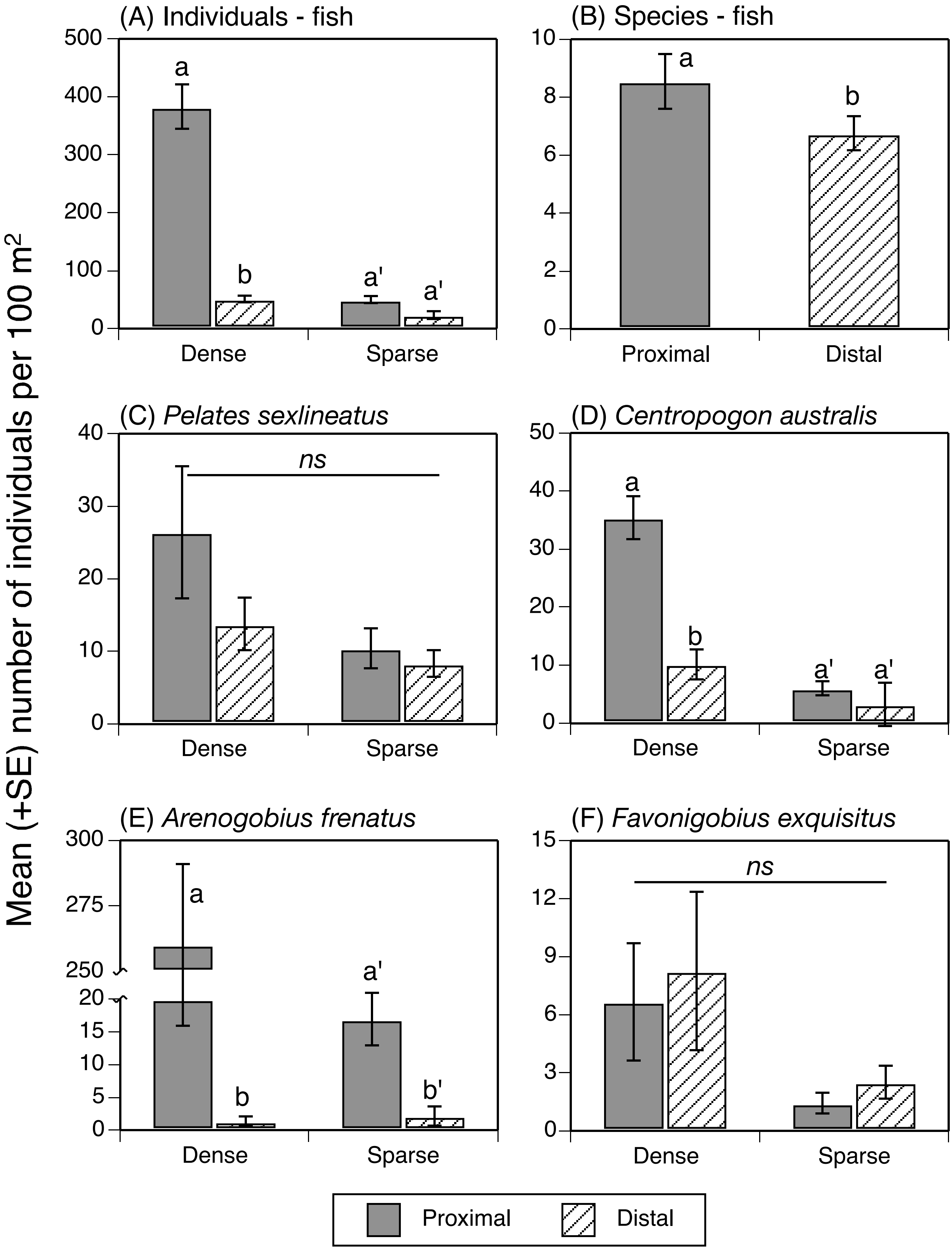


Figure 5.6: Mean (+SE) number of (A) fish individuals, (B) fish species, (C) *Pelates sexlineatus*, (D) *Centropogon australis*, (E) *Arenogobius frenatus* and (F) *Favonigobius exquisitus* per 100 m^2 of intertidal seagrass in four categories: proximal dense,proximal sparse, distal dense and distal sparse in western Moreton Bay, sampled in September-October, 2002. Results of SNK tests after ANOVA are shown: means topped by the same letter were not significantly different from each other ($P > 0.05$). A bar topping all means indicates no significant difference among treatments in ANOVA. Note the differences in the scale on the Y-axes.

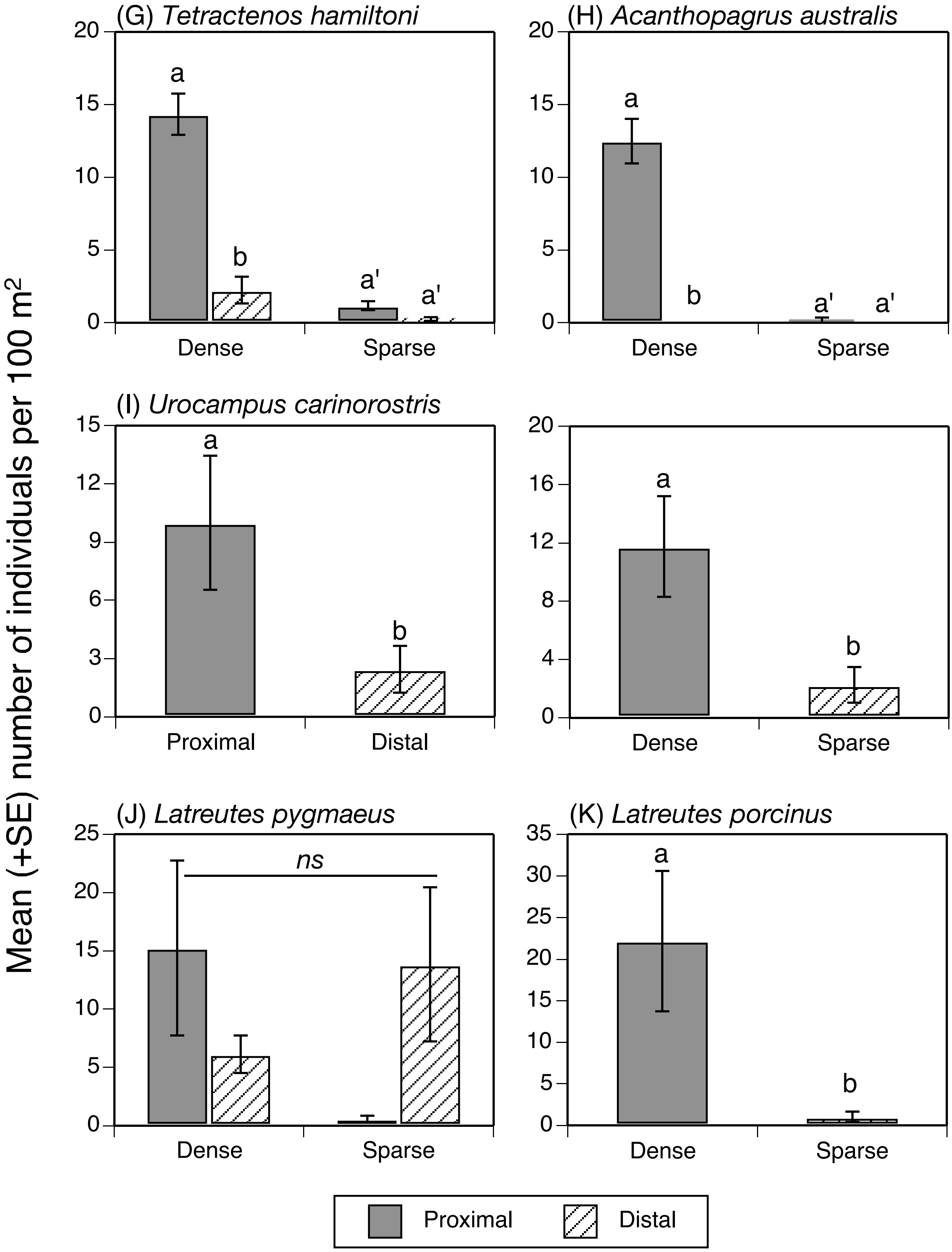


Figure 5.6 continued: Mean (+SE) number of (G) *Tetractenos hamiltoni*, (H) *Acanthopagrus australis*, (I) *Urocampus carinorostris*, (J) *Latreutes pygmaeus* and (K) *Latreutes porcinus* per 100 m^2 of intertidal seagrass in four categories: proximal dense,proximal sparse, distal dense and distal sparse in western Moreton Bay, sampled in September-October, 2002. Other details as in Figure 5.5.

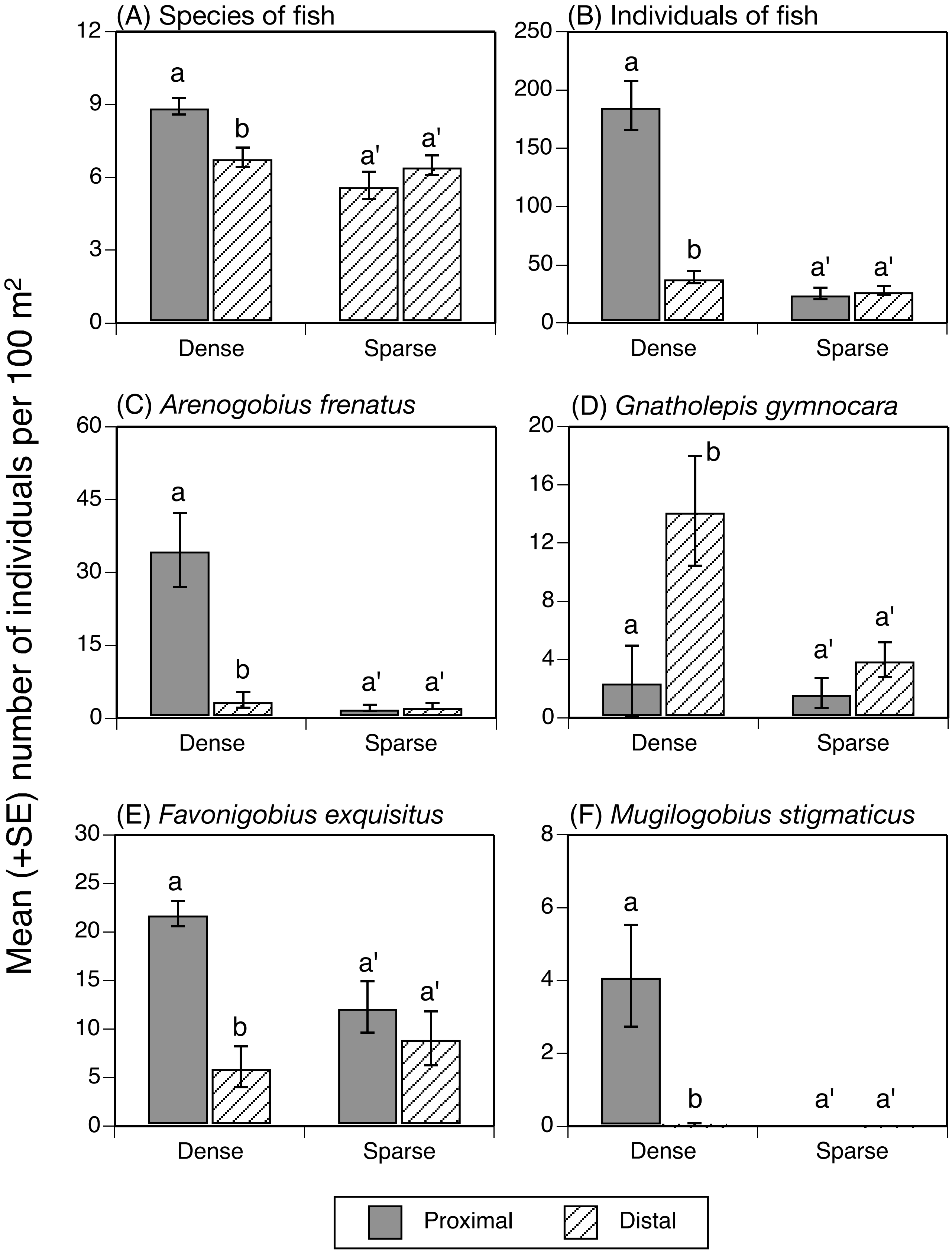


Figure 5.7: Mean (+SE) number of (A) fish species, (B) fish individuals, (C) *Arenigobius frenatus*, (D) *Gnatholepis gymnocara*, (E) *Favonigobius exquisitus* and (F) *Mugilogobius stigmaticus* per 100 m^2 of intertidal seagrass in four categories: proximal dense,proximal sparse, distal dense and distal sparse in western Moreton Bay, sampled on two separate occasions (Periods 1 & 2) in January-February, 2003. Other details as in Figure 5.5.

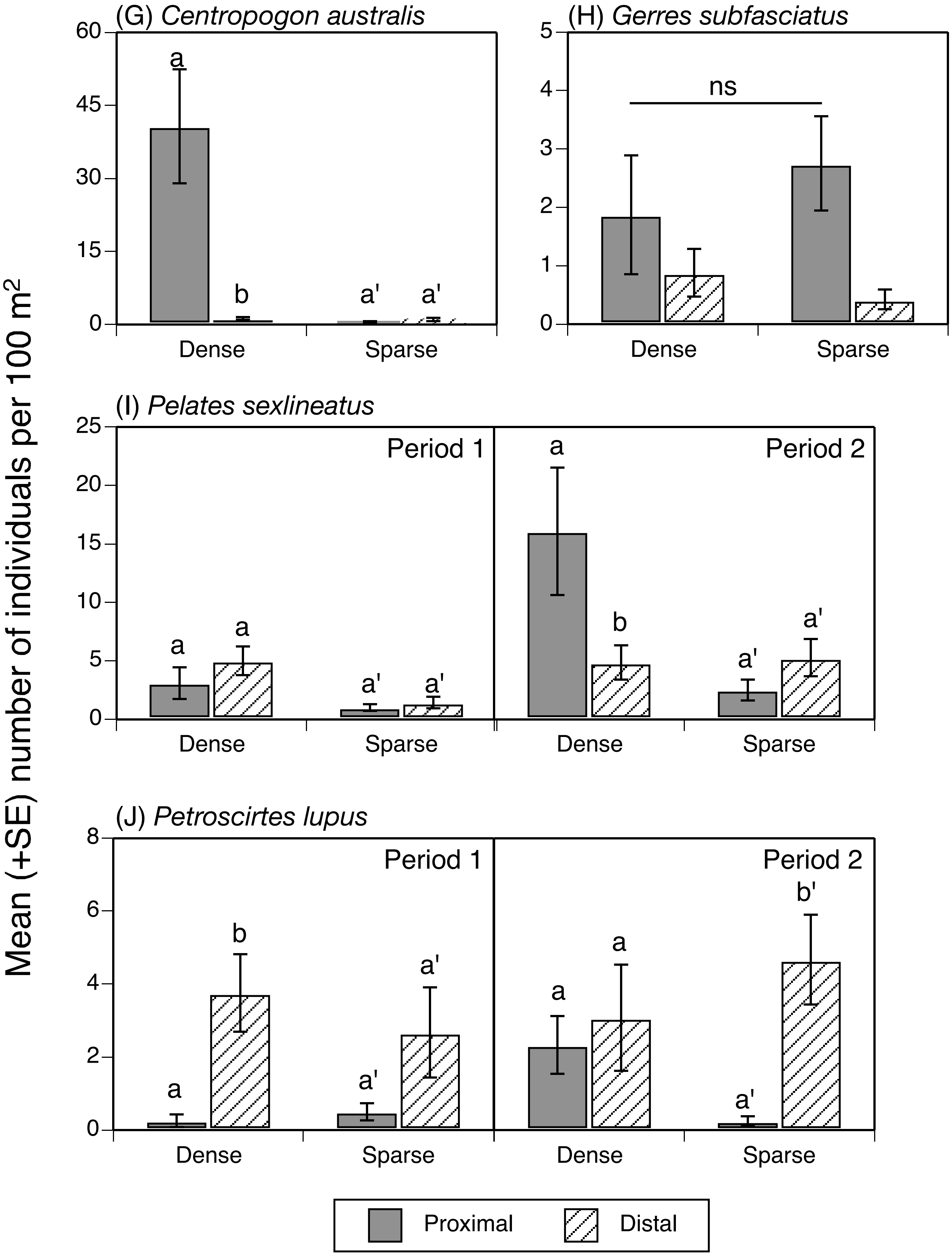


Figure 5.7 continued: Mean (+SE) number of (G) *Centropogon australis*, (H) *Gerres subfasciatus*, (I) *Pelates sexlineatus* and (J) *Petroscirtes lupus* per 100 m^2 of intertidal seagrass in four categories: proximal dense,proximal sparse, distal dense and distal sparse in western Moreton Bay, sampled sampled on two separate occasions (Periods 1 & 2) in January-February, 2003. Other details as in Figure 5.5.

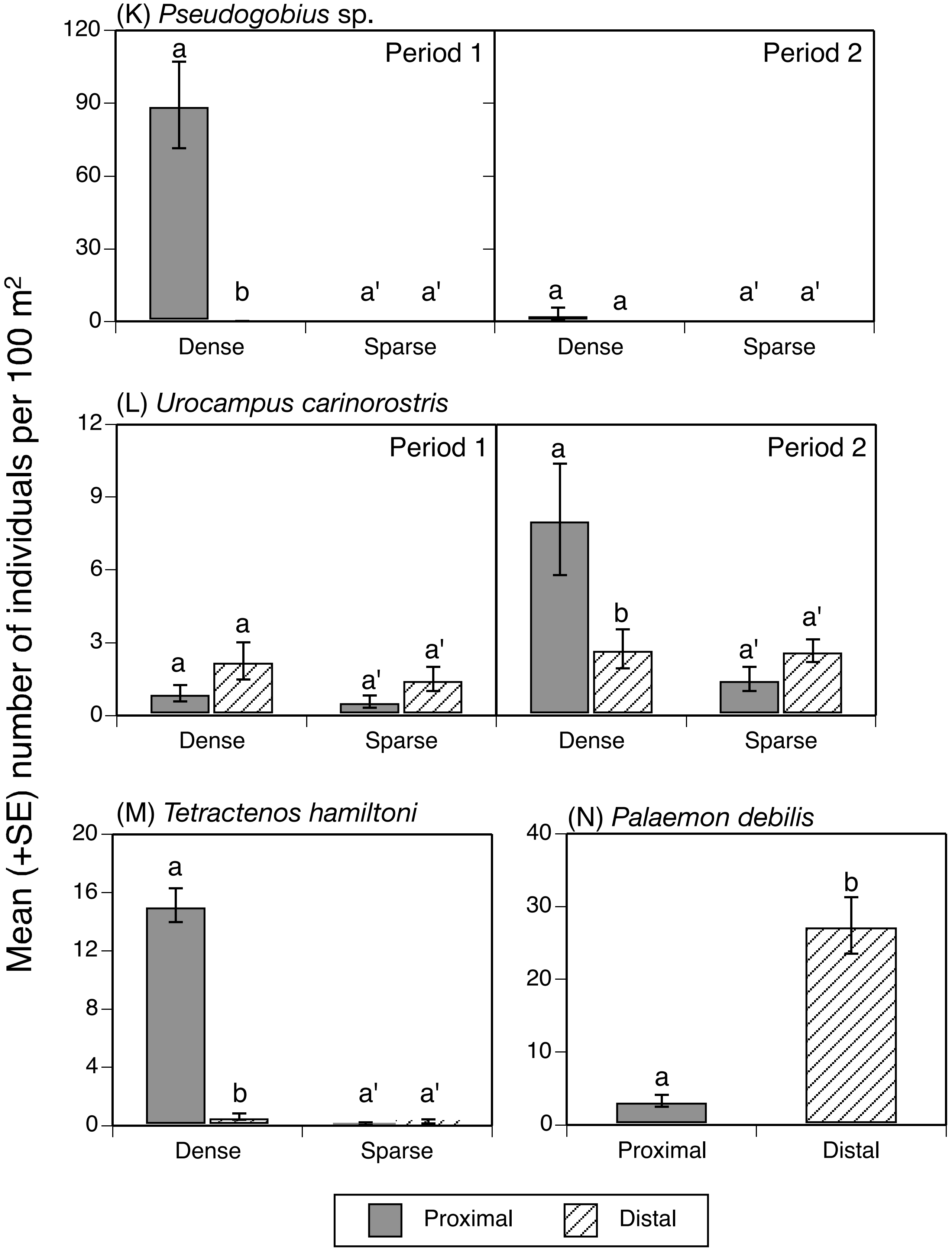


Figure 5.7 continued: Mean (+SE) number of (K) *Pseudogobius* sp., (L) *Urocampus carinorostris*, (M) *Tetractenos hamiltoni* and (N) *Palaemon debilis* per 100 m^2 of intertidal seagrass in four categories: proximal dense,proximal sparse, distal dense and distal sparse in western Moreton Bay, sampled in January-February, 2003. Other details as in Figure 5.5.

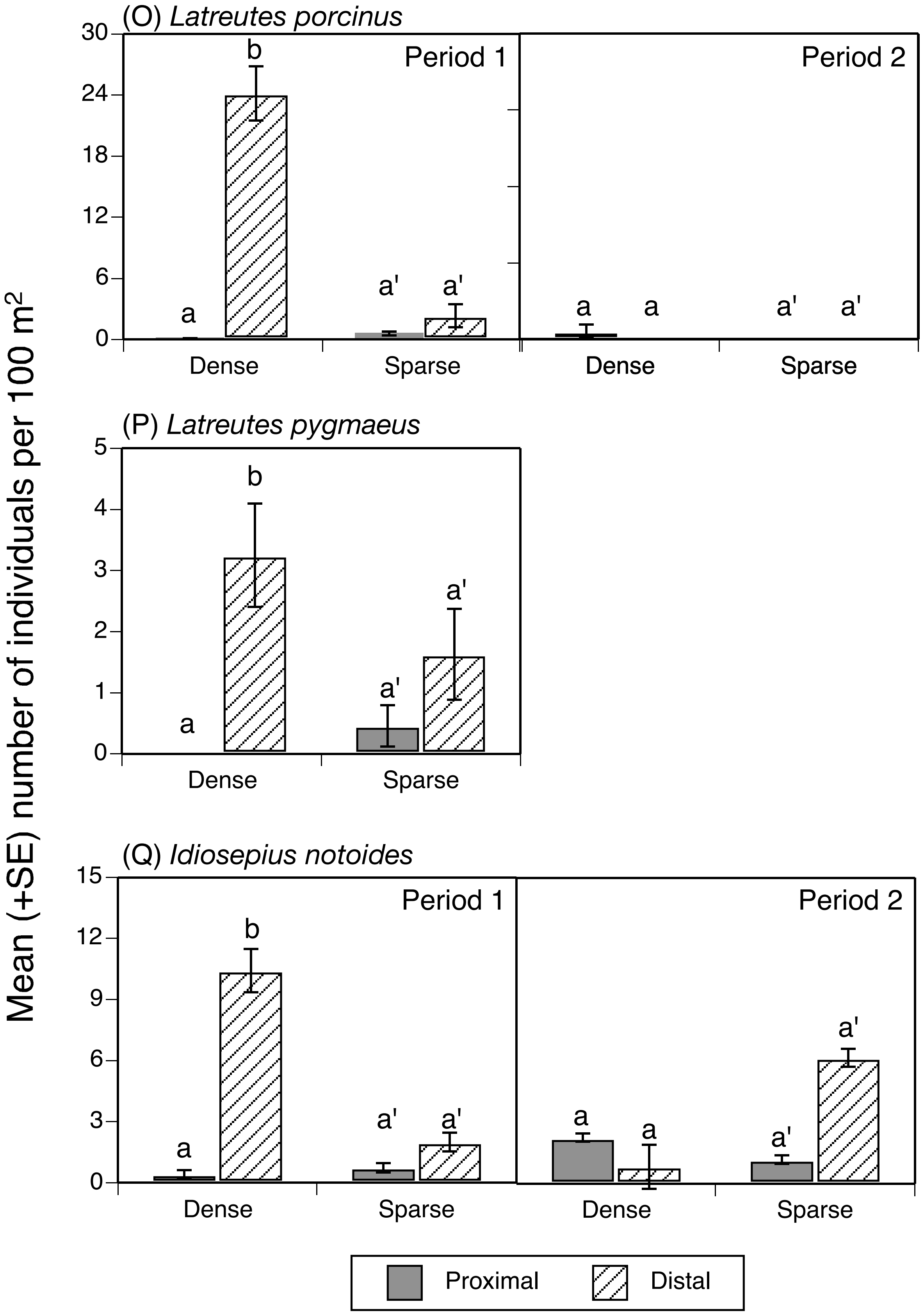


Figure 5.7 continued: Mean (+SE) number of (O) *Latreutes porcinus*, (P) *Latreutes pygmaeus* and (Q) *Idiosepius notoides* per 100 m^2 of intertidal seagrass in four categories: proximal dense,proximal sparse, distal dense and distal sparse in western Moreton Bay, sampled in January-February, 2003. Other details as in Figure 5.5.

Section 6: The Role of Corridors in Determining the Utilisation of Mangroves by Nekton

6.1. Introduction

Isolation of patches of habitat is considered to be one of the greatest threats to biodiversity (Saunders et al., 1991; Kareiva et al., 1993; Debinski and Holt, 2000), affecting a broad range of ecological processes (Collinge, 2000 and references therein). In terrestrial environments, corridors are considered to be an important mechanism to reduce the effects of habitat fragmentation (Saunders et al., 1991) and patches of habitat connected by suitable corridors are predicted to support greater densities and species richness than patches that are isolated (e.g. Fahrig and Merriam, 1985; Gilbert et al., 1998; Haddad and Baum, 1999; Pardini et al., 2005; but see Simberloff et al., 1992 for important considerations).

Movement through corridors may be hazardous for animals, affecting patterns of dispersal (Stevens et al., 2006), although it is rarely a complete barrier to dispersal (Castellón and Sieving, 2006). Connectivity of different habitats is a function though of the degree of movement between different patches by animals and the composition of the intervening matrix can have a significant influence on these movements (e.g. Ricketts, 2001; Haynes and Cronin, 2003, 2006). In estuarine landscapes, the use of critical intertidal nursery areas such as saltmarshes and mangroves is dependent on the capacity of nekton to move from subtidal refuges into these habitats during high tide (Rozas and Odum, 1987; Kneib and Wagner, 1994; Irlandi and Crawford, 1997; Vance et al., 1996, 2002). In doing so, nekton must move through the intervening matrix, which is often bare sediment or sparse and patchy seagrass (Skilleter et al., 2005; Zharikov et al. 2005) that provides less protection from predators (Coen et al., 1981; Heck and Thoman, 1981; Stoner, 1982; Summerson and Peterson, 1984; Skilleter, 1994) than the habitat to which the animals are travelling. Thus, the intervening matrix is more than just a conduit (*sensu* Fraser et al., 1999) connecting the intertidal and subtidal habitats: it comprises habitats that provide variable but potentially important resources to the animals traversing it. Moreover, the composition and configuration of these estuarine landscapes, including the extent of the intervening matrix, changes with the rise and fall of the tide, in an analogous way (albeit at different temporal scales) to the influence of floods in riverine landscapes (Robinson et al., 2002). Recognition of the dynamic nature of riverine landscapes, including the changed connectivity between aquatic and terrestrial habitats during floods, has led to a reassessment of the factors controlling biodiversity in river systems (Ward and Tockner, 2001; Poole, 2002) and has now been incorporated into areas such as ecological risk assessment for river systems (Leuven and Poudevigne, 2002).

Here, we specifically test the effects of how the composition of the corridor influences the use of subtropical, intertidal mangrove forests by nekton moving from subtidal habitats during high tide. Connectivity between the forests will potentially influence the value of mangroves in supporting biodiversity and fisheries production (Beck et al., 2001; Manson et al., 2005b). In Section 5 we demonstrated that the proximity between mangroves and patches of seagrass influenced the use of the seagrass by prawns and fish and proposed seagrass beds close to mangroves allowed for greater access by nekton to the resources provided by the mangroves. Changes to the composition of the matrix connecting subtidal refuges with intertidal mangroves also have the potential to influence the level of access to the forest, so we compared mosaics that comprised mangroves and subtidal seagrass beds with similar levels of structural complexity but varied in the composition of the intervening intertidal habitat.

6.2. Methods

6.2.1. Experimental Design

Sampling was done during in spring (27 August –12 September) 2003 to examine the effects of spatial arrangement of the downshore habitats on the patterns of utilisation by nekton of mangrove-based mosaics in western Moreton Bay. Three different types of mosaics were compared, all based around relatively high quality mangrove sites. The three mosaics varied primarily in relation to the characteristics of the intertidal corridor that connected the subtidal region with the mangrove forests: (i) mangroves adjacent to dense intertidal seagrass (M+Dense Grass), (ii) mangroves adjacent to patchy intertidal seagrass (M+Patchy Grass) and (iii) mangroves adjacent to bare sediments (M-Grass) (Figure 6.1).

The three mosaics were all within approximately 1.5-2.0 km of each other (Figure 6.1). One of the types of mosaic (mangroves + bare sediments) was separated from the other two by a narrow boating channel running between the intertidal banks at low tide, but the entire area floods at high tide, so the three mosaics are all interconnected by the same water body during this time. Two sites, each separated by approximately 200-250 metres, were sampled within each of the mosaics. The mangroves were sampled with fyke nets set just prior to high tide and allowed to fish for between 2-3 hours, depending on the height on the tide. The total soak time for each net was recorded. Five replicate fyke nets were deployed on each occasion at each site x height combination. All sampling was done during the period of spring tides at the September equinox, a period when the mangroves are inundated completely, twice in a 24-hour period.

The original design for this study was complicated, involving several spatial and temporal components:

Factor A:	Mosaic Type (comparing three mosaics)
Factor B:	Distance into mangroves (1-2 m and 15-20 m into the forest)
Factor C:	Sites (nested within mosaic – spatial variation within any mosaic)
Factor D:	Diel Comparisons (day versus night)

The factors of primary interest were the comparison of the three mosaics and the distance into the mangroves. The factor Sites was included to avoid confounding in the comparisons among the three mosaics. The comparison of catch between night and day was included in order to determine whether a different suite of species, or size range of individuals, were accessing the mangroves at different stages.

6.2.2. Statistical Analyses

A high pressure system over the region during the sampling period (27 August –12 September, 2003) resulted in lower than predicted high tides and the mangroves failed to flood on several occasions. One site within the Mangroves + Patchy Seagrass (M+PS) mosaic did not flood during the night, due to the high pressure system positioned over the region during the period sampling was being done. On some occasions, not all replicate fykes deployed at some sites could be retrieved, usually the presence of large sharks, rays and crabs (the mud crab, *Scylla serrata*) caused the bottom of the wings on the fyke net to lift and it was not clear whether the net had fished for a predictable period of time. Data from these fykes were therefore discarded.

As a consequence of this, different numbers of samples were available for different treatments, so it was not possible to analyse the abundance of individuals using the complete statistical model defined by the experimental design. Furthermore, initial analysis of the data (see Section 6.3.1.2) indicated that the assemblage of fish moving into the mangroves on high tide was significantly different at night compared with day. To determine whether the numbers of species and individuals and the abundance of individual species moving into the mangrove forest varied among the three mosaics and at different distances into the forest, two separate sets of univariate analyses, for day and night samples, were therefore done.

6.2.2.1. Multivariate analyses - Diel Comparisons of Community Composition

Qualitative examination of the data suggested that a different suite of species was using the mangroves at night than in the day. Differences in the composition of the nekton assemblage moving into the mangrove forest during the day and at night were therefore examined separately for each of the three types of mosaic using non-metric multidimensional scaling on untransformed data, using the Bray-Curtis similarity measure. The statistical significance of any differences in the composition of the fauna between day and night was tested with analysis of similarities (ANOSIM). Taxa contributing greater than 5 % to the total diel separation of the assemblages were identified with SIMPER (Clarke, 1993).

6.2.2.2. Univariate Analyses – Species Richness and Numbers of Individuals

Day time: Differences in species richness, the total number of individuals and the abundance of common species using the three different mosaics during the day were analysed with three-factor, mixed model analyses of variance. The factors were (i) Mosaic Type (a=3, fixed), (ii) Site (b=2, random, nested within Mosaic Type) and (iii) Distance into the forest (c=2, fixed). Homoscedasticity of variances was checked using Cochran's test and data were transformed to loge(x+1) if necessary. Where this transformation did not remove the heteroscedasticity of variances, untransformed data were analysed but a more conservative P-value was used to determine significance. Post-hoc pooling of terms in the analyses was done where possible, following the convention (Winer et al. 1991) that appropriate terms could be pooled if they were not significant at P > 0.25).

Night time: As noted above, one site within the Mangroves + Patchy Seagrass (M+PS) mosaic did not flood during the night, due to the high pressure system positioned over the region during the period sampling was being done, leading to an unbalanced design. Two separate sets of analyses were done in order to examine whether species richness and abundance varied among the types of mosaic. First, data comparing the utilisation of the Mangroves + Dense Seagrass (M+DS) and Mangroves – Seagrass (M-S) mosaics were analysed with three factor, mixed model analyses of variance, with factors Quality (fixed, 2 levels), Site nested within Quality (random, 2 levels) and Distance (fixed, 2 levels). The primary aim of these analyses was to determine if there was significant variation in the abundance of any of the species at the scale of sites. In all cases, there was no evidence of significant site to site variation, at the conservative level of $P > 0.25$, so the assumption was therefore made that this terms could be removed from the statistical model (Underwood, 1981; Winer et al., 1991). There is an implicit assumption then made that site to site variation would also have not been statistically important for the Mangroves + Patchy Seagrass mosaic.

Second, data comparing the utilisation of all three mosaics (i.e. M+DS, M+PS and M-S) were then analysed with two factor analyses of variance, with factors Quality (fixed, 3 levels) and Distance (fixed, 2 levels), with the data pooled across the two sites in each of the mosaics where such data were available. Only the results from this second set of analyses is presented, as they represent the specific test of hypotheses about the influence of mosaic type and distance into forest on the abundance of individual species and on species richness.

6.3. Results

6.3.1. Qualitative Results

Overall, 128 fyke samples were collected successfully. A total of 44,121 animals was processed from these samples, comprising 52 different species, although the number of species varied considerably depending on whether samples were collected during the day or night and the distance into the forest the nets were set (Table 6.1). The greatest number of species and individuals was collected from the Mangroves + Dense Grass mosaic sampled during the night.

6.3.2. Diel Comparisons of Community Composition

The composition of the assemblage of nekton using the mangrove forests varied significantly on a diel basis (Table 6.2A; Figure 6.2) for each of the three mosaic types, although the difference was most pronounced for the Mangroves + Patchy Grass mosaic (Figure 6.2B). Between four and six species contributed over 75 % to the diel separation of the assemblages in each mosaic type (Table 6.2B). The common toadfish (*Tetractenos hamiltoni*) and the flat-tailed mullet (*Liza argentea*) were the only two species that contributed to the diel separation of the assemblages in each of the three mosaics, with the toadfish consistently more abundant during the day and the mullet more abundant at night. On the basis of these highly significant differences in the diel patterns of utilisation of the mangroves, all subsequent analyses comparing the three mosaic types were done for diurnal and nocturnal data separately.

6.3.3. Comparisons of Community Composition Among Mosaics

6.3.3.1. Day time

The composition of the nekton assemblage utilising the mangroves during the day varied significantly among the three types of mosaic separated (ANOSIM, $P < 0.001$). The mosaic without any seagrass in the downshore corridor (M-S) was noticeably different from the other two mosaics (M+DS and M+PS), which tended to show some overlap in composition, although these two were still significantly different from each other (Figure 6.3A). There were also significant differences in the composition of the nekton assemblage between the two distances into the forest (1-2 m versus 15-20 m) in each of the mosaics (Figure 6.3B).

The different mosaics could be distinguished from each other based on the abundance of 4-5 species of fish, *Valamugil georgii* (silver mullet), *Liza argentea* (flat-tailed mullet), *Pseudomugil signifer* (Pacific blue-eye), *Tetractenos hamiltoni* (common toadfish), *Acanthopagrus australis* (yellowfin bream), *Marilyna pleurosticta* (common pufferfish) and a species of shrimp, *Palaemon debilis* (Table 6.3) which accounted for ~ 70-75 % of the cumulative percentage contributions to the separation of the assemblages.

These same species were also primarily responsible for distinguishing between the assemblages of the fish using the mangroves at different distances into the forest, in each of the three mosaics (Table 6.4). There were usually very marked differences in the abundance between the two heights for each of these species (Table 6.4), but this will be examined in more detail below.

6.3.3.2. Night time

The composition of the nekton assemblage utilising the mangroves at night also varied significantly among the three types of mosaic (ANOSIM, $P < 0.001$), although the separation of the three mosaics was more pronounced (Figure 6.4A) than it was during the day (cf. Figure 6.3A). Again, there were also significant differences in the composition of the nekton assemblage between the two distances into the forest in each of the mosaics (Figure 6.4B).

Essentially the same group of fish species at night as during the day were responsible for the separation of the nekton assemblages among the three mosaics (Table 6.5), but the prawns *Metapenaeus bennettae* and *M. ensis* now also contributed to the distinction among the mosaics. The distinction between the nekton assemblages moving different distances into the forest at night was, however, also a function of additional species being present at one distance and not the other (*Sillago analis* – sand whiting; *Atherinomorus ogilbyi* - Ogilby's hardyhead; *Ambassis marianus* – estuary perchlet) (Table 6.6).

6.3.4. Comparisons of Abundance Among Mosaics

6.3.4.1. Day time

The number of species using the mangroves during the day was similar in each of the three mosaics, at the two heights and in both sites. None of the interactions between main effects was significant (ANOVA, Table 6.7). The number of species averaged between 4 and 6 species in each of the mosaics, except in the upper mangrove in the Mangroves–Seagrass mosaic where on average 7 species were caught (Figure 6.5A).

The number of individuals using the mangroves did not vary significantly between the two distances into the forest or the two sites within each mosaic, but there were significant differences among the three mosaic types (Table 6.7). Twice as many fish made use of each of the M+Dense Seagrass and M-Seagrass mosaics compared with the M+Patchy Seagrass mosaics (Figure 6.5B).

The number of yellowfin bream (*Acanthopagrus australis*) moving into the mangroves during the day varied among the three mosaics, but only in the lower forest (significant Mosaic x Distance interaction, P < 0.001; Table 6.7). Significantly fewer yellowfin bream crossed the dense seagrass corridor than the patchy seagrass or bare sediment corridors to reach the lower forest (Figure 6.5C). Similar numbers of yellowfin bream moved into the upper forest in each of the mosaics, approximately the same numbers that were moving into the lower forest of the Mangroves+Dense seagrass mosaic. There was no significant variation between the two sites.

The number of estuary perchlet (*Ambassis marianus*) moving into the mangroves during the day was generally small, with most of the individuals using the Mangroves–Seagrass mosaic (Figure 6.5D). Significantly more estuary perchlets crossed the corridor without seagrass than either of the other mosaics into both the upper and lower forest, with almost three times as many staying in the lower forest than moving into the upper forest (significant Mosaic x Distance interaction, P < 0.001; Table 6.7).

Three species of gobies, *Arenigobius frenatus*, *Favonigobius exquisitus* and *Mugilogobius stigmaticus*, were abundant in the mangroves. Many of the individuals of *Arenigobius frenatus* and *Favonigobius exquisitus* were small juveniles and it was difficult to separate them confidently and reliably so the catch for these two species was combined and analysed together. There was considerable variation in the abundance of these gobies caught in the mangroves with significant variation between the sites, although the magnitude of this variation also depended on the distance into the forest (significant Distance x Site(Mosaic) interaction, P < 0.001; Table 6.7). There was a trend towards more gobies in the two mosaics having a seagrass corridor downshore (Figure 6.5E), but there were no significant effects of Mosaic Type as either a main effect or interaction. There was also a trend towards more individuals in the upper forest, but again this was not significant and the small-scale variation (between sites) prevented any post-hoc pooling to increase the power (through greater degrees of freedom) of tests of main effects.

Flat-tailed mullet (*Liza argentea*) moved into the upper forest in large numbers, but only in the Mangrove-Seagrass mosaic (Figure 6.5F) and there were few individuals caught in the lower forest in any of the mosaics. There was a significant main effect of Mosaic Type detected (Table 6.7), although this was dominated by the very large numbers of individuals caught in the upper forest of Mangrove-Seagrass mosaic. The Mosaic x Distance interaction was, however, not significant (ANOVA, P < 0.07; Table 6.7) due to the significant small-scale variation at the site level (ANOVA; Distance x Site(Mosaic) interaction, P < 0.02, Table 6.7).

The common pufferfish (*Marilyna pleurosticta*) was significantly more abundant in the Mangrove-Seagrass mosaic (ANOVA, Mosaic Type, $P < 0.01$, Table 6.7), with only small numbers caught in either of the other two mosaics (Figure 6.5G). There was also significant small scale variation, between sites, in the abundance of the pufferfish (Table 6.7). The number of pufferfish did not differ significantly between the two distance into the forest.

The third species of goby caught in the mangroves, *Mugilogobius stigmaticus*, was significantly more abundant in the upper forest than the lower (Figure 6.5H; ANOVA, Height – $P < 0.02$, Table 6.7), but there were no significant differences among the three mosaic types, either as a main effect or as an interaction (Table 6.7). There was some variation between the two sites within each mosaic, although this was not significant ($P < 0.08$).

The common toadfish, *Tetractenos hamiltoni*, was very abundant in all mosaics and at both distances into the forest during the day. More toadfish moved into the upper forest in the mosaics with a seagrass corridor downshore (M+DS, M+PS) than the mosaic with a bare sediment corridor (Figure 6.5I), but there were no significant differences among the three mosaics in the lower forest (ANOVA, significant Mosaic x Distance interaction, $P < 0.05$, Table 6.7).

Silver mullet (*Valamugil georgi*) were also very abundant in the mangrove forest during the day, but their pattern of utilisation was complicated and differences among the three mosaics depended on the distance into the forest (ANOVA; significant Mosaic x Distance interaction, $P < 0.05$, Table 6.7). In the upper forest, there were significantly more silver mullet caught in the mosaic without low-shore seagrass (M-D) than the other mosaics (M+DS, M+PS). In contrast, in the lower forest, more silver mullet moved into the mangroves for the mosaic with dense seagrass (M+DS) and the mosaic without seagrass (M-S) compared with the mosaic with patchy seagrass (M+PS) (Figure 6.5J).

There were only two species of decapod crustaceans that occurred in reasonable numbers in the mosaics during the day, the prawn *Metapenaeus bennettae* and the carid shrimp *Palaemon debilis*. A second species of prawn, *M. ensis* occurred in small numbers in each of the mosaics but was not analysed because of the large number of samples that did not contain individuals of this species.

The number of shrimp, *Palaemon debilis*, moving into the mangroves varied between the upper and lower forest (ANOVA; significant Distance effect, $P < 0.04$), although the difference only amounted to ca. 10 % more shrimp in the lower forest than the upper. There was, however, also a significant difference among the three mosaics (ANOVA; significant Mosaic effect, $P < 0.05$), with considerably more shrimp moving into the forest through the dense seagrass (M+DS) than the other two mosaics (Figure 6.5K).

The pattern of usage of the mangrove forest was quite different for *Metapenaeus bennettae*. Very few prawns moved into the upper forest (Figure 6.7L) through any of the mosaics. Significantly more *M. bennettae* moved into the lower forest through the areas without low-shore seagrass (M-S) than the other two mosaics (ANOVA; significant Mosaic x Distance interaction, $P < 0.05$, Table 6.7). There was no significant small scale variation (between sites within each mosaic).

6.3.4.2. Night time

Generally, a greater number of species moved into the forest in each of the mosaics at night ($\overline{X_T}$ = 7.4 species ± 0.3 SE; Figure 6.6A) than during the day ($\overline{X_T}$ = 5.6 species ± 0.3 SE; Figure 6.5A). During the night, more species moved into the upper forest than were caught in the lower forest in each of the mosaic types, but the magnitude of this difference varied (ANOVA, significant Mosaic x Distance interaction, $P < 0.001$; Table 6.8). Significantly fewer species were caught in the lower forest for the Mangroves-Seagrass mosaic than the other two mosaics, but there was no significant difference in the number of species using the upper forest among the three mosaics (Figure 6.6A).

A greater number of individuals also moved into the upper forest ($\overline{X_T}$ = 304.3 ± 67 SE) than were caught in the lower forest ($\overline{X_T}$ = 164.6 ± 26 SE) (ANOVA, significant Distance effect, $P < 0.03$; Table 6.8). The number of individuals using the forest varied among each of the mosaics (Figure 6.6B), with the largest number of individuals used the forest in the Mangroves + Dense Seagrass mosaic.

All but a single individual of the estuary perchlet, *Ambassis marianus*, were caught in the forest associated with the Mangroves – Seagrass mosaic (Figure 6.6C; significant Mosaic effect, $P < 0.001$, Table 6.8). There was no significant difference in the number of *A. marianus* caught in the upper and lower forest, although there was ca. 30 % more caught in the upper than lower forest, the opposite pattern to that observed during the day.

More gobies, primarily juveniles of *Arenogobius frenatus* and *Favonigobius exquisitus*, were caught in the upper ($\overline{X_T}$ = 6.3 ± 1.5 SE) than lower ($\overline{X_T}$ = 1.7 ± 0.4 SE) forest (ANOVA, significant Distance effect, $P < 0.002$). Significantly more gobies were caught in the mangroves associated with dense seagrass downshore than either of the other two mosaics (Table 6.8; Figure 6.6D).

Generally, more flat-tailed mullet, *Liza argentea*, were caught in the upper than the lower forest (Figure 6.6E), although the extent of this difference varied among the three mosaics (ANOVA, significant Mosaic x Distance interaction, $P < 0.02$; Table 6.8). Significantly more flat-tailed mullet were caught in the forest associated with the dense seagrass mosaic (M+DS) in both the upper and lower forest. There were also more *L. argentea* caught in the mosaic without low shore seagrass (M-S) than in the mosaic with patchy seagrass (M+PS), but only in the upper forest (Figure 6.6E).

Almost all the individuals of the common pufferfish, *Marilyna pleurosticta*, caught in the mangrove forest at night were using the mosaic without downshore seagrass (M-S) (Figure 6.6F). There was no significant difference in the abundance of the pufferfish using the upper or lower forest (Table 6.8).

There was a strong effect of mosaic type on the number of mangrove gobies, *Mugilogobius stigmaticus*, using the mangrove forest. Significantly more mangrove gobies (65 % of individuals) moved across the Mangroves + Patchy Seagrass mosaic than either the Mangroves + Dense Seagrass (14%) or the Mangroves – Seagrass (21%) (Figure 6.6G). Significantly more mangrove gobies, *Mugilogobius stigmaticus*, were caught in the upper ($\overline{X_T}$ = 13.3 ± 4 SE) than lower forest ($\overline{X_T}$ = 3.4

± 1 SE) (ANOVA, Distance effect, $P < 0.001$; Table 6.8), consistent with the pattern seen during the day.

Differences in the abundance of the common toadfish, *Tetractenos hamiltoni*, among the mosaics at night varied with the distance into the forest (ANOVA, significant Mosaic x Distance interaction, $P < 0.05$; Table 6.8). In the lower forest, there was slightly more toadfish found in the mosaic without seagrass (M-S) but differences among the mosaics were not significant (Figure 6.6H). In the upper forest, however, there were significantly more toadfish using the mosaic without seagrass compared with the others. There were also more toadfish in the mosaic with dense seagrass than in the mosaic with patchy seagrass (Figure 6.6H).

In contrast to the complicated pattern seen in the day, utilisation of the mangrove forest by silver mullet (*Valamugil georgii*) at night was more straightforward. Significantly more silver mullet were caught in the upper forest ($\overline{X_T} = 115.3 \pm 42$ SE) than lower forest ($\overline{X_T} = 49.0 \pm 14$ SE) (ANOVA, Distance effect $P < 0.05$; Table 6.8). Silver mullet were more abundance in the mosaic with dense seagrass downshore (M+DS), followed by the mosaic without seagrass (M-S), with the smallest number caught from the mosaic with patchy seagrass (M+PS) (Figure 6.6I).

Very large numbers of *Palaemon debilis* were caught in the mangroves at night and the pattern among the three mosaic types was the same as seen during the day. Significantly more shrimp were caught in the mosaic with dense seagrass (M+DS, ca. 85 %) than in the other two mosaics (M+PS, ca. 10 %; M-S, ca. 5 %) (Figure 6.6J). There was no significant difference in the abundance of the shrimp between the upper and lower forest (ANOVA, Table 6.8).

More *Metapenaeus bennettae* were caught in the upper and lower forest in the mosaic without downshore seagrass (M-S) than in the other mosaics (Figure 6.6K), followed by the mosaic with dense seagrass (M+DS). Very few greasyback prawns were caught in the upper or lower forest in the mosaic with patchy seagrass (M+PS) (Figure 6.6K). A significant interaction between mosaic and distance (ANOVA, $P < 0.01$) resulted from significantly more greasyback prawns using the lower forest than the upper in the mosaic without downshore seagrass (M-S), consistent with the pattern observed during the day (cf. Figure 6.5L with Figure 6.6K).

In contrast to the situation in the day when few *Metapenaeus ensis* were caught in the mangroves, reasonable numbers of the prawns moved into the forest at night. In the upper forest, more *M. ensis* were caught in the two mosaics with downshore seagrass (M+DS, M+PS) than in the mosaic without downshore seagrass (M-S) (Figure 6.6L). In the lower forest, however, more prawns were caught in the mosaic with patchy seagrass (M+PS) than in either of the other mosaics (ANOVA, significant Mosaic x Distance interaction, $P < 0.001$; Table 6.8; Figure 6.6L).

6.4. Discussion

The nature of the corridor connecting subtidal seagrasses with intertidal mangroves had marked effects on the numbers and types of fish and invertebrates that used the mangroves. The composition of the assemblage using the different mosaics varied during the day and during the night, but also varied at different distances into the forest. During the day, almost twice as many individuals moved into the mangroves when the corridor had dense seagrass or was bare compared with when it was a patchy seagrass habitat. At night the pattern was similar, except the number of individuals moving across the unvegetated corridor was intermediate between the dense seagrass and patchy seagrass corridors. Some species were clearly more strongly associated with the mosaic containing the unvegetated corridor (e.g the estuary perchlet, *Ambassis marianus*) whereas other species were associated with vegetated corridors (e.g. yellowfin bream, *Acanthopagrus australis*, gobies and the estuarine shrimp, *Palaemon debilis*). These results have important implications for understanding the linkages between nekton and the critical estuarine habitats that support them. Nekton do not utilise habitats such as mangroves independently of the surrounding landscape and, as such, conservation and management of nursery areas will not be effective unless this is taken into account (see also Beck et al., 2001)

The transition from one habitat to another often requires animals to move through intervening habitats that may offer increased risk and so abundances are often smaller in these connecting habitats (Brown, 1984). Fish moving to the mangroves from subtidal habitats must cross intertidal areas that have variable cover, ranging from dense seagrass, rubble patches to unvegetated mudflats (Zharikov et al., 2005). Dense seagrass generally provides a greater refuge from predation for small fish and invertebrates than less structured environments (e.g. Coen et al., 1981; Heck and Thoman, 1981; Summerson and Peterson, 1984; Skilleter, 1994) so we had anticipated that mosaics containing a corridor of dense seagrass would be utilised more than one containing patchy seagrass or unvegetated mudflat. This was not always the case though.

During the day, juvenile yellowfin bream, *Acanthopagrus australis*, were more abundant in the mosaics with a less structured corridor downshore than those with a more heavily vegetated corridor. This result seems to be inconsistent with the general patterns of specific habitat use by yellowfin bream. *Acanthopagrus australis* appears to have a preference for structured habitats: they are more abundant in dense seagrass (Section 5) and were never caught over unvegetated mudflats (Section 1). Furthermore, yellowfin bream were significantly more abundant in dense seagrass beds close to mangroves than those further away (Section 5). This association with structure suggests that there are advantages to *Acanthopagrus australis* utilising mangroves during the high tide, either to avoid predators (Sheaves, 2005) or to take advantage of increased abundances of macroinvertebrate prey in mangroves compared with nearby habitats (Bell et al., 1984; Laegdsgaard and Johnson, 1995; Roach, 1998) but these benefits must be traded against a heightened risk of predation on the mudflats in order to reach the structural complexity of the mangrove forest. Our results indicate that the abundance of *Acanthopagrus australis* in mangroves at any time is a function of the entire mosaic. In a mangrove mosaic with a vegetated corridor, the yellowfin bream make use of both the forest and the intertidal habitat, foraging and sheltering in both areas: the total number of fish using the mosaic is therefore spread over both habitats. In a mangrove mosaic with an unvegetated corridor, the benefits of accessing the mangroves result in fish moving there on the rising tide, but they do not also make use of the unvegetated or sparsely vegetated intertidal areas below the mangroves: the total number of fish in the mosaic is therefore now only found in the mangrove habitat. Detailed gut contents analysis of fish caught in the different mosaics would provide a partial answer to this hypothesis. The species composition of potential prey in the mangroves and seagrass beds are very different (Skilleter, unpublished data), so bream using the different mosaics (mangroves with and without adjacent seagrass) are likely to be consuming a different range of species.

Another important predator in the system, the common toadfish, *Tetractenos hamiltoni*, was more abundant in the upper than lower forest during night and day. Toadfish follow the rising tide up the shore, often positioned right at the water's edge and they appear to feed continuously (personal observation). The greater abundance in the upper than lower forest may simply reflect a concentration of animals feeding along this frontal region, where prey may be more abundant or more easily captured. Increased predation at frontal regions is common in oceanic systems (e.g. Bakun, 2006). The numbers of fish using the upper forest was, however, influenced by the nature of the downshore corridor through which they moved. During the day, more toadfish accessed the upper forest via the vegetated corridor, but at night the opposite was true with larger numbers accessing the upper forest via the unvegetated corridor. Numbers using the lower forest were similar in all three mosaics. It is tempting to suggest that the toadfish use the vegetated corridors during the day, but the unvegetated corridors at night, as a response to a heightened risk of predation during the day (e.g. James and Heck, 1994), but this seems unlikely given the toadfish exhibit the common characteristics of their family: they are poisonous and able to swell up when threatened. They do not appear to have any obvious predators, at least in the shallow water habitats examined here. An explanation based on variable risk of predation would also be unlikely to account for the lack of any significant difference in the numbers of toadfish using the lower forest in the different mosaics.

Tetractenos hamiltoni may be attracted to pneumatophores covered with algae because of the enhanced abundance of food found there (Laegdsgaard and Johnson, 2001), but we have no data on whether there are differences in the abundance of food at different distances into the mangrove forest and whether this varies among mosaics (i.e. a distance x mosaic interaction in food abundance). Again, this would not account for the reversal of the pattern at night in the upper forest. There are few data available in the literature on this or related species to throw any light on these patterns. A related species, *Tetractenos glaber*, was found in temperate saltmarsh (Victoria, Australia) only during the day, but numbers were generally quite small and few samples were collected at night (Crinall and Hindell, 2004). *T. glaber* is also common at the edge of nearby mangroves (Hindell and Jenkins, 2004, 2005), but no samples were taken at night in that study nor was sampling done within the forest.

For other species of fish, there were complicated diel shifts in the use of the different mosaics. During the day-time, juvenile mullets, *Liza argentea* and *Valamugil stigmaticus*, were more abundant in the upper mangrove forest in mosaics without downshore seagrass, but at night shifted to using the mosaics with dense seagrass downshore. There is no obvious explanation for this diel shift in use from bare to densely vegetated corridors. The substantial shifts in use of the different mosaics between night and day suggest the mullet are obtaining different resources at different times. Mullets are detritivores (Blaber, 1977; Blay, 1995), ingesting the sediment to extract organic material and these species were rarely caught in seagrass beds (Section 1), so presumably move in schools directly to the mangroves on the rising tide in order to feed on the rich supply of detritus within the forest (Hogarth, 1999). During the day, significantly more individuals of *Liza argentea* were caught in the upper than lower forest suggesting juveniles may further into the forest to take advantage of shallower water (McIvor and Odum, 1988; Paterson and Whitfield, 2000; Vance et al., 2006) to avoid the many piscivores that specialise on small individuals (Humphries et al., 1992; Sheaves, 2005). Juvenile *L. argentea* are strongly associated with mangroves, apparently in response to risk of predation (Laegdsgaard and Johnson, 2001). If predation is an important factor in determining the use of corridors by mullet, the expectation would have been that use of the dense seagrass corridor would be greater during the day than night, when visually-searching piscivores are more active. We found the opposite pattern though. Furthermore, juvenile *Valamugil stigmaticus* were also abundant in the lower forest during the day, although they made the same shift to using the densely vegetated corridors at night. Either risk of predation is not a proximal driver for the patterns we observed, there are other predators influencing the abundance of juvenile mullets and these predators are more active at night than day or mullets are using the mangroves for reasons other than avoidance of predators.

The carid shrimp, *Palaemon debilis*, was almost exclusively associated with the mosaic containing the dense seagrass corridor and this is most likely explained by a reduction in the rates of predation associated with increased structural complexity (references above). Shrimp are preyed on by a number of species of fish that utilise the mangrove and seagrass mosaics in Moreton Bay, including *Centropogon australis* (Bell et al., 1978), *Acanthopagrus australis* (Blaber and Blaber, 1980; Moreton et al., 1987), *Monocanthus chinensis* (Conacher et al., 1979) and adults of *Sillago* spp. (Gunn and Milward, 1985; Brewer and Warburton, 1992), so this pattern of use of different mosaics is not surprising. In contrast though, small *Metapenaeus bennettae*, also subject to considerable predation from these same species of fish, were more abundant in the mangroves connected to the subtidal by the unvegetated corridor. This pattern was pronounced during the night and the day, suggesting some process other than avoidance of visually searching predators is involved. *Metapenaeus ensis*, which only occurred in the mangroves at night, was more abundant in the mosaic with the patchy corridor than the one with the dense seagrass corridor. The abundance of the gobies, *Arenigobius frenatus* and *Favonigobius exquisitus*, was also greatest in the mosaic with the dense seagrass corridor, with the pattern more pronounced during the night than the day. In contrast, the abundance of the other common species of goby, *Mugilogobius stigmaticus*, was greatest in the mosaic with the patchy seagrass mosaic at night, with no differences among mosaics during the day. Taken together, these results suggest that predation along may not be a sufficient explanation to account for the patterns of habitat utilisation at landscape scales, given some species show patterns consistent with avoidance of predators while other, similar sized and related species (either taxonomically or morphologically), showed the opposite pattern. Experimental work to determine the processes causing such complex patterns are clearly needed: sampling has identified the importance of linkages and connectivity among habitats, but cannot identify the reasons for those patterns.

Despite the variability in how nekton used the different corridors, the results clearly indicate that definitions of connectivity in estuarine systems must be based on the basis of the movements and utilisation of different component habitats by different species, as has been reported for terrestrial systems (Simberloff et al., 1992; Wiens et al., 1997). One of the principal benefits of corridors, from a conservation perspective, is to facilitate movement between habitats, providing animals with access to a wider range of resources and refuges from disturbances and predators (Saunders et al., 1991), but most of the evidence for the values of corridors has come from terrestrial systems and some of this evidence has been challenged (Simberloff et al., 1992). Studies on (non-migratory) movements of forest birds suggest that they less prone to traverse unstructured habitats than move through areas where patches are connected by structure (Belisle et al., 2001). In aquatic habitats, ontogenetic shifts in habitat use have been found to be affected by the nature of the corridors between habitats used by different life history stages: seagrass corridors enhanced dispersal of juvenile lobsters to adult habitats (Acosta, 1999). Corridors between transitional habitats such as mangroves (and other intertidal habitats such as saltmarshes) and subtidal refuges such as seagrass beds may be more important, however, as habitat linkages rather than movement corridors (*sensu* Rosenberg et al., 1997). Mangroves are only available to nekton for a limited period of time, during high tides, and animals must generally return to the same area from which they emerged, rather than passing through mangroves on the way to some other type of habitat. Access to the mangroves requires passage across the intertidal corridor downshore from the forest, an important consideration in evaluating the relative importance of different corridors (Simberloff et al., 1992; Rosenberg et al, 1997) but, more importantly, indicating that any assessment of the conservation value of mangroves should not be done in isolation of the composition of the surrounding landscape.

Seagrasses are subjected to a large number of natural and anthropogenic perturbations (Shepherd et al. 1989), including disturbances such as wave action, recreational boating, trampling and storms and bait-harvesting (Zieman, 1976; Dawes et al., 1997; Creed & Amado Filho, 1999; Eckrich & Holmquist, 2000; Skilleter et al., 2006). There has been an increased focus in recent years on understanding the implications of seagrass loss because of the worldwide decline in the distribution and health of seagrass beds (e.g. Shepherd et al. , 1989; Short & Wyllie-Echeverria, 1996). Much of this work though has focussed on impacts within a seagrass system (e.g. Hovel and Lipcius, 2001; Hovel, 2003; Stockhausen and Lipcius, 2003; Tanner, 2003, 2005), rather than on indirect effects on surrounding habitats. The consequences of degradation or loss of seagrass on the value of surrounding habitats to groups such as nekton are less well understood (but see Irlandi and Crawford, 1997; Eggleston et al., 1998, 1999; Nagelkerken et al., 2000). Our results indicate that connectivity between habitats is a critical aspect of determining the value of individual components of a mosaic to mobile species such as fish. Protection directed at specific habitats, such as mangroves, may be ineffective as a means of maintaining biodiversity and ecosystem function, unless surrounding habitats are also protected (see also Acosta, 1999).

Table 6.1: Summary of the numbers of fyke nets successfully sampled and the numbers of individuals and species caught in fyke nets deployed at each of two distances into the mangrove forest from the lower edge (pneumatophore zone) in three different types of mosaic, all based around dense mangrove areas. Mangroves + Dense Grass is defined as a mosaic with extensive, continuous seagrass beds in the intertidal below the mangrove forest. Mangroves + Patchy Grass is defined as a mosaic with extensive but patchy seagrass beds in the intertidal. Mangroves - Grass is defined as a mosaic with bare sediments, or the occasional patch of seagrass, in the intertidal below the mangroves. Values are the total catches for 5 fyke nets deployed at each height in each of two sites per mosaic type, during the day or night time high tide (see text for other details).

Mosaic Type	Night vs. Day	Distance into Forest	Number of Fykes	Number of Individuals	Species Richness
Mangroves +	Day	Upper (D1)	16	1412	11
Dense Grass		Lower (D2)	9	3604	24
	Night	Upper (D1)	10	10706	34
		Lower (D2)	10	6413	33
Mangroves +	Day	Upper (D1)	10	1230	22
Patchy Grass		Lower (D2)	10	1235	28
	Night	Upper (D1)	4	361	13
		Lower (D2)	5	606	20
Mangroves -	Day	Upper (D1)	19	2774	20
Grass		Lower (D2)	15	7510	31
	Night	Upper (D1)	10	3621	30
		Lower (D2)	10	4231	25
Total			128	44121	52

Table 6.2: Results of (A) multivariate analysis of similarities (ANOSIM) and (B) percentage contributions to differences in composition (SIMPER) for the nekton assemblage using three different types of mangrove mosaic during the day and at night in Spring 2003. Data are the number of animals, standardised for a 2 hour soak time, caught in fyke nets set two different distances into the forest. (Nominally n=5 replicate fyke nets deployed at each of two sites within each mosaic, although some sites did not flood on all occasions. See text for further details).

(A) ANOSIM results

	Global R	**P**
M + Dense Grass	0.376	< 0.001
M + Patchy Grass	0.184	< 0.009
M – Grass	0.225	< 0.001

(B) Simper results. The average abundance for each species during the day and night is shown, plus the % contribution of each species to the diel separation of the nekton assemblages. Only those species contributing to > 5 % to the separation of the communities at day and night are shown.

M + Dense Grass	**DAY**	**NIGHT**	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Valamugil georgii	37.8	145.6	34.7
Palaemon debilis	47.2	120.05	29.3
Tetractenos hamiltoni	38.3	9.3	11.4
Liza argentea	4.8	46.7	7.4
M + Patchy Grass	**DAY**	**NIGHT**	
	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Tetractenos hamiltoni	44.2	5.7	31.3
Mugilgobius stigmaticus	6.5	20.6	14.8
Palaemon debilis	2.2	14.1	12.2
Acanthopagrus australis	7.6	2.0	5.9
Sillago analis	1.1	7.7	5.7
Liza argentea	0.4	6.3	5.1
M - Grass	**DAY**	**NIGHT**	
	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Valamugil georgii	44.1	52.7	25.3
Liza argentea	24.1	17.1	12.0
Tetractenos hamiltoni	32.5	17.6	11.3
Ambassis marianus	6.7	21.4	9.2
Metapenaeus bennettae	2.1	18.3	8.8
Metapenaeus ensis	2.1	15.2	7.3

Table 6.3: Results of SIMPER (Similarity Percentages) analysis showing the contributions to differences in composition for the nekton assemblage using three different types of mangrove mosaic (M+DS = Mangroves plus adjacent Dense Seagrass; M+PS = Mangroves plus adjacent patchy Seagrass; M-S = Mangroves without adjacent Seagrass) during the day in September (Spring) 2003. The average abundance for each species is shown, plus the % contribution of each species to the separation of the nekton assemblages in pairs of mosaics. Only those species contributing to > 5 % to the separation of the communities, up to a total of ~ 70 %, between pairs of mosaics are shown. Data are the number of animals, standardised for a 2 hour soak time, caught in fyke nets set two different distances (1-2 and 15-20 m) into the forest. (Nominally n=5 replicate fyke nets deployed at each of two sites within each mosaic, although some sites did not flood on all occasions. See text for further details).

	M+DS	**M-S**	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Valamugil georgii	37.8	44.12	23.5
Palaemon debilis	47.2	4.9	18.9
Tetractenos hamiltoni	38.3	32.5	17.5
Liza argentea	4.8	24.1	11.1
	M+DS	**M+PS**	
	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Palaemon debilis	47.2	2.2	25.9
Tetractenos hamiltoni	38.3	44.2	22.4
Valamugil georgii	37.8	1.1	17.0
Pseudomugil signifer	10.8	0.4	6.4
Acanthopagrus australis	2.3	7.6	5.2
	M+PS	**M-S**	
	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Tetractenos hamiltoni	44.2	32.5	24.4
Valamugil georgii	1.1	44.1	20.7
Liza argentea	0.4	24.1	15.1
Acanthopagrus australis	7.6	9.3	6.2
Marilyna pleurosticta	0.1	11.1	5.6

Table 6.4: Results of SIMPER (Similarity Percentages) analysis showing the contributions to differences in composition for the nekton assemblage moving two different distances (Lower =1-2; Upper =15-20 m) into the mangrove forest for each of the three different mosaics during the day in September (Spring) 2003. The average abundance for each species is shown, plus the % contribution of each species to the separation of the nekton assemblages in pairs of mosaics. Other details as in Table 6.3.

M+DS	**Upper**	**Lower**	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Palaemon debilis	42.6	50.3	26.7
Valamugil georgii	2.9	61.1	23.3
Tetractenos hamiltoni	54.7	27.3	19.6
Pseudomugil signifer	24.7	1.5	13.1
M+PS	**Upper**	**Lower**	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Tetractenos hamiltoni	54.6	33.8	43.6
Acanthopagrus australis	3.9	11.2	11.5
Mugilogobius stigmaticus	8.3	4.7	10.0
Pelates sexlineatus	1.9	5.9	7.6
Gobies	4.8	0.9	6.0
M-S	**Upper**	**Lower**	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Tetractenos hamiltoni	44.2	32.5	24.4
Valamugil georgii	1.1	44.1	20.7
Liza argentea	0.4	24.1	15.1
Acanthopagrus australis	7.6	9.3	6.2
Marilyna pleurosticta	0.1	11.1	5.6

Table 6.5: Results of SIMPER (Similarity Percentages) analysis showing the contributions to differences in composition for the nekton assemblage using three different types of mangrove mosaic during the night in September (Spring) 2003. The average abundance for each species is shown, plus the % contribution of each species to the separation of the nekton assemblages in pairs of mosaics. Other details as in Table 6.3.

	M+DS	**M-S**	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Palaemon debilis	120.01	6.7	29.9
Valamugil georgii	145.6	52.7	28.5
Ambassis marianus	0.1	21.4	7.2
Liza argentea	46.7	17.1	7.1
Metapenaeus bennettae	4.4	18.3	5.1
	M+DS	**M+PS**	
	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Valamugil georgii	145.6	2.9	34.1
Palaemon debilis	120.1	14.1	29.3
Liza argentea	46.7	6.3	8.2
Mugilogobius stigmaticus	4.4	20.6	7.3
	M+PS	**M-S**	
	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Valamugil georgii	2.9	52.7	20.4
Ambassis marianus	0.0	21.35	10.7
Mugilogobius stigmaticus	20.6	6.6	9.9
Metapenaeus bennettae	0.4	18.3	9.5
Metapenaeus ensis	0.9	15.2	7.5
Liza argentea	17.1	6.3	7.4
Palaemon debilis	14.1	6.7	6.6
Tetractenos hamiltoni	5.7	17.6	6.2

Table 6.6: Results of SIMPER (Similarity Percentages) analysis showing the contributions to differences in composition for the nekton assemblage moving two different distances (Lower =1-2; Upper =15-20 m) into the mangrove forest for each of the three different mosaics during the night in September (Spring) 2003. The average abundance for each species is shown, plus the % contribution of each species to the separation of the nekton assemblages in pairs of mosaics. Other details as in Table 6.3.

M+DS	**Upper**	**Lower**	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Valamugil georgii	199.2	91.9	38.1
Palaemon debilis	140.4	99.7	36.2
Liza argentea	76.4	17.0	8.9
M+PS	**Upper**	**Lower**	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Mugilogobius stigmaticus	34.2	9.6	29.1
Mullet sp.	8.8	1.2	10.6
Palaemon debilis	10.3	17.2	9.4
Sillago analis	11.0	5.0	8.7
Liza argentea	3.8	8.4	8.5
Gobies	5.8	0.4	7.0
Atherinomorus ogilbyi	3.8	5.4	6.1
M-S	**Upper**	**Lower**	
Species	$\overline{X}$ Abundance	$\overline{X}$ Abundance	% Contribution
Valamugil georgii	76.0	29.4	28.4
Liza argentea	28.9	5.2	11.6
Metapenaeus bennettae	10.6	25.9	10.0
Metapenaeus ensis	7.6	22.8	5.3
Ambassis marianus	25.3	17.4	4.7
Mugilogobius stigmaticus	11.4	1.8	6.4
Tetractenos hamiltoni	22.6	12.6	5.8

Table 6.7: Summary analyses of variance comparing species richness, the total number of individuals and the abundance of common species using mangroves during the day in each of three different mosaic types (M+DS = Mangroves plus adjacent Dense Seagrass; M+PS = Mangroves plus adjacent Patchy Seagrass; M-S = Mangroves without adjacent Seagrass) in September (Spring) 2003. P-values for terms in the analysis are shown. P-values shown in bold are interpreted in post-hoc tests and shown in associated figures.

Variable Analysed	Mosaic Type	Site (Mosaic)	Distance	Mosaic x Distance	Distance x Site (Mosaic)
Species Richness	0.13	0.88	0.29	0.42	0.43
Total Abundance [1]	**0.02**[A]	0.41	0.72	0.37	0.46
Acanthopagrus australis [1]	0.22	0.25	0.01	**0.001**[B]	0.25
Ambassis marianus [1]	0.01	0.73	0.11	**0.03**[C]	0.43
Gobies [1]	0.44	0.21	0.18	0.14	**0.001**
Liza argentea [1]	0.001	0.90	0.82	0.07	**0.02**
Marilyna pleurosticta [1]	**0.01**	**0.04**	0.97	0.74	0.94
Mugilogobius stigmaticus [1]	0.23	0.08	**0.02**	0.35	0.55
Tetractenos hamiltoni [1]	0.04	0.78	0.30	**0.05**[D]	0.27
Valamugil georgii [1]	0.01	0.83	0.10	**0.05**[E]	0.29
Palaemon debilis [1]	**0.05**	**0.04**	**0.04**	0.41	0.73
Metapenaeus bennettae [1]	0.001	0.90	0.02	**0.002**[F]	0.64

[1] – data transformed to $\log_e(x+1)$

[A] – test for differences among mosaic types based on pooled terms for Site (Mosaic) + Residual

[B, C, D, E, F] – test for differences in Mosaic x Distance interaction based on pooled terms for Distance x Site (Mosaic) + Residual

Table 6.8: Summary analyses of variance comparing species richness, the total number of individuals and the abundance of common species using mangroves during the night in each of three different mosaic types (M+DS = Mangroves plus adjacent Dense Seagrass; M+PS = Mangroves plus adjacent Patchy Seagrass; M-S = Mangroves without adjacent Seagrass) in September (Spring) 2003. P-values for terms in the analysis are shown. Data are pooled across two sites for the M+DS and M-S mosaic; data were only available from a single site in the M+PS mosaic. P-values shown in bold are interpreted in post-hoc tests and shown in associated figures.

Variable Analysed	Mosaic Type	Distance	Mosaic x Distance
Species Richness [1]	0.05	0.001	**0.001**
Total Abundance [1]	**0.001**	0.023	0.76
Ambassis marianus [1]	**0.001**	0.46	0.27
Gobies [1]	**0.02**	**0.002**	0.66
Liza argentea [1]	0.001	0.02	**0.02**
Marilyna pleurosticta [1]	**0.001**	0.12	0.06
Mugilogobius stigmaticus [1]	**0.01**	**0.001**	0.67
Tetractenos hamiltoni [1]	0.001	0.08	**0.05**
Valamugil georgii [1]	**0.001**	**0.05**	0.61
Palaemon debilis [1]	**0.001**	**0.58**	0.44
Metapenaeus bennettae [1]	0.001	0.07	**0.01**
Metapenaeus ensis [1]	0.001	0.11	**0.001**

[1] – data transformed to $\log_e(x+1)$

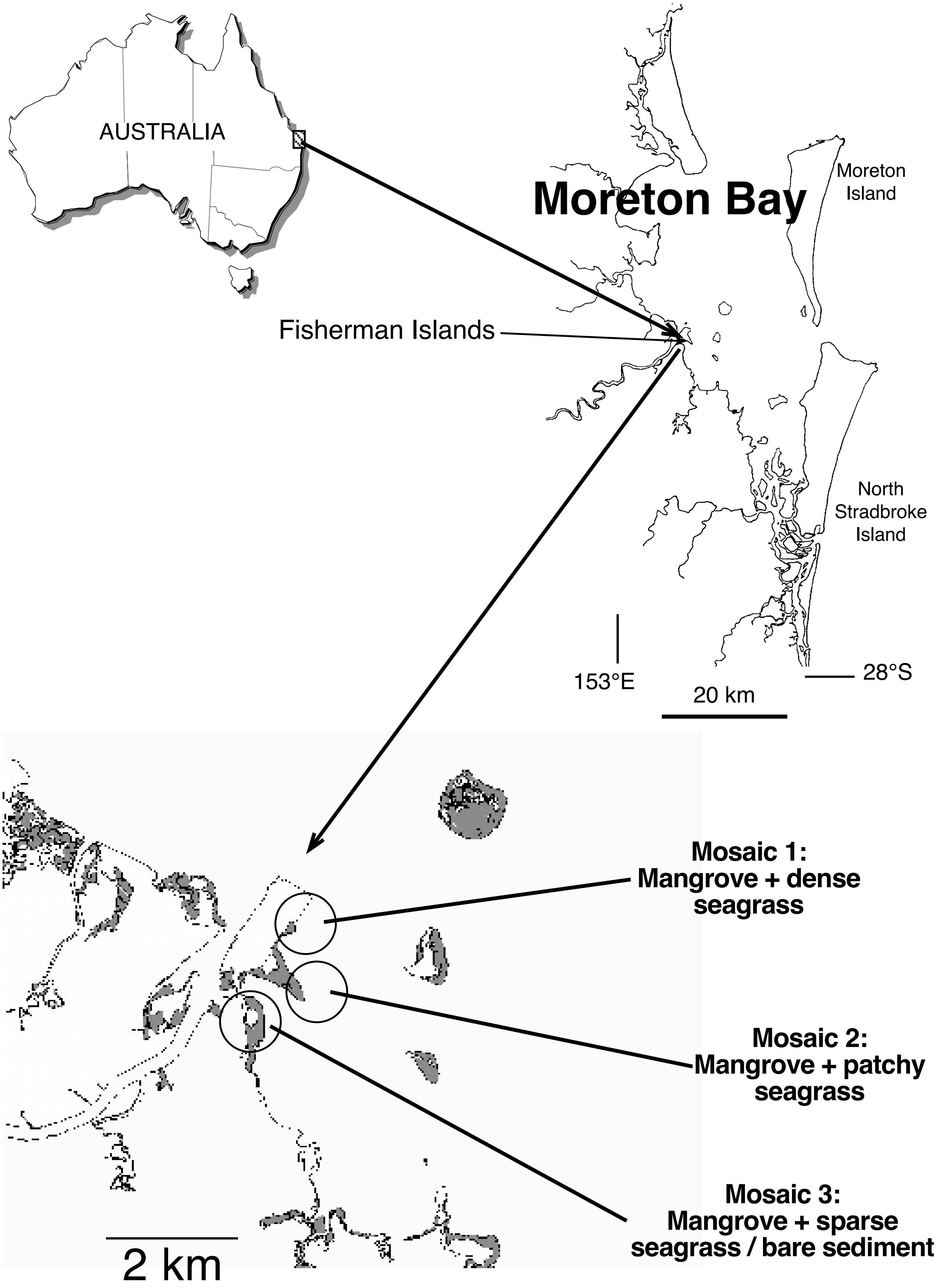


Figure 6.1: Map of Australia and the Moreton Bay region of SE Queensland showing the different sites sampled around Fisherman Islands to determine the effects of spatial arrangement of the downshore habitats on utilisation of mangrove forests by nekton.

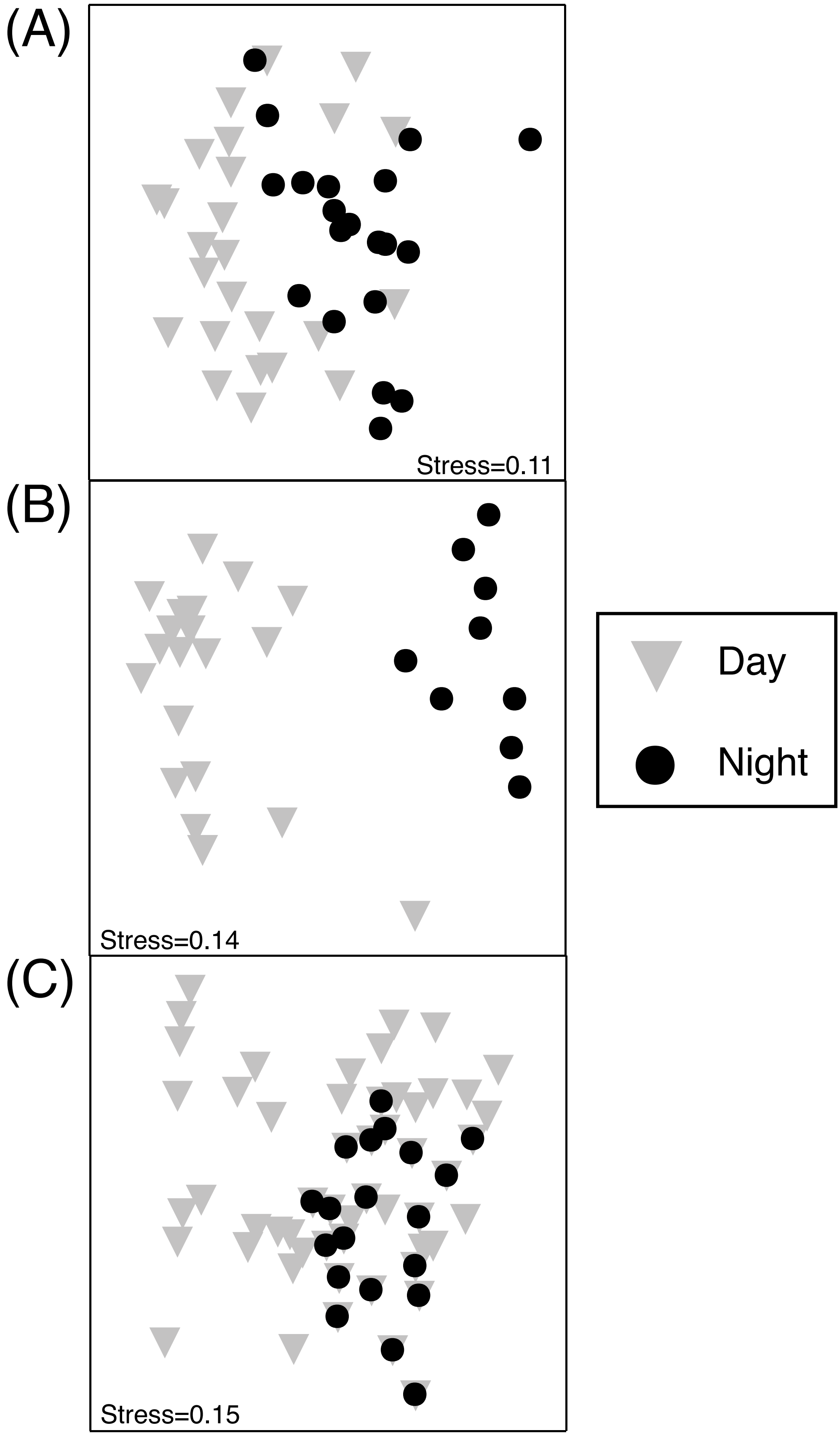


Figure 6.2: nMDS ordinations of the composition of the nekton assemblage utilising mosaics comprising (A) mangroves + dense seagrass (M+Dense Grass), (B) mangroves + patchy seagrass (M+Patchy Grass) and (C) mangroves - seagrass (M-Grass) during the day and at night in Spring 2003. Data are the number of animals, standardisedfor a 2 hour soak time, caught in fyke nets set two different distances into the forest. (Nominally n=5 replicate fyke nets deployedat each of two sites within each mosaic, although some sites did not flood on all occasions. See text for further details).

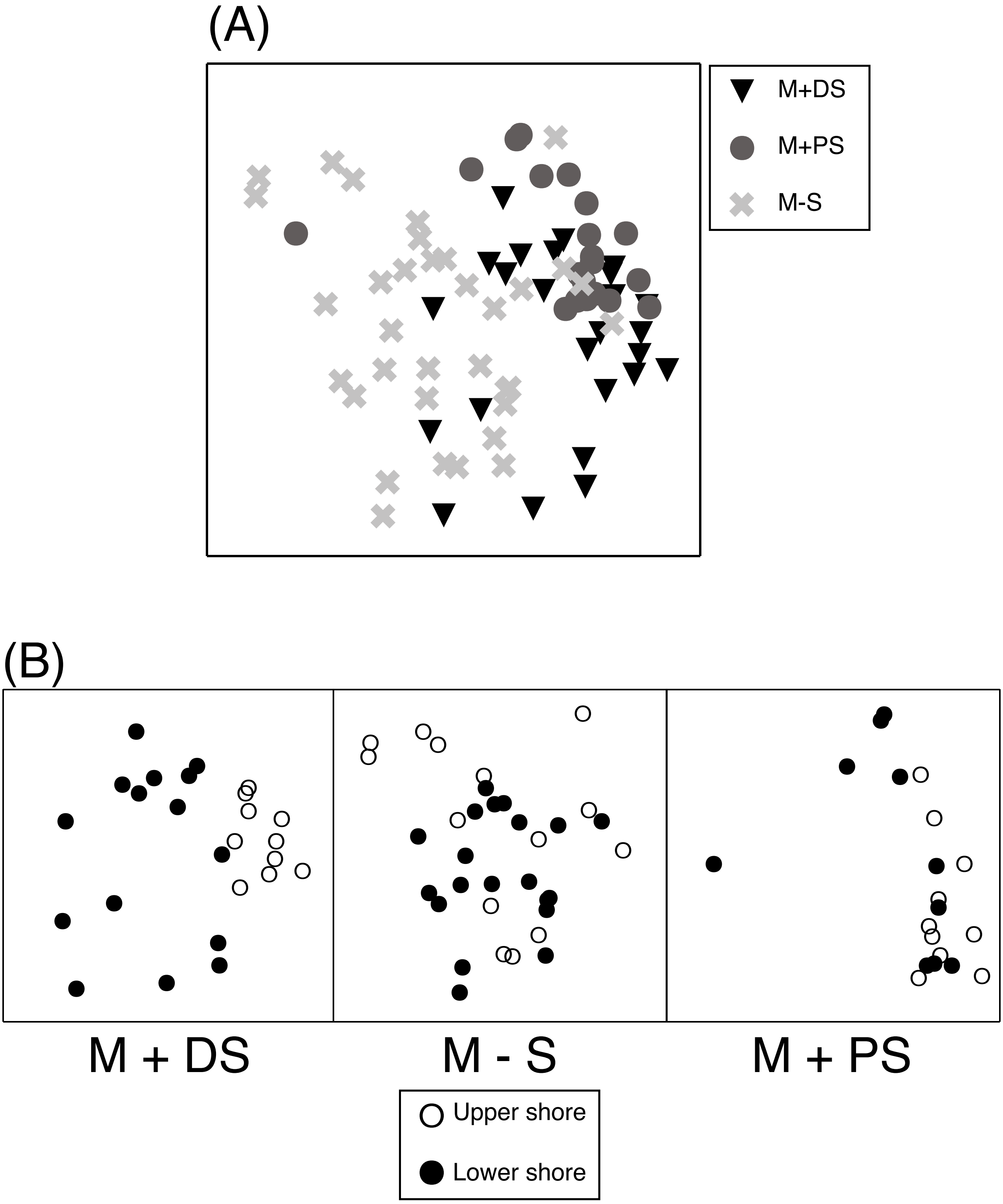


Figure 6.3: nMDS ordinuations showing the composition of the nekton assemblage utilising three different types of mangrove-based mosaics (M+DS = mangroves with adjacent dense seagrass, M+PS = mangroves with adjacent patchy seagrass, M-S = mangroves without adjacent seagrass) in western Moreton Bay during the day in September (spring) 2003. (A) compares the three mosaics, with catches pooled across each of two heights and two sites per mosaic. (B) shows the variation between each of the two distances (1-2 & 15-20 m) into the forest for each of the three types of mosaic separately. Stress = 0.12.

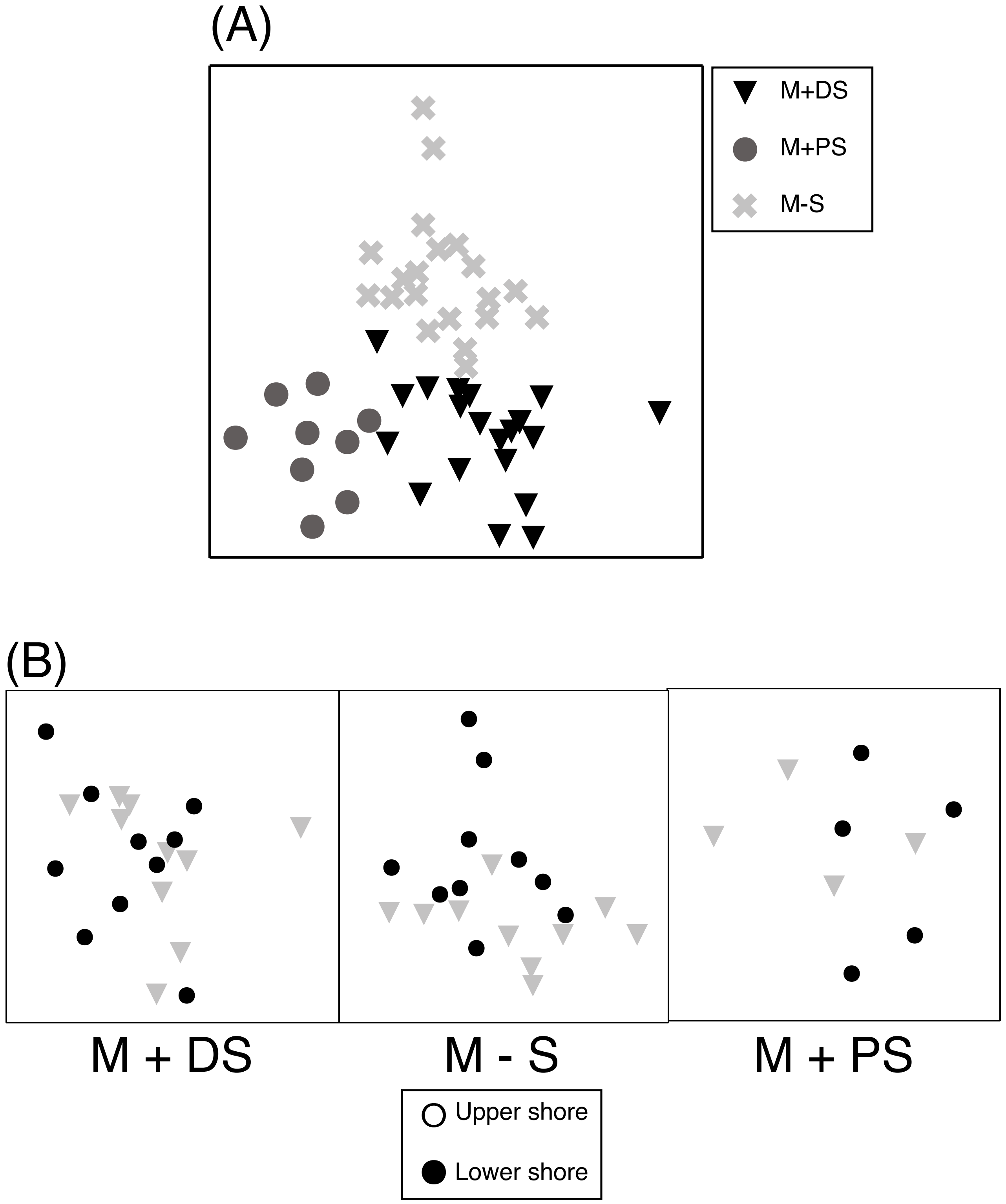


Figure 6.4: nMDS ordinuations showing the composition of the nekton assemblage utilising three different types of mangrove-based mosaics (M+DS = mangroves with adjacent dense seagrass, M+PS = mangroves with adjacent patchy seagrass, M-S = mangroves without adjacent seagrass) in western Moreton Bay during the night in September (spring) 2003. (A) compares the three mosaics, with catches pooled across each of two heights and two sites per mosaic. (B) shows the variation between each of the two distances (1-2 & 15-20 m) into the forest for each of the three types of mosaic separately. Stress = 0.13.

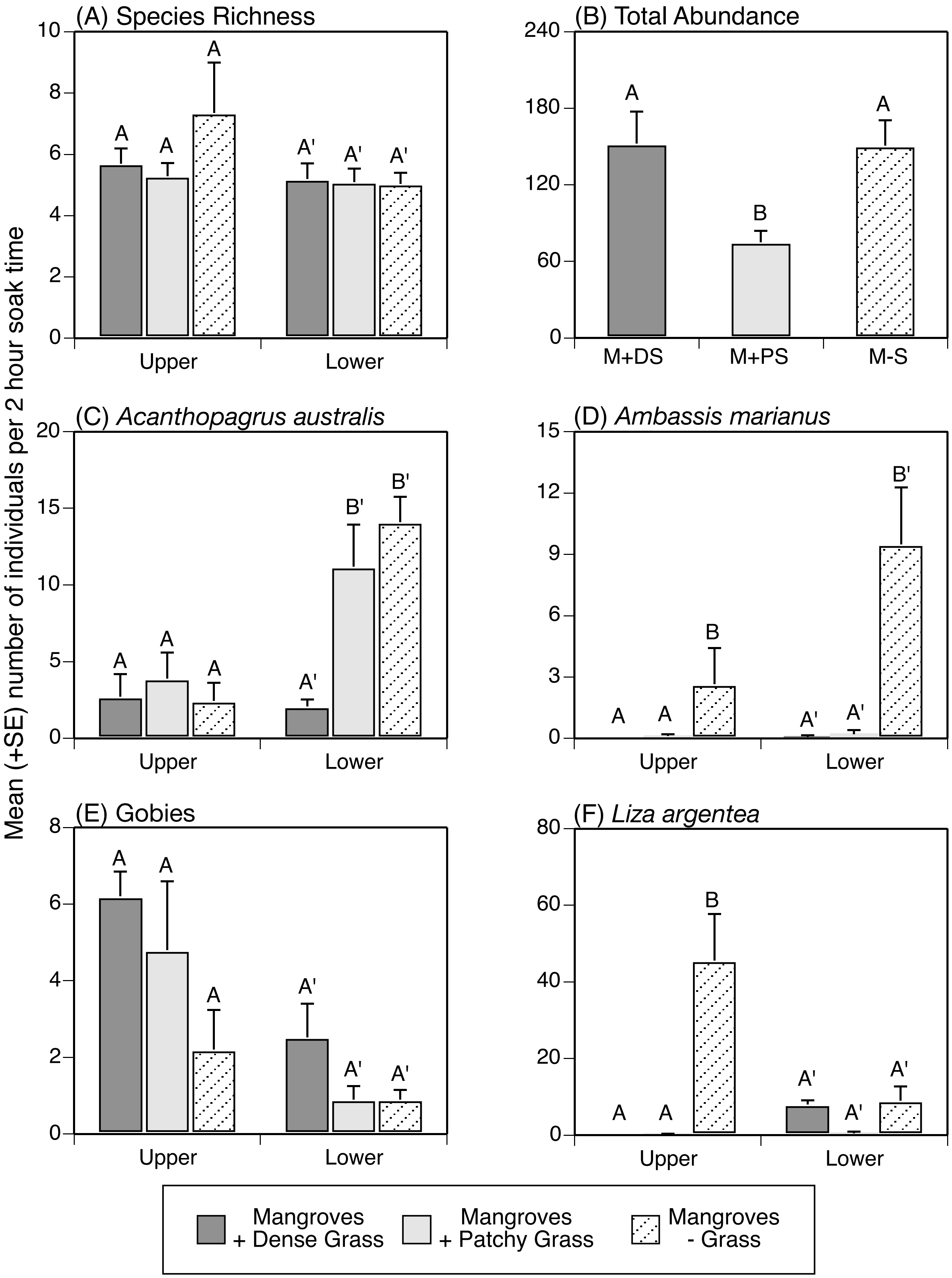


Figure 6.5: Mean (+SE) number of (A) species, (B) all individuals, (C) *Acanthopagrus australis*, (D) *Ambassis marianus*, (E) gobies and (F) *Liza argentea* caught during the day in fyke nets deployed in three different types of mangrove mosaics in September, 2003. N=5 replicate nets deployed at each of two distances into the forest (upper = 15-20 m, lower = 1-2 m) at each of 2 sites per mosaic.

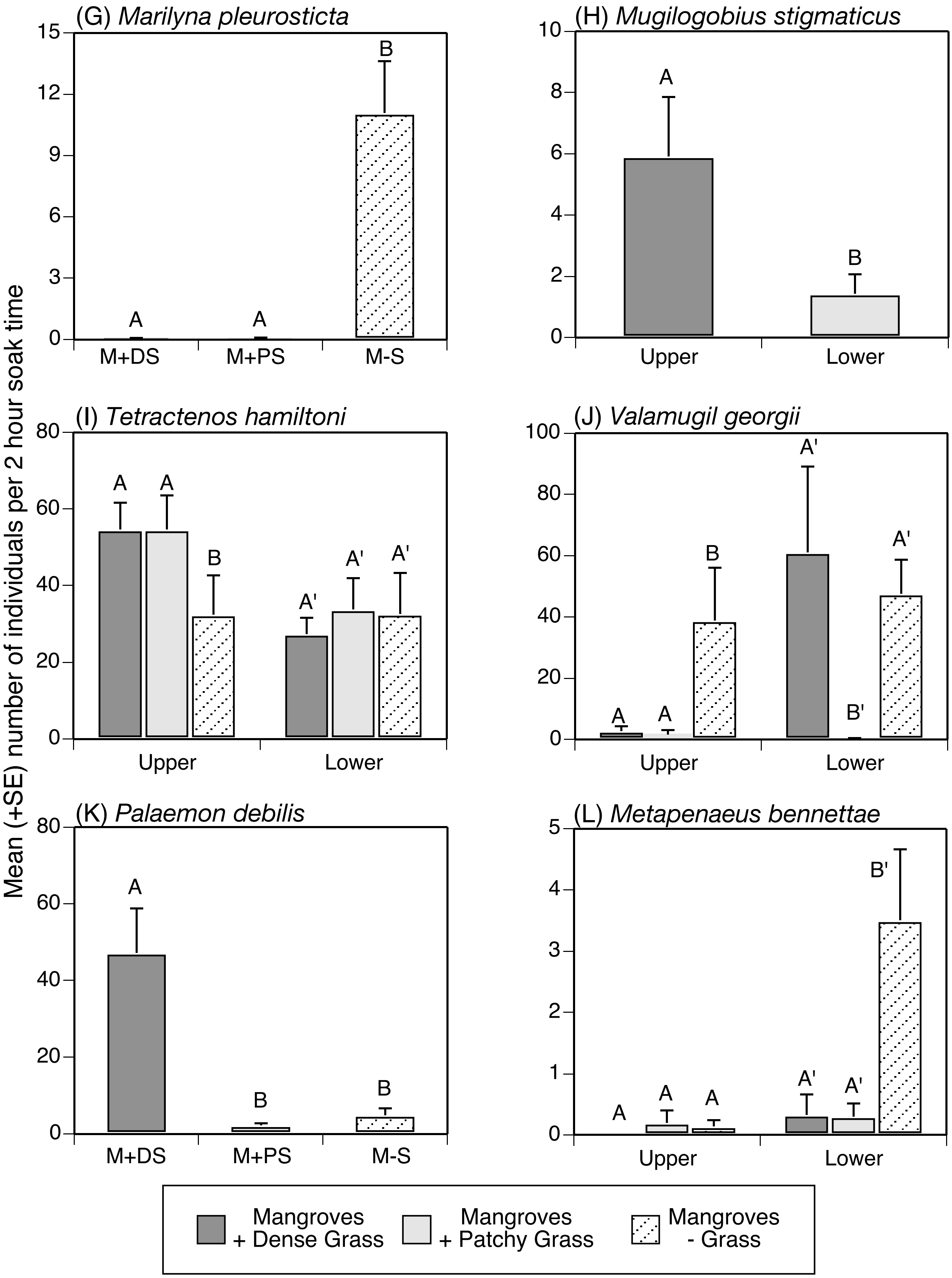


Figure 6.5 cont: Mean (+SE) number of (G) Marilyna pleurosticta, (H) *Mugilogobius stigmaticus,* (I) *Tetractenos hamiltoni,* (J) *Valamugil georgii*, (K) *Palaemon debilis* and (L) *Metapenaeus bennettae* caught during the day in fyke nets deployed in three different types of mangrove mosaics in September, 2003. N=5 replicate nets deployed at each of two distances into the forest (upper = 15-20 m, lower = 1-2 m) at each of 2 sites per mosaic.

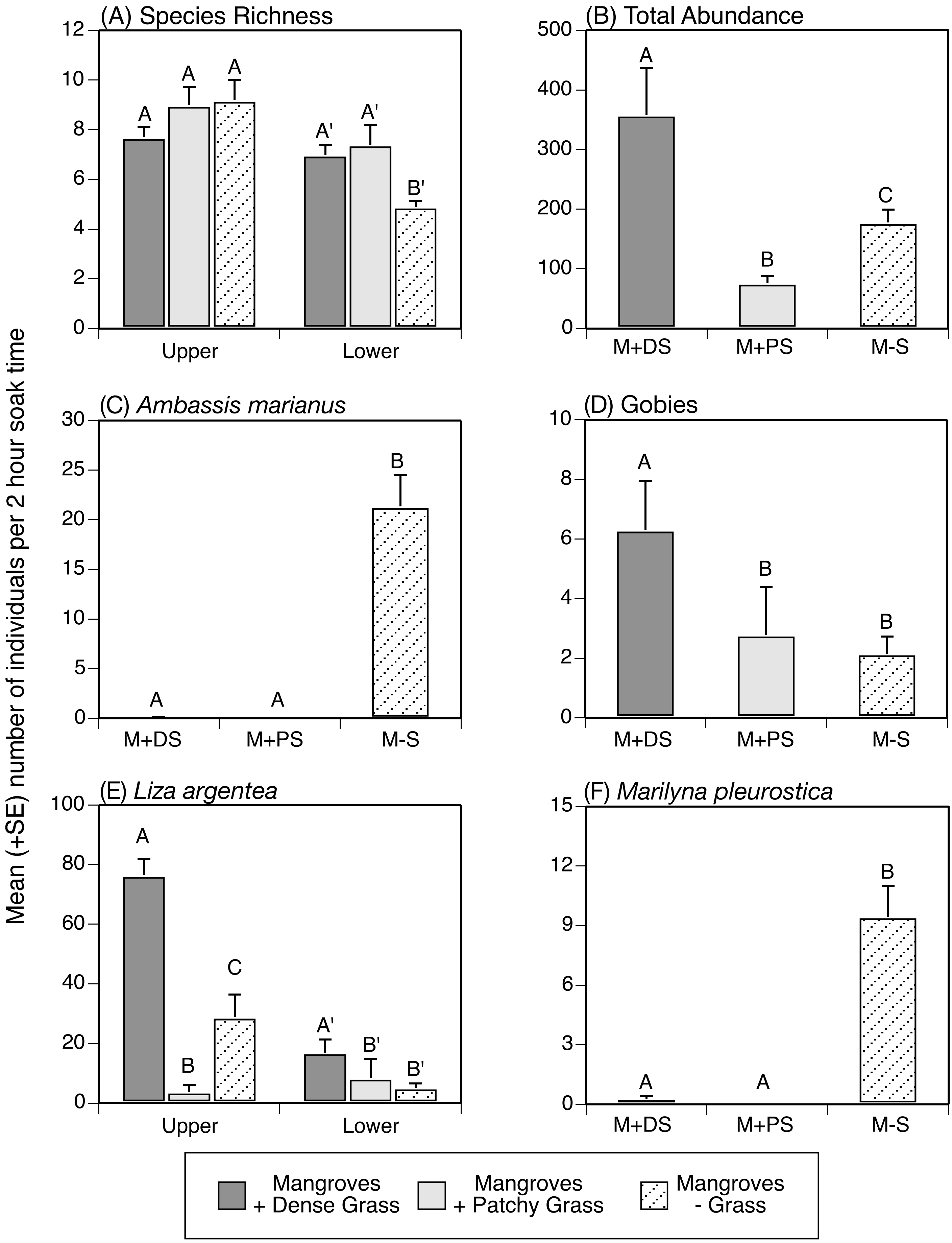


Figure 6.6: Mean (+SE) number of (A) species, (B) all individuals, (C) *Ambassis marianus*, (D) gobies, (E) *Liza argentea* and (F) *Marilyna pleurosticta* caught at night in fyke nets deployed in three different types of mangrove mosaics in September, 2003. N=5 replicate nets deployed at each of two distances into the forest (upper = 15-20 m, lower = 1-2 m) at each of 2 sites per mosaic.

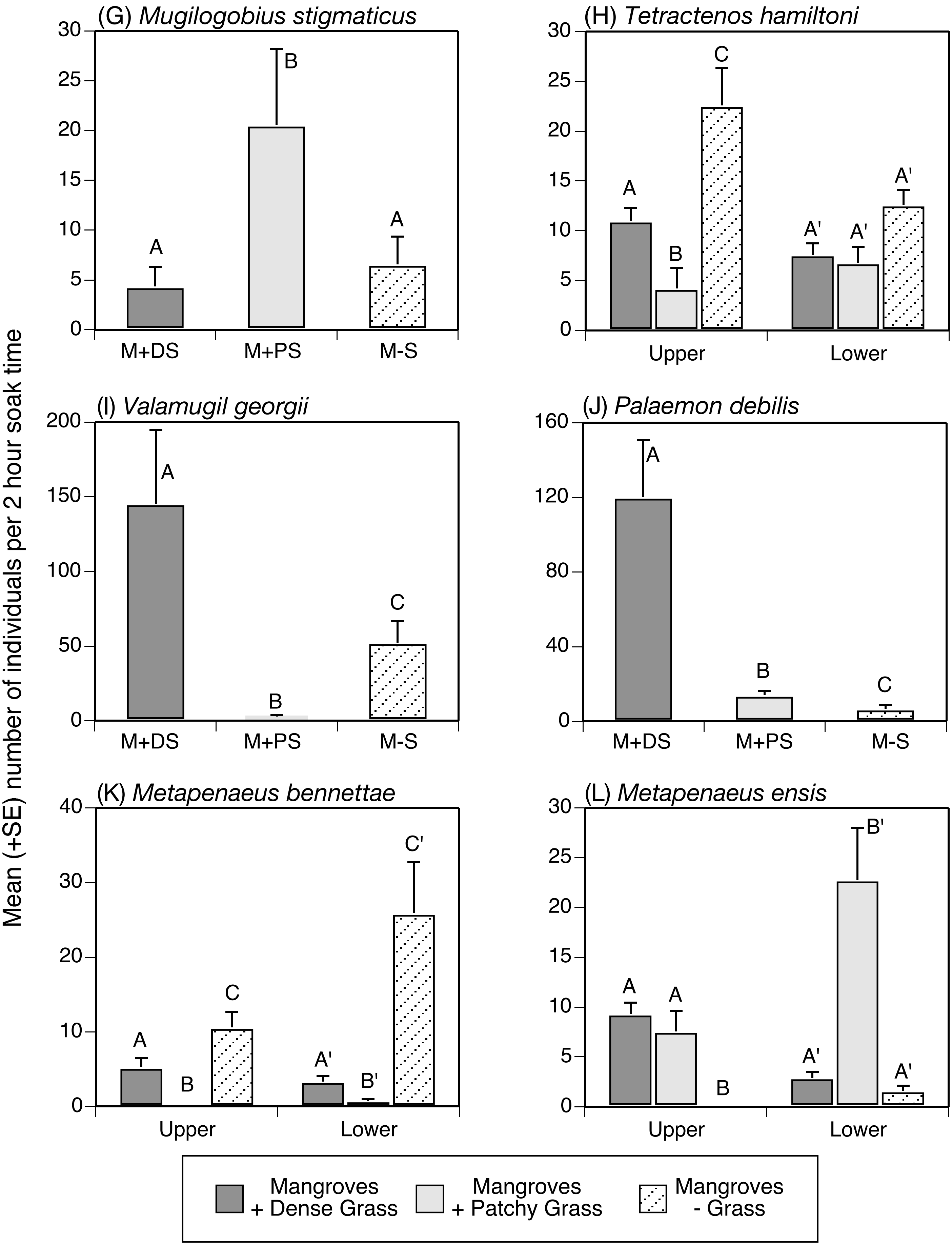


Figure 6.6 cont: Mean (+SE) number of (G) Mugilogobius stigmaticus, (H) *Tetractenos hamiltoni*, (I) *Valamugil georgii*, (J) *Palaemon debilis*, (K) *Metapenaeus bennettae* and (L) *Metapenaeus ensis* caught at night in fyke nets deployed in three different types of mangrove mosaics in September, 2003. N=5 replicate nets deployed at each of two distances into the forest (upper = 15-20 m, lower = 1-2 m) at each of 2 sites per mosaic.

Section 7: Utilisation of Different Mosaics in Southern Moreton Bay

7.1. Introduction

In terrestrial and aquatic ecosystems, adjacent and nearby patches of different habitats form a connected and complex mosaic (Wiens et al., 1993, 1997; Micheli and Peterson, 1999; Virkkala et al., 2004; Skilleter et al., 2005; Zharikov et al., 2005). In aquatic environments, patches may be either directly or indirectly linked through the movement of animals (Irlandi and Crawford, 1997; Micheli and Peterson, 1999) but also via the flow of carbon and nutrients (Alongi, 1990; Koch and Madden, 2001; Dorenbosch et al., 2004; Guest et al., 2004; Melville and Connolly, 2005) and these patches represent heterogeneity at a landscape scale (Forman and Godron, 1986; Robbins and Bell, 1994; Irlandi and Crawford 1997, Skilleter and Loneragan 2003).

In estuaries and coastal regions, many of the habitats that are considered to be critical to different life history stages of animals, including fish, prawns and crabs (i.e. nekton, *sensu* Kneib, 1997), are accessible for different amounts of time. Mangroves and saltmarsh, found high in the intertidal, may only be inundated for short periods of time (Skilleter and Loneragan, 2003; Skilleter et al., 2005), sometimes only during spring tidal periods (e.g. Australian saltmarshes - Connolly, 1999), severely limiting their use by nekton (Sheaves, 2005). The area of accessible high intertidal habitat, in effect, expands and shrinks with the rising and falling tide (Kneib and Wagner, 1994), but provides expanded foraging areas or refuges from predation for the short amount of time that they are available (Minello and Zimmerman, 1983; Boesch and Turner, 1984; Allen et al., 1994). When the habitats are no longer available (i.e. during low tide) fish must seek out alternative habitats in which to forage and take refuge from predators (Irlandi and Crawford, 1997). Any consideration of the importance of specific habitats to mobile animals such as fish must take into account their dependence on the neighboring habitats within a mosaic (Schlosser, 1995).

Different types of habitat within a mosaic will provide animals with different resources needed for their growth and survival (McIntyre and Barrett, 1992; Wiens 1997), making each of these habitats vital to the survival of a species. It is, however, not only the presence and proximity of different types of habitat (e.g. Skilleter et al., 2005) that are important to the animals using them. The specific features and characteristics of each individual habitat and of the broader landscape scale are also likely to be important in determining the distribution and abundance of species through their effects on many ecological processes (Brotons et al. 2003a). The combinations of habitats that make up different mosaics may directly or indirectly influence populations through a variety of interactions, including predation/grazing (Micheli, 1997; Irlandi and Crawford, 1997; Micheli and Peterson, 1999; Harrison and Hildrew, 2001), competition (Silver et al., 2000; Buckley and Roughgarden, 2005), susceptibility to disturbance (Ward et al., 2002; Argent et al., 2003) and access to resources (Kouki, 1991; Wellnitz et al., 2001; Silver et al., 2000). It is well established that the specific characteristics of a mosaic and the component habitats, such as the structural complexity of seagrass (Bell and Westoby, 1986a, b; Hovel et al., 2002) or the density of mangrove pneumatophores (Laegdsgaard and Johnson, 1995), have the potential to affect fish community dynamics. What has not been established though, is how the role of the structural complexity of habitats such as seagrass beds is dependent on the mosaic in which they are situated and how these relationships vary as the overall composition of the mosaics change. The characteristics of mosaics and the component habitats may together be important in governing what ecological processes occur in a particular mosaic (Wiens, 2002; Brotons et al., 2003).

The work described in the previous sections of this report was all based around mosaics found in the open areas of northern Moreton Bay. These areas are characterised by large open, expanses of coastline within the estuarine embayment. We have shown that the abundance and composition of nekton assemblages within this part of Moreton Bay are strongly influenced by the landscape characteristics of the system, through important attributes such as the proximity between different habitats and the nature of the corridors connecting different habitats. But how general are these relationships and patterns? Are the effects of attributes such as connectivity among patches and the structure and composition of corridors between patch types on the use of estuarine mosaics by nekton consistent in a different landscape context (sensu Wiens, 2002)?

The primary aim in this final section of the report was to determine the extent to which these relationships between the abundance and composition of the nekton assemblage and the spatial arrangement of estuarine habitats occurred in a different system. Southern Moreton Bay was selected as the region for this independent validation of the application of landscape techniques because the configuration of habitats is markedly different from eastern and western Moreton Bay where all previous work has been done. In particular, the southern region of Moreton Bay is dominated by small, mangrove-covered islands separated by deep channels, as opposed to the extensive, elongated intertidal flats present in the western and eastern regions.

The following core questions are addressed in this component of the study:
(i) How does the composition of the nekton community vary among different mosaics in southern Moreton Bay?; and
(ii) How does the extent and physical characteristics of the component habitat types within each mosaic affect the numbers and types of nekton using that mosaic and does the influence of variation in habitat characteristics affect comparisons among the mosaics?

7.2. Methods

7.2.1. Study Area

This study took place in southern Moreton Bay, south-east Queensland (27°S, 153°E, Figure 7.1). Southern Moreton Bay contains all of the main habitats that are associated with estuarine ecosystems (seagrass meadows, unvegetated mudflats, mangrove forests and saltmarsh). Seagrass beds are dominated by *Zostera capricorni* (Young and Kirkman 1975) and mangrove forests (primarily *Avicennia marina* and *Rhizophora stylosa*) line much of upper intertidal regions (Manson et al. 2003).

7.2.2. Identification of Mosaics and Sites

The extent of the different habitat types, extending from the shallow subtidal zone to supratidal areas, was the primary criterion used to define the broad categories of mosaic, but there is often considerable variation in the extent of these habitats within a category. It is important to identify the relative influence of this variation on the composition of the nekton communities compared with the effect of differences in the categories of mosaic because this information can then be used to design future sampling and monitoring programmes.

The different types of mosaic, the composition of the mosaics and the habitat characteristics represent three different layers of data in an increasing order of resolution (decreasing order of scale). Most past research linking nekton communities to critical estuarine habitats has been done at the smallest scale (physical characteristics of the habitat: e.g. saltmarshes - Vince et al., 1976; Minello and Zimmerman, 1983; seagrass – Coen et al., 1981; Heck and Thoman, 1981; Stoner, 1982; Summerson and Peterson, 1984; Leber, 1985; Kenyon et al., 1995). The sampling programme that has been implemented in southern Moreton Bay has incorporated each of these scales in order to assess the role of different mosaics in determining the composition of nekton communities.

Data on the spatial distribution of different estuarine habitats were obtained from aerial photographs and entered into a GIS database. Methods for quantifying the spatial extent of the different habitats and for identifying recurring combinations of these habitats – that is, what mosaics are present – are fully described in Zharikov et al. (2005) and Section 3 of this report. This approach identifies, based on the presence and extent of different habitats, the most prominent of the various combinations of habitats. Five mosaics were found to dominate the landscape-scale composition of the estuarine region of Southern Moreton Bay.

1. Urbanised coast with a narrow intertidal zone. This mosaic is dominated a bare subtidal zone but with small amounts of intertidal seagrass and mangroves. This type of mosaic represents 42 % of the coastal study area in southern Moreton Bay, with n = 113 different sites available for sampling (hereafter 'Coast-Urbanised');

2. Mangrove dominated shoreline with up-shore wetlands present, no intertidal seagrass and a bare subtidal zone, representing 29 % of the shoreline, with n = 77 different sites present (hereafter 'Coast-Mangroves');

3. Areas dominated by intertidal continuous seagrass, with a vegetated (seagrass) subtidal zone and abundant mangroves, representing 10 % of the shoreline, with n = 28 sites present (hereafter 'Continuous Seagrass');

4. Off-shore banks with extensive mangrove cover, continuous intertidal seagrass cover and an unvegetated subtidal zone, representing 46 % of the total area of offshore habitat, with n = 18 sites present (hereafter Islands-Mangroves'); and

5. Off-shore banks without mangrove cover, but dominated by continuous intertidal seagrass beds and either bare of sparsely vegetated subtidal

zone, representing 54 % of offshore habitat, with n = 21 sites present (hereafter 'Islands-No Mangroves').

7.2.3. Determining Fish Abundance and Assemblage Composition

7.2.3.1. Experimental Design

The different mosaics and their component habitats in southern Moreton Bay were sampled during summer (30 January - 3 March) 2005 to determine the numbers and types of fish using them. Sampling focused on mid-intertidal areas, which comprised seagrass and unvegetated mudflats. Ideally, it would have been useful to include the mangroves in the sampling regime but logistical and technical constraints precluded this. Sampling of fish in mangroves requires different sampling gear from that employed in seagrass/mudflat habitats (see Section 1) and there is currently no way of standardising the catch from these two methods in order to allow valid comparisons. Subtidal areas were not sampled because of the lack of detailed information on the characteristics of the habitats in these areas: the detailed aerial photography available for intertidal areas does not extend to the subtidal.

Ten sites were selected at random from the total number available for each of the five mosaics. The only criterion applied to this selection was that, as much as possible, sites were spread over a broad area within southern Moreton Bay to reduce the potential for spatial confounding to affect comparisons between the different mosaics (Skilleter et al., 2005). To achieve this, the southern region was divided into 5 separate sections, each comprising approximately the same total intertidal area and the sites were allocated so that roughly equal numbers of sites of each mosaic were allocated to each section (Figure 7.1). The nature of each of the five mosaics that were prevalent in southern Moreton Bay meant that there tended to be either one or two different types of intertidal habitat present at each site: vegetated (seagrass) or unvegetated mudflat.

Samples at each site were collected using a small seine net (length of 6 m, height of 2 m, stretched mesh size of 2 mm). Each individual seine sample was collected along a 25 m long transect, with the net held to provide a haul of 4 m width for a total area of 100 m^2. Each seine was separated by 10 m to avoid any disturbance caused by previous hauls and therefore potential non-independence of the data. Four replicate samples were collected in each habitat in each site. In the case of the offshore bank mosaics (hereafter referred to as islands), the four replicate samples were taken from either side of the island. Sampling of the 50 sites was randomised through time in terms of which sites were sampled on any day to avoid any potential temporal confounding that would have arisen if all the sites within any mosaic were sampled together. The catch from each seine was bagged, put on ice and placed into the freezer (-20°C) for preservation.

7.2.3.2. Sample Processing

All fish from the samples were identified to species, where possible, and counted. Larval fish were excluded due to taxonomic difficulties in their identification but also because there was no way of determining whether the larvae would have settled into the habitats being sampled. For the two island mosaics, the gobies were identified to species. For the remaining three mosaics, the gobies were grouped together at the family level due to the difficulty in identifying small specimens to species and the time required for this task.

7.2.4. Habitat Characterisation

The sampling intensity (multiple habitats sampled in each of 50 landscape sampling units) necessary for an investigation of associations between nekton abundance and diversity and habitat characteristics at the landscape and habitat levels has made it logistically impossible to characterise in detail the habitat structure for every sampling (seining) site as was done in other components of the broader study (details in Section 4).

A pilot study, examining seagrass beds in 6 different, widely-spaced sites was done to determine if there were relationships between measures of % coverage and blade length (measures that can be quantified non-destructively in the field) and above-ground biomass (which requires destructive sampling and extensive laboratory processing of samples). At each site, 10 replicate cores were collected. The % coverage, average length of seagrass blades and above-ground (dry) biomass of seagrass was determined for each core. A 'structure index' representing the product of mean % cover and blade length was used to represent structural complexity of this above-ground component of the seagrass bed. The above-ground biomass of seagrass was strongly correlated with this structure index (Figure 7.2), with over 75 % of the variation in the above-ground biomass accounted for by variation in the structure index. Given the considerable time (and therefore cost) required to collect and process samples in the laboratory to determine dry biomass and this strong correlation, measures of shoot density and blade length were used as a surrogate for measures of seagrass biomass.

A simplified but comparable approach for habitat characterisation was then developed and employed in this study. The structural complexity of individual habitats, was done at three points approximately 60 m apart along a transect through the centre of each of the habitats in each site. At each of these three points, three 1 m^2 quadrats were placed randomly and the percent cover of seagrass, average canopy height and sediment compaction were measured. Percent coverage of seagrass was determined using the point intercept method. The average canopy height was determined by randomly selecting 10 shoots within each quadrat and measuring the longest blade to the nearest 1 mm. To determine sediment compaction, 10 penetrometer readings were taken to the nearest 1 cm. The penetrometer (278 g) was released through a 1.8 m long PVC pipe to ensure that the penetrometer entered the sediment at a constant angle and was released from a constant height.

In unvegetated sediments, only penetrability was measured using the same method and sampling intensity as described in Section 4 (and see Skilleter et al., 2005). No grain size and organic matter samples were collected. This was justified by strong correlations among these variables (sediment penetrability, grain size and organic content) established in other studies (e.g. Yates et al. 1993; Rodriguez et al. 2001; Ryu et al. 2004). In vegetated sediments sediment penetrability was again measured using the same methods and sampling intensity described in Section 4 but no samples for analysis of organic matter or grain size were collected.

Water quality and physical conditions were also recorded at each site. A hand-held water quality probe (YSI) was used to record the following parameters: water temperature (°C), conductivity (mS/cm), total dissolved solids (g/l), salinity (%), dissolved oxygen (%, mg/l) and pH. Along with these readings, cloud cover (%), wind speed (knots) and wind direction were estimated. The water depth (cm) at the time of the first seine was also measured. These parameters were recorded at each of the habitats that were sampled.

7.2.5. Statistical Analysis

Stage 1: Differences in the abundances and species richness of fish (see Table 7.1 for list of species analysed) among the different mosaics were analysed using a nested analysis of variance (ANOVA), with site nested within mosaic. Homogeneity of variances was tested using Cochran's test. Data were log transformed where necessary.

There was considerable variation in the specific characteristics of the different mosaics. The characterisation of the mosaics, based on the remote-sensing data meant that there was considerable variation in the actual coverage of seagrass from site to site. This was particularly evident for the 'Coast-Urbanised'. This mosaic was defined (see Section 7.2.2 above) as a bare subtidal zone but with small amounts of intertidal seagrass and mangroves. When sites were selected at random from the total number of potential sites for each mosaic, in some cases there was very little seagrass present. This issue also applied to the 'Coast-Mangroves' mosaic. Thus, not every site in some mosaics had a defined vegetated habitat present so only the unvegetated component was sampled. This complicated analyses because for these two mosaics, many of the ten sites that were sampled lacked a seagrass habitat. Analysing data for utilisation of the seagrass across all five mosaics would have resulted in an extremely unbalanced and potentially biased analysis (Underwood, 1981; Underwood and Chapman, 2003). Subsets of data were therefore analysed in order to assess the effects of mosaics in different groupings of mosaics (see Table 7.2). Data on the habitat characteristics of the different mosaics were also compared with the same model of ANOVA, again testing for the assumption of homoscedastic variances with Cochran's test and log transforming data where necessary, prior to analysis.

Stage 2: Analysis of the data with ANOVA indicated there were very large effects sizes (differences in average abundance and/or species richness) among different mosaics, but few analyses indicated a significant effect of mosaic (see Results below). There were also strong correlations evident between the abundance of fish and specific habitat characteristics, especially in the seagrass habitat. In order to try and account for some of the variation in numbers of fish related to small-scale (among site) variation in habitat characteristics, data were re-analysed using analysis of covariance, with covariates of canopy height (seagrass blade length), % cover of seagrass (*Zostera capricorni*) and penetrometer depth (sediment compaction). The data on the habitat characteristics for each site were not collected on the same scale as the individual seine samples for the fish (habitat data were collected from three points across the site) so it was not possible to do these analyses matching the covariates with individual replicate seine samples. For these analyses, an average value per site for the abundance of fish and the habitat characteristics was used.

Non-metric Multidimensional Scaling (nMDS) and an analysis of similarities (ANOSIM) were done on habitat characteristics data to determine the extent of variation in the habitat characteristics among sites within a mosaic type for the seagrass habitat. nMDS procedure was also used to examine the extent of variation on the composition of the fish assemblage among the different mosaics.

7.3. Results

7.3.1. General Characteristics

There were no significant differences among sites within a mosaic or among mosaics for any of the water quality variables. Over the period sampling was done (30 January - 3 March) weather conditions were relatively constant without rainfall. Temperature ranged from 26 - 31°C. Salinity in the southern region of Moreton Bay ranged from 30 - 34‰.

A total of 29,131 fish, comprising 36 species, were sampled from the different mosaics in southern Moreton Bay (Table 7.1). Of these, 10 species were sufficiently abundant for individual analysis, although the species examined varied between the vegetated and unvegetated habitats (Table 7.3). Overall, there were very large differences (% effects sizes) in the average number of each of these species between the mosaic with the largest and smallest abundances (Table 7.3). Where effect sizes were very large (> 1000 %; e.g. *Ambassis marianus* in vegetated habitats) numbers were generally small in all mosaics, so interpretation of the biological importance of these values should be done with caution. Despite this caveat, it was clear that the numbers of fish using the different mosaics in southern Moreton Bay varied considerably among the different mosaics.

7.3.2. Variation in Abundance of Fish Among Mosaics

The number of individuals using the different mosaics varied considerably depending on which of the two habitats (seagrass or unvegetated) was sampled. Not surprisingly, more individuals were caught using the seagrass habitat than the unvegetated mudflat, but the contrast among the mosaics was not consistent (Figure 7.3A). For the seagrass, more individuals were caught where there was adjacent mangroves on the mainland (Coast-Mangroves), with the smallest number of individuals using the offshore mosaics without upshore mangroves (Islands-No Mangroves). In contrast, for the mudflat habitat, the Islands-Mangroves mosaic supported the largest number of individuals with the smallest number in the Islands-No Mangroves. The number of species of fish was relatively constant across all five mosaics for the vegetated and unvegetated habitats (Figure 7.3B), but there were more species present in the seagrass than on the mudflat.

The estuary perchlet, *Ambassis marianus*, was only abundant in coastal mosaics (Coast-Mangroves and Coast-Urbanised), with slightly larger numbers caught in the seagrass than the unvegetated mudflat (Figure 7.3C). In contrast, the common fortescue, *Centropogon australis*, was predominantly found in the vegetated habitat (Figure 7.3D), with larger numbers associated with the two mosaics dominated by mangroves in the high intertidal (Coast-Mangroves and Islands-Mangroves).

The most abundant group of fish were the gobies (Table 7.1) with 18,515 fish caught (63.56% of the total catch for the study). Gobies occurred in vegetated and unvegetated habitats in all mosaics (Figure 7.3E), although numbers were very small in the unvegetated habitat for the Islands-No Mangroves mosaic. The gobies were represented by eight different species (Table 7.1), although *Pseudogobius* sp. comprised > 60 % of the total number of gobies (NB: species of gobies were only identified for two of the mosaics, so it is possible that the contribution of different species to the total numbers in other mosaics may be different). Detailed analysis of individual species of gobies was done for two mosaics – Islands-Mangroves vs Islands-No Mangroves. These two mosaics were selected because of the large differences in abundance between them (~ 100 %) in the total number of gobies present (Figure 7.3E). Analysis of individual species though simply reflected the same pattern as for the broader analysis – highly variable numbers among sites within the two mosaics, with no detectable significant difference between mosaics for either of the two habitats (Table 7.6).

The sandy sprat, *Hyperlophus translucidus*, was the only one of the common species that was relatively more abundant in the mudflat than seagrass habitat (Figure 7.3F). Numbers were similar across all five mosaics in the seagrass, but there were more sandy sprat present in the Coast-Urbanised and Island-Mangroves mosaics for the unvegetated mudflats. The fan-bellied leatherjacket, *Monocanthus chinensis*, was predominantly founds in the seagrass, with slightly more occurring in the Coast-Mangroves mosaic compared with the others (Figure 7.3G) and very small numbers over the mudflats.

The eastern striped trumpeter, *Pelates sexlineatus*, was the second most abundant species, with 3776 individuals (12.96 % of the total catch). *P. sexilineatus* was an order of magnitude more abundant in the seagrass than mudflat habitats (Figure 7.3H), with a two-fold difference among mosaics for the seagrass. Finally, the estuarine anchovy, *Thryssa aesturia*, was caught in both habitats in all mosaics except the mudflat component of the continuous seagrass mosaic (Figure 7.3I).

As noted in the Methods above (Section 7.2.5), two separate analyses were done because of the variation in the total number of sites that were sampled for each habitat in the different mosaics (Table 7.2). The first contrasted the Coast-Mangroves against the Coast-Urbanised mosaic. In the seagrass, despite the 115 % difference in the number of individuals using these two mosaics (Table 7.3), there was no significant difference in abundance of fish detected (ANOVA, Mosaics - $P > 0.17$), but there was considerable variation among the different sites (ANOVA, Sites(M) – $P < 0.005$) (Table 7.4). A similar outcome was obtained for the analysis comparing the total number of individuals in the other three mosaics (Continuous Seagrass, Islands-No Mangroves and Islands-Mangroves), with highly variable numbers occurring in the different sites (ANOVA, Sites(M) – $P < 0.001$). The analysis of the abundance of fish caught on the mudflat was also dominated by highly significant variation at the scale of sites within mosaics, but no significant difference detected among the five mosaics (Table 7.5).

The number of species and the abundance of individual species all showed similar results in the formal analysis of the data (Table 7.4 and Table 7.5): significant variation at the scale of sites, but no significant differences in abundance among mosaics, despite the often considerable differences in overall effect sizes (Figure 7.3).

7.3.3. Influence of Variation in Habitat Characteristics

Despite the large effects sizes (% differences between largest and smallest means) evident in comparisons of the utilisation of the different mosaics in southern Moreton Bay, formal analysis failed to detect significant differences among the mosaics. Analyses were dominated by highly significant variation among the sites within each of the mosaics for both the seagrass and the unvegetated habitats (Tables 7.4 and 7.5). There was also considerable variation at the scale of sites in the physical characteristics of the vegetation, especially for those sites dominated by seagrass. Examination of the data on the physical characteristics of the different habitats, especially the seagrass, indicated there were strong relationships between the abundances of fish and one or more of these habitat characteristics. For example, variation in the height of the seagrass canopy (predominantly *Zostera capricorni*) had a large influence on the abundance of two species that are strongly associated with vegetation (see Section 1), *Pelates sexlineatus* and *Centropogon australis*. This measure of structure accounted for 45 – 82 % of the variation in the abundance of these two species in the seagrass habitat across the five mosaics (Figure 7.4).

To determine whether small-scale (i.e. among sites within a mosaic) variation in the physical characteristics of the habitats could account for some of the variation in the abundance of fish at that scale, data were re-analysed using analysis of covariance (ANCOVA), using three different covariates: canopy height, % cover and sediment compaction (penetrometer depth). These analyses were based on data averaged per site for the abundance of fish and the habitat characteristics because of the different intensities at which each were sampled (see Methods above).

Canopy height and sediment compaction showed the strongest relationships with the abundance of fish in the different mosaics, with highly significant linear relationships between these two covariates and most variables (Table 7.7). In contrast, the % coverage of *Zostera* was generally a poor predictor of the variation in the abundance of any of the variables. Despite these relationships, ANCOVA did not reveal any significant differences among the mosaics (Table 7.7).

7.3.4. Composition of Fish Assemblage in Mosaics

The composition of the fish assemblage utilising the seagrass habitat varied among the different mosaics (ANOSIM, $P < 0.001$), with two distinct groupings evident in the ordination (Figure 7.5A) and based on pairwise tests among groups in ANOSIM. The assemblage using the Islands-Mangroves and Islands-No Mangroves were distinct from the assemblage using the other mosaics.

In contrast, differences in the assemblage of fish using the mudflats were only marginally significant (ANOSIM, $P < 0.06$) but there were no clear groupings of mosaics evident in the ordination or in pair-wise tests. Again though, the assemblage using the mudflats in the two Island mosaics were most distinct from the other mosaics (Figure 7.5B), based on the magnitude of the R-statistic in pairwise comparisons.

7.1. Discussion

Detailed sampling of 50 sites across southern Moreton Bay indicated that there were large differences in the average abundance of fish utilising five different mosaics. The increase in abundance between the mosaics supporting the largest and smallest numbers of fish were commonly greater than 2-3 fold in vegetated and unvegetated habitats. These very large differences in abundance in different mosaics is consistent with the studies done in other part of Moreton Bay and described in earlier sections of this report.

Despite these very large effect sizes though, the abundances of fish was not detected as varying significantly among the different mosaics, in formal analysis. There was consistently large and significant variation among the sites within each mosaic and tests (ANOVA and ANCOVA) for differences among mosaics were dependent on the variation in the nested term, Sites, which was almost always significant. Thus, the lack of significant differences in analyses could be a result of low statistical power. Alternatively, a second explanation could be that the fish communities in southern Moreton Bay, in contrast to those further north, are not influenced at the large spatial scale of the different mosaics, but respond instead to smaller-scale characteristics of the component habitats which tended to vary from site to site.

7.1.1. Insufficient Power to Detect Differences

Although there were no significant effects of mosaic in any of the analyses done for sampling in southern Moreton Bay, differences in the abundance of fish in different mosaics were, on average, as large or larger than those detected as being significant from sampling in the northern Bay. This suggests that the broader composition of the mosaic (the nature and abundance of the component habitats) is important in determining the composition of fish assemblages that are using shallow water habitats, often only accessible during high tide periods.

The value of different mosaics is likely to stem from the fact that the different habitats comprising each mosaic will provide fish with access to different resources during high tide, including enhanced (or different) food (Boesch and Turner, 1984; Allen et al., 1994) and shelter from predators (Manson et al., 2005b). This may be either indirectly or directly through outwelling of nutrients and detritus from the mangroves and saltmarsh, high in the intertidal (Alongi 1990, Koch and Madden 2001), or through fish moving into the adjacent mangroves from the downshore seagrass beds (Irlandi and Crawford 1997, Micheli and Peterson 1999). Where a mosaic contains extensive seagrass, adjacent or in close proximity to habitat such as mangroves, the combination of habitats may provide fish with increased access to the range of resources that are provided (Skilleter and Loneragan, 2003). Mosaics that lack some of the components (e.g. a mosaic without mangroves) may provide fewer resources and therefore are accessed less by fish, or fish using these mosaics suffer greater mortality.

The spatial arrangement of patches of habitat within a mosaic has been found to be important in structuring nekton communities in the more open regions of Moreton Bay (this report) and also in other parts of the world (Irlandi and Crawford, 1997; Mumby et al., 2004). Factors such as patch size and fragmentation (Eggleston et al., 1999), proximity to and nature of adjacent patches of habitat (Raposa and Oviatt 2000; Skilleter et al. 2005; Jelbart et al., 2007) and the position within a patch (Bowden et al. 2001) have all been found to influence the composition of animal communities. The results of this study in southern Moreton Bay are consistent with these findings in that large differences were present in abundances of fish species using the different mosaics. An additional component may, however, also be critical in determining the actual patterns of utilisation of different mosaics by fish: the small-scale (within site) variation in the structural characteristics of the habitat found in each mosaic.

When identifying and classifying different areas within southern Moreton Bay into the five categories of mosaic for this study, the presence and extent of particular habitats (i.e. the composition of the mosaics), as classified from analysis of remote sensing data (aerial digital imagery), were used to define each mosaic (see Zharikov et al., 2005 for detailed description of these methods). Classification of the component habitats was thus based primarily on qualitative information, rather than detailed quantitative data such as measures of structural complexity. Such information is not readily interpretable from camera-based digital imagery. The results here though indicate that the detailed information on characteristics such as canopy height and sediment characteristics are still important in determining the abundance of fish even when the focus is on broader, landscape scale patterns. Thus, it may be important that this detailed information be incorporated into the classification of different sites within the categories of mosaics, requiring detailed mapping prior to selecting specific sites and subsequent sampling of fish. Such mapping entails the collection of samples for analysis of the biomass of seagrass and sediment characteristics (e.g gravel content) and this requires a significant investment of resources.

For the purposes of this study, it was not logistically feasibly to complete detailed mapping of all 50 sites prior to commencing field sampling. In recognition of the potential importance of such information on structural complexity though, we collected data on habitat characteristics from 6 sites in southern Moreton Bay and determined that an index of structure, represented by measures of canopy height and % coverage, provided a good representation of the variation in seagrass biomass. We determined that this structure index, in combination with measures of sediment compaction, would provide a realistic surrogate for the more detailed and costly measures of habitat and this approach was implemented here across all 50 sites. These measures indicated there was substantial variation in habitat characteristics among sites selected to represent specific categories of mosaics and this variation appeared to have a strong influence on the abundance of fish at this scale.

It seems likely that the inclusion information on habitat characteristics at the scale of sites, would result in the identification of more than 5 specific categories of mosaic. For example, consideration of the habitat characteristics would result in the 'Continuous-Seagrass' mosaic being split into two categories: one with seagrass with a tall canopy and one with a much shorter canopy. Differences in canopy height, one component of our structure index, had a significant influence on the abundance of fish. Classification of the different areas within southern Moreton Bay, based on more detailed habitat information, would markedly increase the resolution with which individual sites were placed into categories of mosaic. This would reduce the small-scale habitat variation within a mosaic, and increase the capacity to identify specific assemblages associated with the different mosaics. These results do indicate though that that the preference for certain small-scale characteristics by fish may be more crucial in determining fish abundance than mere presence and extent of habitats and that both local and landscape scale habitat structure are important in governing the composition of fish communities (Hovel et al. 2002).

7.1.2. Fish Are Not Influenced by Landscape Scale Effects

An alternative explanation for the results of this study is that the fish of southern Moreton Bay are not affected by the differences in the composition of the mosaics, but respond to the smaller scale characteristics of the specific habitats. There are several possible causes for such an explanation. Fish are mobile organisms and have the ability to move between adjacent patches of habitat (Rozas and Odum, 1987; Vance et al., 1996; Irlandi and Crawford 1997). Consequently, organisms with such a level of mobility may move between different habitat mosaics, in addition to accessing the different habitats within any mosaic. Thus, they take advantage of resources, including food and shelter, accessible over scales larger than identified as discrete mosaics, so the numbers and types of fish that were caught in each of the five mosaics in this study were simply those species that happened to be there at the time.

A range of factors can influence fish abundance and distribution, with the distribution and availability of food resources considered a main determinant in structuring fish communities (Edgar and Shaw, 1995; Grenouillet and Pont, 2001; Grenouillet et al., 2002). If the distribution of prey is not influenced by the specific composition of the mosaic though, the distribution of a predator may vary at different scales that those represented by different mosaics. For example, species such as the hairy pipefish, *Urocampus carinirostris* (Kendrick and Hyndes, 2005) and the hardyhead, *Atherinomorus ogilbyi* (Ivantsoff and Crowley 1991), feed on copepods and other planktonic prey. Plankton, unlike nekton, have a limited capacity for control over their horizontal transport and are moved around in currents (Forward, 1989; Leis and Reader, 1991). The different mosaics were not specifically categorised in relation to information on hydrodynamics, so it is unlikely that these mosaics would influence or reflect the patterns of distribution and dispersal of plankton. Consequently, planktivorous fish, such as *U. carinirostris* and *A. ogilbyi*, may in fact follow the movement of tides and water currents to access food resources and their distribution will not be closely aligned to the distribution and arrangement of mosaics. Other species of fish have the ability to feed on multiple types of prey. For example, *Pelates sexlineatus* is a generalist feeder (Sanchez-Jerez et al. 2002) and *Ambassis interruptus*, a species closely related to *A. marianus*, has been found to adapt its feeding strategy in accordance with the availability of prey (Hajisamae et al. 2004). Such species are likely to access food resources from many different mosaics, and so the particular mosaic in which they are caught at any time is just circumstantial. More detailed information is needed on the diet of fish in the local (Moreton Bay) systems and whether diet varies in relation to the spatial arrangement of the component habitats within a mosaic (see also Section 5).

Overall, the results of this study in southern Moreton Bay are consistent with the specifics identified for other parts of the embayment: the spatial arrangement of habitats has a strong influence on the numbers and types of fish that use different areas within an estuary. The specifics characteristics of individual habitats is clearly very important in determining small scale patterns of distribution and abundance, but over and above this, there are broader patterns that reflect the landscape composition of the region.

Table 7.1: List of species sampled from the 50 sites in southern Moreton Bay in summer 2005. (A) The total number of individuals for each species is shown. Species in **bold** were those analysed in detail. (B) Species of gobies sampled from the two island mosaics – Islands with Mangroves and Islands without Mangroves. Difficulties in identifying very small individuals of the gobies and the large numbers of individuals that were caught precluded species-level identification for the samples from the other mosaics.

(A)

Species	Number	Percent of Total Individuals
Acanthopagrus australis	5	0.02%
Ambassis marianus	**163**	**0.56%**
Arothron hispidus	7	0.02%
Arrhampus sclerolepis	1	0.00%
Atherinomorus ogilbyi	**152**	**0.52%**
Centropogon australis	**2523**	**8.66%**
Eurysthmus leptus	3	0.01%
Gerres subfasciatus	55	0.19%
Gobiidae	**18515**	**63.56%**
Herklotsichthys castelnaui	57	0.20%
Hyperlophus translucidus	**1123**	**3.85%**
Hyporhampus regularis ardelio	6	0.02%
Liza argentea	1	0.00%
Lutjanus fulvifamma	22	0.08%
Lutjanus russellii	2	0.01%
Marilyna pleurosticta	3	0.01%
Monacanthus chinensis	**472**	**1.62%**
Parachaetodon ocellatus	1	0.00%
Pegasus volitans	1	0.00%
Pelates sexlineatus	**3776**	**12.96%**
Petroscirtes lupus	75	0.26%
Pseudorhombus arsius	14	0.05%
Pseudorhombus jenysii	2	0.01%
Rhabdosargus sarba	41	0.14%
Siganus fuscescens	**120**	**0.41%**
Sillago analis	6	0.02%
Sillago maculata	99	0.34%
Sillago sp.	3	0.01%
Sphyraena obtusata	49	0.17%
Synanceia horrida	1	0.00%
Synaptura nigra	2	0.01%
Tetractenos hamiltoni	42	0.14%
Thryssa aesturia	**1265**	**4.34%**
Torquigener pleurogramma	4	0.01%
Tripodichthys angustifrons	34	0.12%
Urocampus carinirostris	**481**	**1.65%**

(B)

Gobiid Species (two mosaics)	Number	% of gobies
Acentrigobius caninus	1	0.01 %
***Amoya* sp.**	**1100**	**14.3 %**
Arenigobius frenatus	**660**	**8.6 %**
Bathygobius kreffti	154	2.0 %
Coryphotpterus neophytes	**715**	**9.3 %**
Gnatholepis gymnocara	89	1.2 %
***Pseudogobius* sp.**	**4938**	**64.4 %**
Redigobius macrostomus	10	0.1 %

Table 7.2: List of analyses that were completed for vegetated and unvegetated habitats for each of the different mosaics. + or - denotes whether an analysis was done. Where a letter symbol (A/ or B) is present alongside the +, this indicates that those mosaics with the same letter were analysed separately from those with the other letter. See text for further detail as to why each of the analyses was done. The number of sites (from a potential total of 10 sites per habitat per mosaic) that were sampled in each of the vegetated and unvegetated habitats is also shown.

Vegetated Habitats							
Mosaic	Number of Sites with Vegetated Habitat	ANOVA (abundance)	ANOVA (goby abundance)	ANCOVA (canopy height)	ANCOVA (pecent coverage)	ANCOVA (penetrometer)	nMDS
Coast - Mangroves	3	+ A	-	+	+	+	+
Coast - Urbanised	4	+ A	-	+	+	+	+
Continuous Seagrass	10	+ B	-	+	+	+	+
Islands - No Mangroves	10	+ B	+	+	+	+	+
Island - Mangroves	9	+ B	+	+	+	+	+

Unvegetated Habitats							
Mosaic	Number of Sites with Unvegetated Habitat	ANOVA (abundance)	ANOVA (goby abundance)	ANCOVA (canopy height)	ANCOVA (pecent coverage)	ANCOVA (penetrometer)	nMDS
Coast - Mangroves	10	+	-	-	-	+	-
Coast - Urbanised	10	+	-	-	-	+	-
Continuous Seagrass	6	+	-	-	-	+	-
Islands - No Mangroves	7	+	+	-	-	+	-
Island - Mangroves	9	+	+	-	-	+	-

Table 7.3: Mean number (±SE) of individuals, species and common species in vegetated and unvegetated habitats for each of the five mosaics. The mosaic with the largest mean abundance for each variable is shown in **bold**; smallest mean abundance is underlined. The final column shows the effect size as the % increase in abundance from the smallest to the largest value for that variable.

			Vegetated Habitats			% Effect Size
	Coast - Mangroves	Coast - Urbanised	Continuous Seagrass	Islands - No Mangroves	Islands - Mangroves	
Number of Individuals	**201.1 (± 18.8)**	93.5 (± 22.0)	157.7 (± 16.0)	84.3 (± 7.8)	171.2 (± 15.7)	138.6
Number of Species	**7.1 (± 0.4)**	5.4 (± 0.5)	6.0 (± 0.4)	5.8 (± 0.3)	5.8 (± 0.3)	31.5
Ambassis marianus	**3.4 (± 0.9)**	2.1 (± 0.6)	0.1 (± 0.0)	0.03 (± 0.0)	0.1 (± 0.0)	11,233.3
Atherinomorus ogilbyi	0.4 (± 0.1)	0.6 (± 0.3)	**2.7 (± 1.3)**	0.03 (± 0.1)	0.1 (± 0.1)	8,900.0
Centropogon australis	**44.3 (± 5.0)**	12.2 (± 2.6)	11.5 (± 1.9)	10.1 (± 1.8)	25.0 (± 4.0)	338.6
Gobiidae	84.2 (± 13.5)	36.2 (± 19.7)	**105.7 (± 12.3)**	48.8 (± 5.0)	94.3 (± 10.8)	292.0
Hyperlophus translucidus	1.8 (± 1.9)	0.8 (± 0.4)	1.9 (± 0.9)	**2.6 (± 0.9)**	1.3 (± 2.4)	225.0
Monocanthus chinensis	**7.3 (± 1.1)**	2.7 (± 0.7)	2.1 (± 0.5)	3.1 (± 0.7)	3.6 (± 0.8)	247.6
Pelates sexlineatus	**50.4 (± 5.3)**	23.3 (± 4.9)	22.3 (± 3.8)	10.7 (± 1.4)	37.9 (± 5.2)	371.0
Siganus fuscescens	0.3 (± 0.1)	**2.4 (± 0.6)**	0.6 (± 0.2)	0.7 (± 0.3)	0.7 (± 0.2)	700.0
Thryssa aestuaria	2.8 (± 2.3)	**9.1 (± 3.5)**	3.8 (± 1.2)	2.7 (± 1.0)	2.1 (± 1.5)	233.3
Urocampus carinirostris	**4.4 (± 0.5)**	2.1 (± 0.4)	3.6 (± 0.5)	3.1 (± 0.5)	3.2 (± 0.5)	109.5

			Unvegetated Habitats			% Effect Size
	Coast - Mangroves	Coast - Urbanised	Continuous Seagrass	Islands - No Mangroves	Islands - Mangroves	
Number of Individuals	72.7 (± 19.7)	67.4 (± 16.4)	18.3 (± 15.9)	14.4 (± 7.3)	**89.5 (± 16.5)**	421.5
Number of Species	2.4 (± 0.3)	2.3 (± 0.3)	1.5 (± 0.4)	2.7 (± 0.3)	**2.8 (± 0.3)**	86.7
Ambassis marianus	**1.4 (± 1.0)**	0.6 (± 0.5)	0.04 (± 0.0)	0.01 (± 0.0)	0.0 (± 0.0)	13,900.0
Atherinomorus ogilbyi	0.03 (± 0.0)	0.01 (± 0.2)	0.3 (± 1.3)	**0.4 (± 0.1)**	0.3 (± 0.1)	9,900.0
Centropogon australis	0.2 (± 0.1)	**0.4 (± 1.9)**	0.1 (± 1.9)	0.01 (± 1.6)	0.2 (± 3.8)	9,900.0
Gobiidae	62.1 (± 15.1)	49.4 (± 14.2)	17.1 (± 12.2)	4.1 (± 5.0)	**71.4 (± 11.6)**	1,641.5
Hyperlophus translucidus	2.5 (± 1.9)	**9.4 (± 4.6)**	0.01 (± 0.9)	3.0 (± 1.0)	9.3 (± 2.6)	93,900.0
Monocanthus chinensis	0.03 (± 1.1)	0.03 (± 0.4)	0.01 (± 0.5)	**0.04 (± 0.7)**	0.0 (± 0.9)	300.0
Pelates sexlineatus	0.4 (± 5.3)	**1.7 (± 2.1)**	0.04 (± 3.8)	0.2 (± 0.5)	0.7 (± 5.6)	4,150.0
Thryssa aestuaria	5.2 (± 2.4)	5.2 (± 2.2)	0.01 (± 1.2)	5.0 (± 1.2)	**6.1 (± 1.6)**	60,900.0
Urocampus carinirostris	0.0 (± 0.5)	**0.2 (± 0.2)**	0.01 (± 0.5)	**0.2 (± 0.5)**	0.03 (± 0.5)	1,900.0

Table 7.4: Results of nested analyses of variance (ANOVAs) examining differences in the abundance of fish using seagrass in different mosaics. (i) Analysis 1 contrasts two coastal mosaics – Coast-Mangroves and Coast-Urbanised; (ii) Analysis 2 contrasts the other three mosaics – Continuous-Seagrass, Islands-No Mangroves and Islands-Mangroves. Two separate analyses were necessary because of the large variation in the total number of sites sampled in each habitat (Table 7.2). Data were transformed to $\log_e(x+1)$.

		(i) Analysis 1			(ii) Analysis 2		
Variable	Source	df	F	P	df	F	P
# Individuals	Mosaic	1	2.47	0.18	2	1.23	0.31
	Site (M)	5	4.75	0.005	26	7.97	0.001
# Species	Mosaic	1	2.12	0.12	2	0.11	0.90
	Site (M)	5	1.46	0.001	26	4.22	0.001
# *Atherinomorus*	Mosaic	1	0.01	0.98	2	2.06	0.15
ogilbyi	Site (M)	5	1.00	0.44	26	3.39	0.001
# *Ambassis*	Mosaic	1	0.18	0.69	2	0.86	0.44
marianus	Site (M)	5	1.98	0.12	26	0.85	0.67
# *Centropogon*	Mosaic	1	0.94	0.38	2	0.62	0.55
australis	Site (M)	5	9.71	0.001	26	7.64	0.001
# Gobies	Mosaic	1	3.25	0.13	2	1.79	0.19
	Site (M)	5	4.12	0.009	26	7.31	0.001
# *Hyperlophus*	Mosaic	1	0.13	0.73	2	0.32	0.73
translucidus	Site (M)	5	1.15	0.36	26	3.74	0.001
# *Monocanthus*	Mosaic	1	0.52	0.50	2	0.01	0.99
chinensis	Site (M)	5	5.6	0.002	26	11.30	0.001
# *Pelates*	Mosaic	1	2.76	0.16	2	0.48	0.62
sexlineatus	Site (M)	5	4.15	0.009	26	9.47	0.001
# *Urocampus*	Mosaic	1	1.54	0.27	2	0.23	0.92
carinirostris	Site (M)	5	1.43	0.26	26	4.17	0.001
# *Thryssa*	Mosaic	1	0.01	0.93	2	0.01	0.99
	Site (M)	5	3.28	0.02	26	2.47	0.001

Table 7.5: Results of nested analyses of variance (ANOVAs) examining differences in the abundance of fish using unvegetated mudflats in different mosaics. Data were transformed to $\log_e(x+1)$. Note-fewer variables were analysed for the mudflat than for the seagrass (Table 7.4) because of the smaller numbers of individuals that were caught in the unvegetated than vegetated habitat.

Variable	Source	df	F	P
# Individuals	Mosaic	4	1.19	0.33
	Site (M)	37	5.84	0.001
# Species	Mosaic	4	1.48	0.23
	Site (M)	37	3.14	0.001
# *Ambassis*	Mosaic	4	1.62	0.19
	Site (M)	37	1.33	0.13
# Gobies	Mosaic	4	1.60	0.20
	Site (M)	37	9.16	0.001
# *Hyperlophus*	Mosaic	4	1.27	0.30
	Site (M)	37	2.63	0.001
# *Pelates*	Mosaic	4	1.84	0.14
	Site (M)	37	1.74	0.02
# *Thryssa*	Mosaic	4	1.79	0.15
	Site (M)	37	2.67	0.001

Table 7.6: Results of nested analyses of variance (ANOVAs) examining differences in the abundance of different species of gobies using vegetated and unvegetated habitats in two different mosaics (Islands-Mangroves vs Island-No Mangroves). Data were transformed to $\log_e(x+1)$. Gobies were only identified to species for these two mosaics because of the large number of individuals that were present and the difficulties in identifying small individuals that had been preserved (see text for details).

		(i) Vegetated			(ii) Unvegetated		
Variable	Source	df	F	P	df	F	P
Amoya sp.	Mosaic	1	1.74	0.21	1	1.68	0.22
	Site (M)	17	7.51	0.001	14	82.92	0.001
Arenigobius Frenatus *	Mosaic	1	1.46	0.24	-		
	Site (M)	17	8.73	0.001			
Coryphotpterus neophytes	Mosaic	1	1.89	0.19	1	1.49	0.24
	Site (M)	17	21.76	0.001	14	3.84	0.001
Pseudogobius sp.	Mosaic	1	0.35	0.56	1	1.27	0.28
	Site (M)	17	9.83	0.001	14	4.08	0.001

* *Arenigobius frenatus* was only caught in the seagrass habitat

Table 7.7: Results of analyses of covariance (ANCOVA) comparing the abundance and species richness of fish using vegetated habitats in each of five mosaics, for each of 3 different covariates (i) canopy height, (ii) % cover *Zostera capricorni*, (iii) sediment compaction (penetrometer depth). Data were the averages per site for the abundance of fish and habitat characteristics (see text for details). Homogeneity of slopes was tested prior to analysis of any effects of mosaics (Huitema, 1980).

		Covariate	Canopy Height		% Cover-*Zostera*		Compaction	
Variable	Source	df	F	P	F	P	F	P
# Individuals	Mosaic	4	0.59	0.67	1.41	0.36	0.53	0.72
	Covariate	1	8.45	0.01	1.62	0.21	17.01	0.001
	Slopes			0.90		0.37		0.84
# Species	Mosaic	4	1.45	0.24	0.61	0.66	0.32	0.86
	Covariate	1	18.93	0.001	1.57	0.22	6.32	0.02
	Slopes			0.25		0.68		
# *Centropogon australis*	Mosaic	4	0.13	0.97	1.23	0.32	0.69	0.61
	Covariate	1	14.55	0.001	0.90	0.35	14.46	0.001
	Slopes			0.96		0.46		0.48
# Gobies	Mosaic	4	1.88	0.14	0.60	0.67	1.62	0.20
	Covariate	1	0.16	0.69	0.15	0.69	8.95	0.006
	Slopes			0.96		0.56		0.88
# *Monocanthus chinensis*	Mosaic	4	1.99	0.12	2.07	0.11	0.49	0.77
	Covariate	1	37.85	0.001	0.01	0.98	5.48	0.03
	Slopes			0.91		0.84		0.16
# *Pelates sexlineatus*	Mosaic	4	0.17	0.95	0.41	0.80	0.70	0.60
	Covariate	1	24.87	0.001	2.47	0.13	16.03	0.001
	Slopes			0.96		0.50		0.39

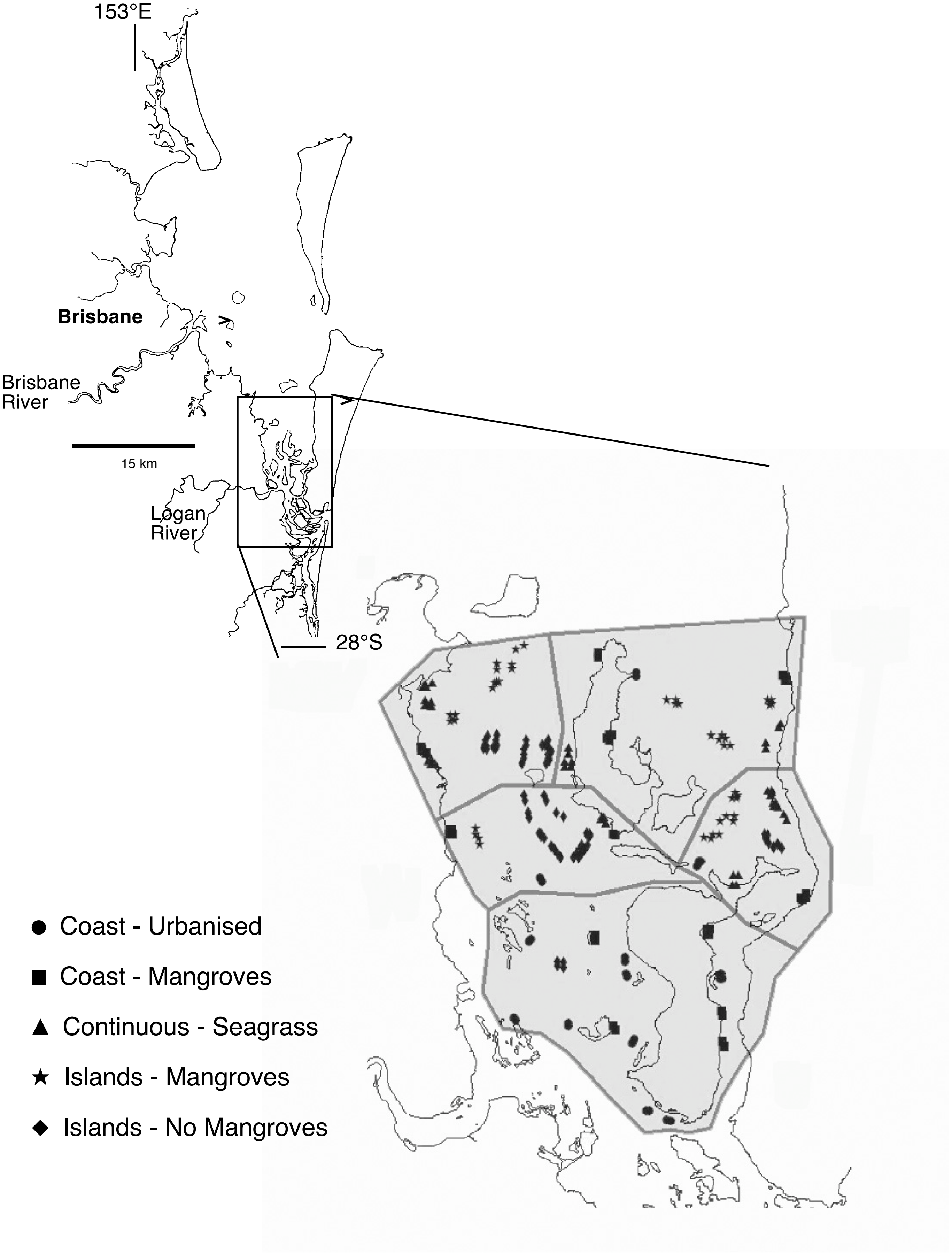


Figure 7.1: Map of Moreton Bay (inset) and the southern region of Moreton Bay, showing the positions of the sites for each of the five mosaics: 'Coast - Mangroves' (squares), 'Coast - Urbanised' (circles), 'Continuous Seagrass' (triangles), 'Islands - No Mangroves' (diamonds) and 'Islands - Mangroves' (stars).

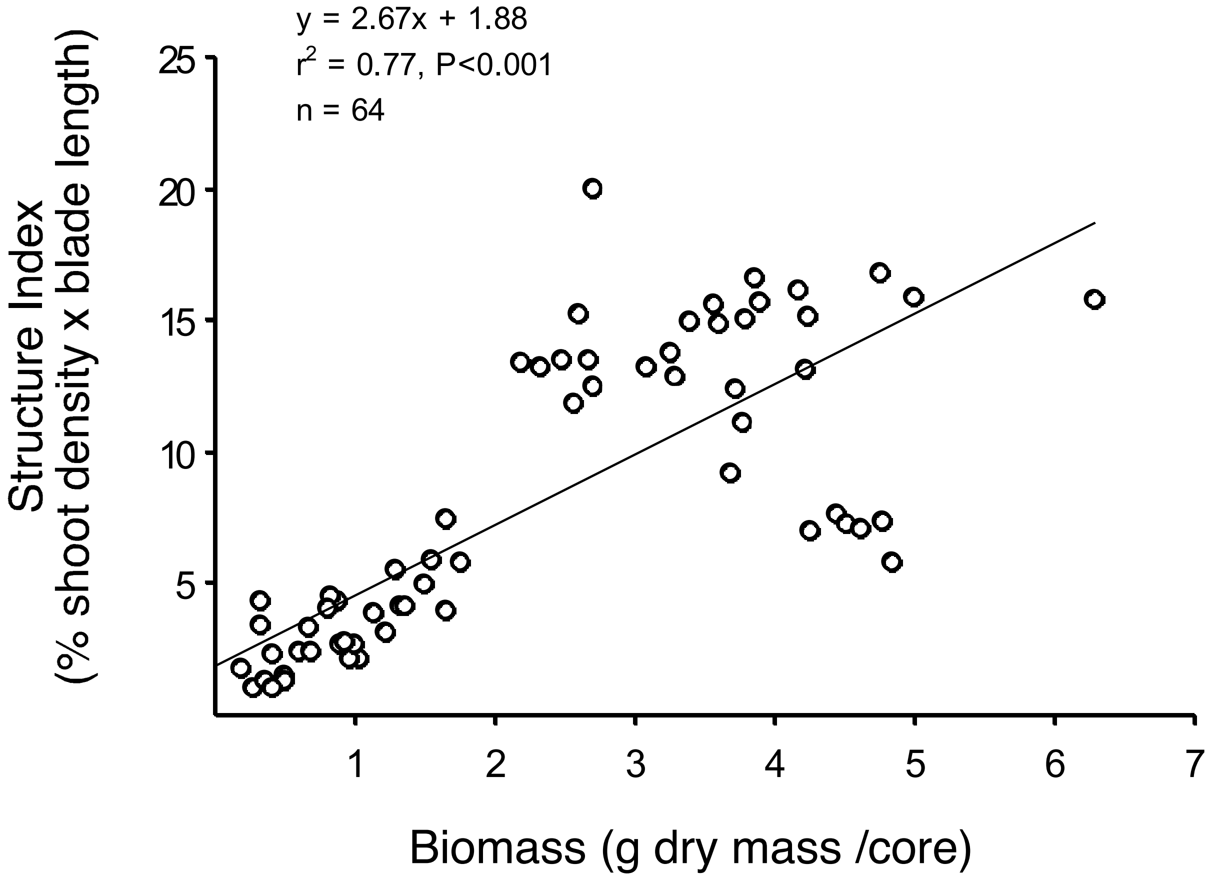


Figure 7.2: The relationship between the Structure Index (product of the mean within plot seagrass shoot density (%) and blade length-mm) and above-ground seagrass biomass density (g DM/core) from replicate cores taken in *Zostera capricorni* beds at 6 sites in southern Moreton Bay.

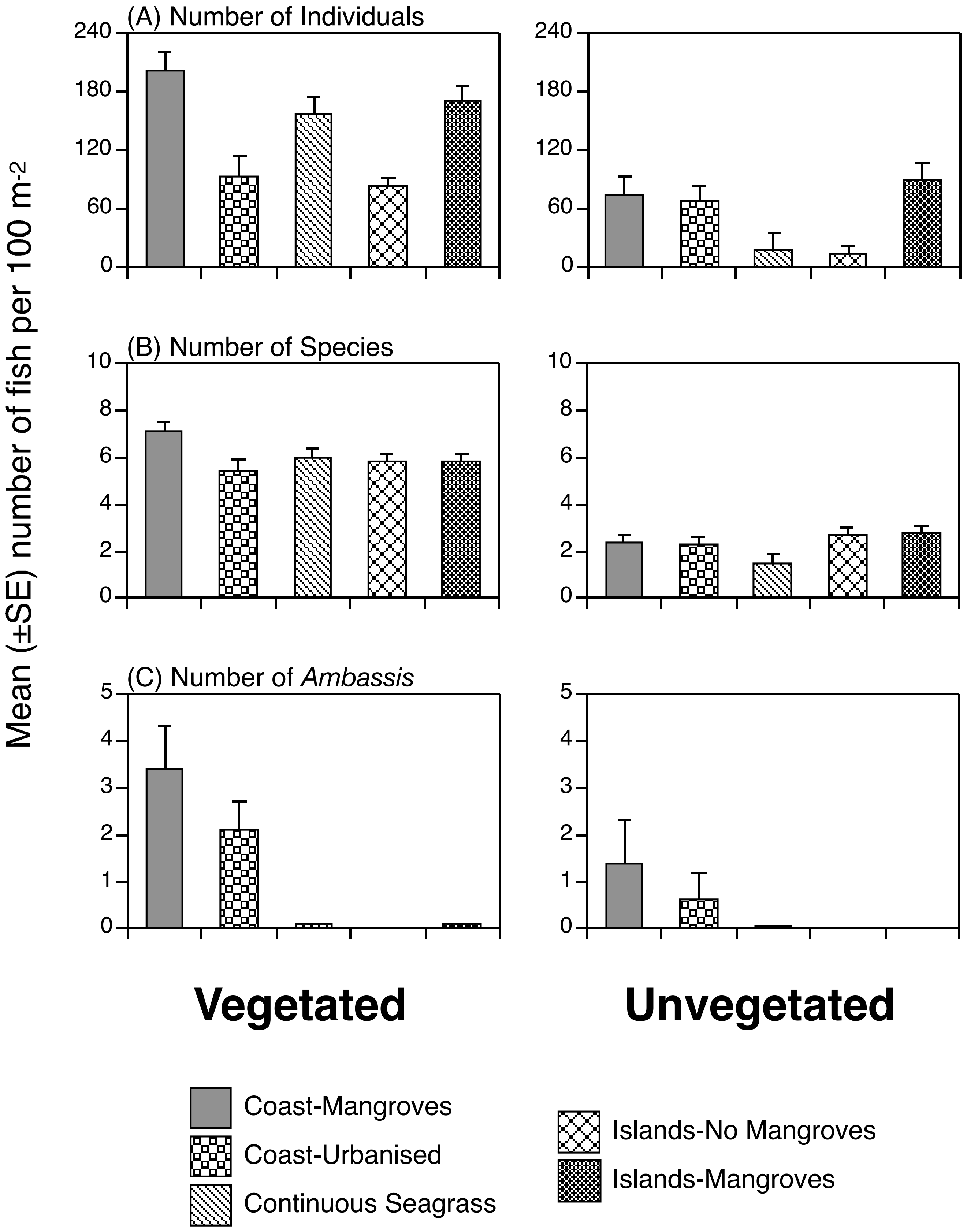


Figure 7.3: Mean (SE) number of (A) fish individuals, (B) fish species and (C) *Ambassis marianus* per 100 m^2 of either vegetated (*Zostera capricorni*) or unvegetated habitat in five different mosaics in southern Moreton Bay, sampled during summer 2005. Note the differences in the scale on the Y-axes.

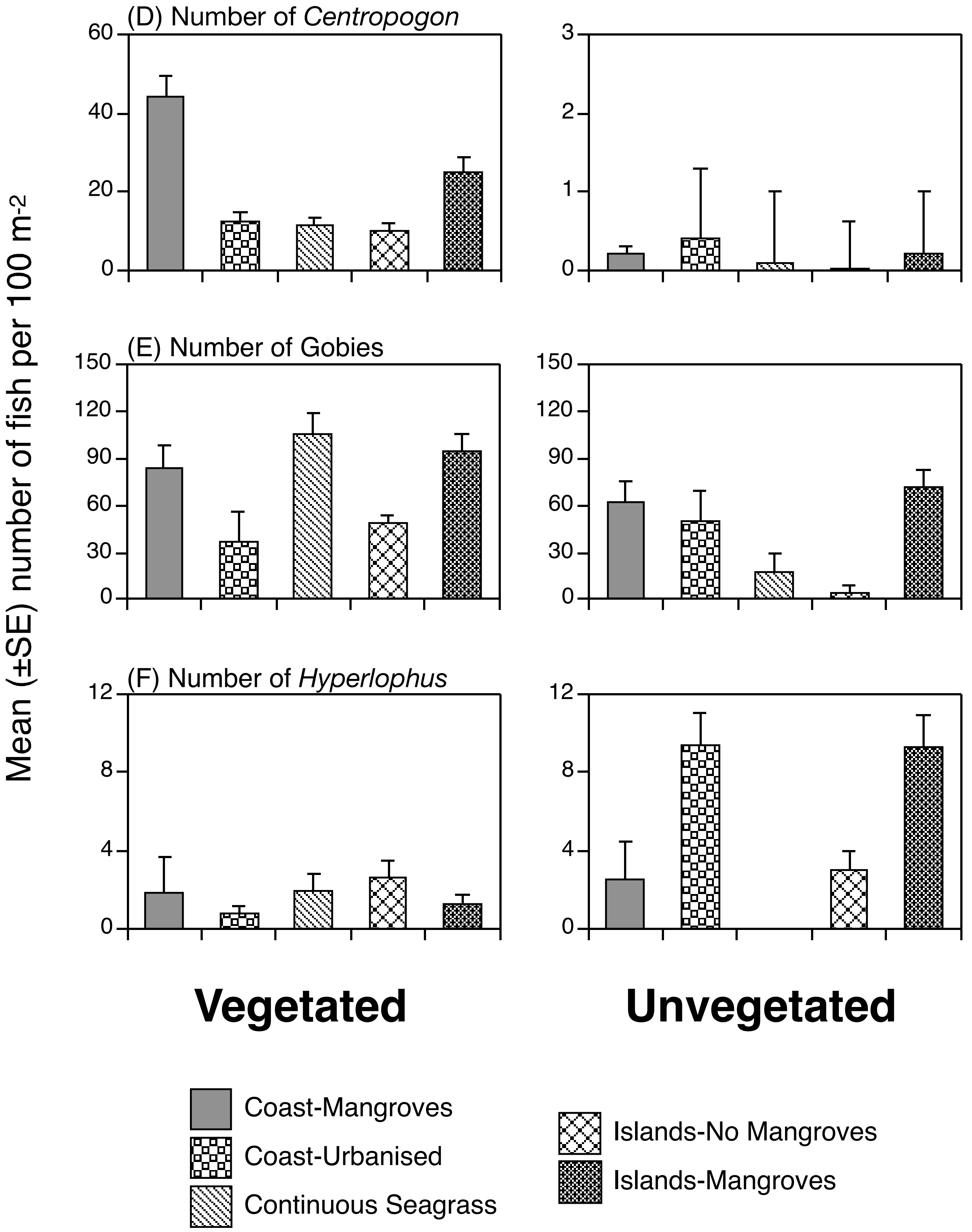


Figure 7.3 cont: Mean (SE) number of (D) *Centropogon australis,* (E) gobies and (F) *Hyperlophus translucidus* per 100 m^2 of either vegetated (*Zostera capricorni*) or unvegetated habitat in five different mosaics in southern Moreton Bay, sampled during summer 2005. Note the differences in the scale on the Y-axes.

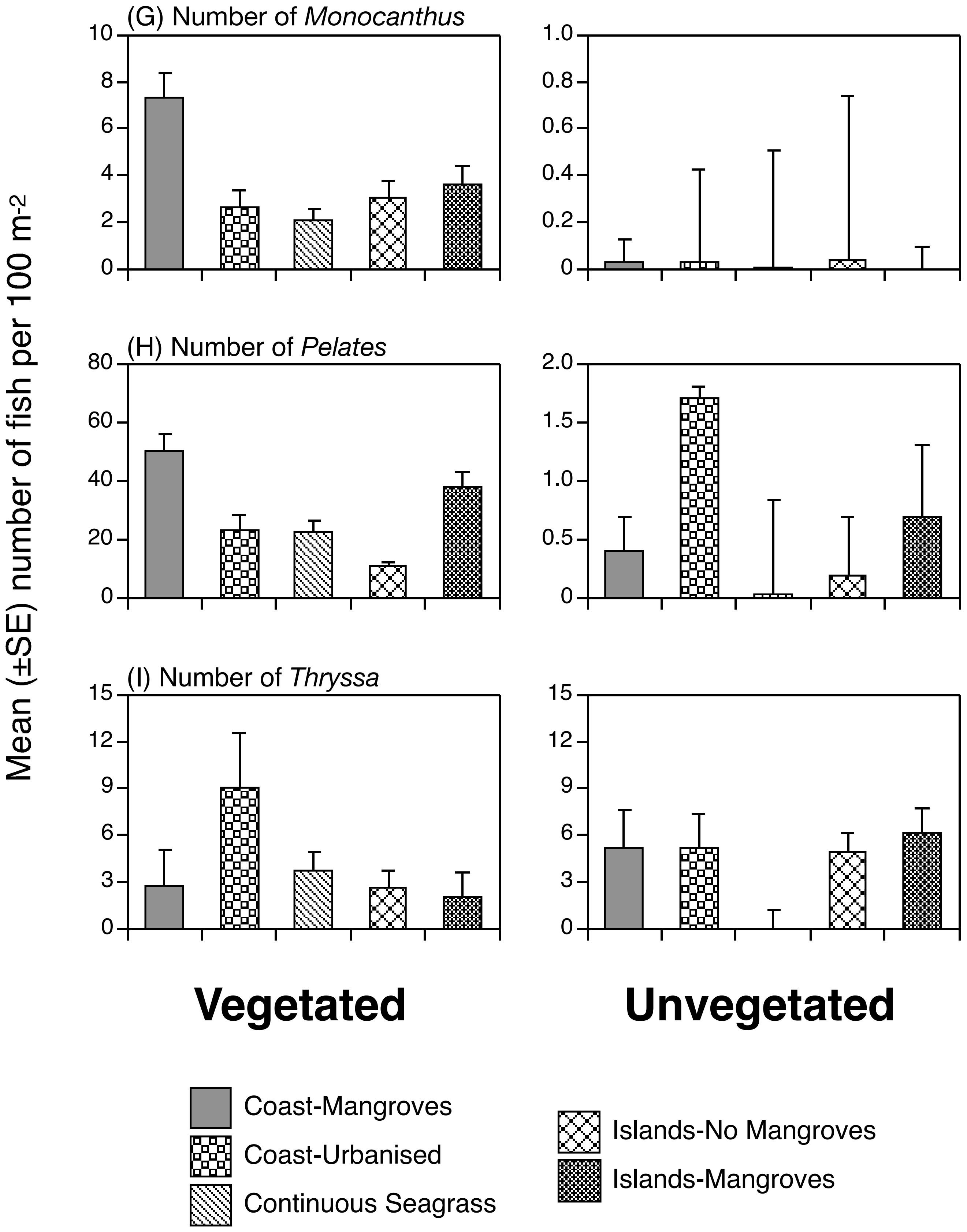
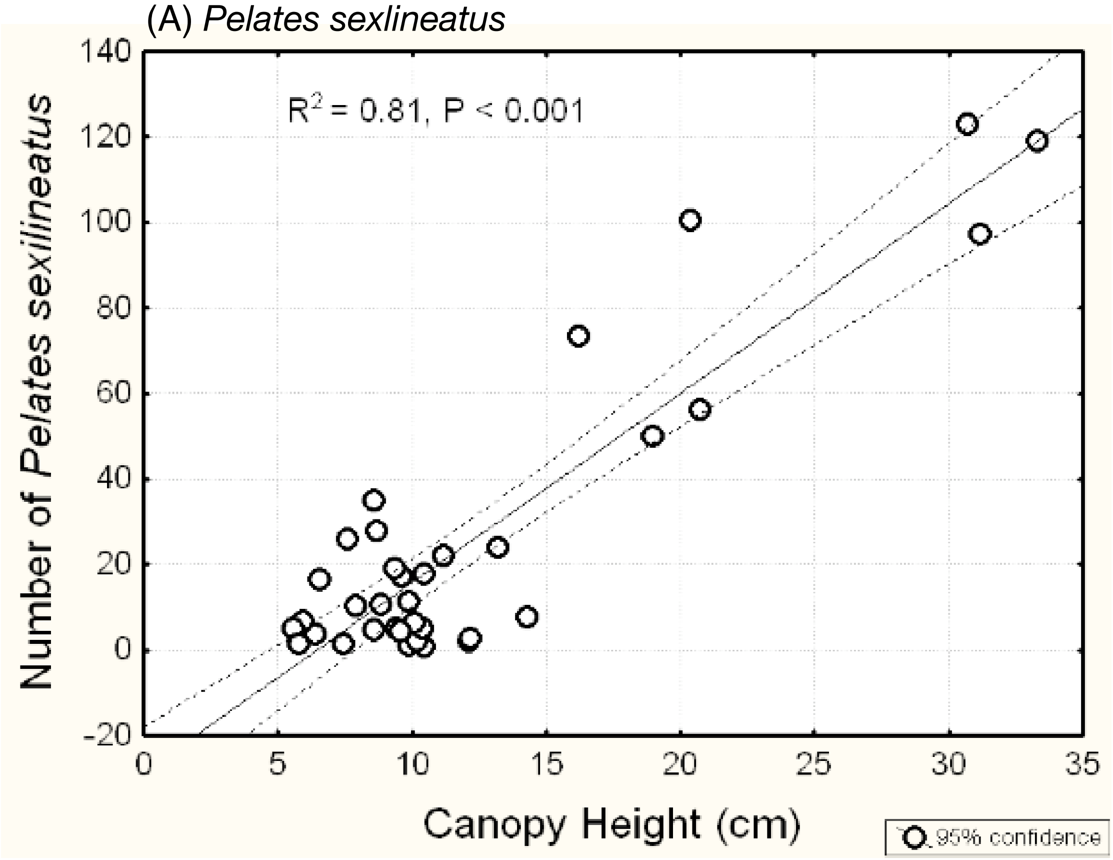
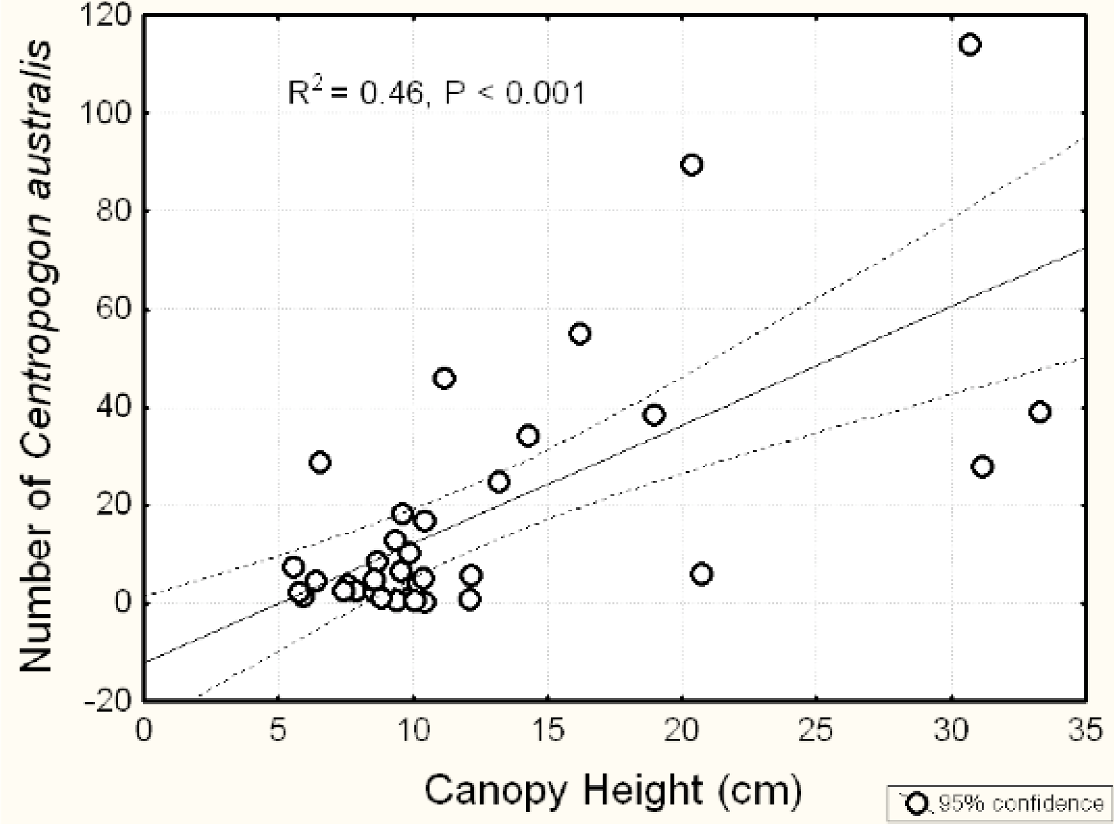


Figure 7.3 cont: Mean (SE) number of (G) *Monocanthus chinensis*, (H) *Pelates sexleaneatus* and (I) *Thryssa aestuaria* per 100 m^2 of either vegetated (*Zostera capricorni*) or unvegetated habitat in five different mosaics in southern Moreton Bay, sampled during summer 2005. Note the differences in the scale on the Y-axes.



(A) *Pelates sexlineatus*



(B) *Centropogon australis*

Figure 7.4: Scatterplots showing the relationship between the abundance of (A) *Pelates sexlineatus* and (B) *Centropogon australis* and the height of the seagrass canopy across all five mosaics in the seagrass habitat. Data are the average abundance of each species of fish per site (n = 4 replicate seines) and the average canopy height of *Zostera capricorni* at that site.

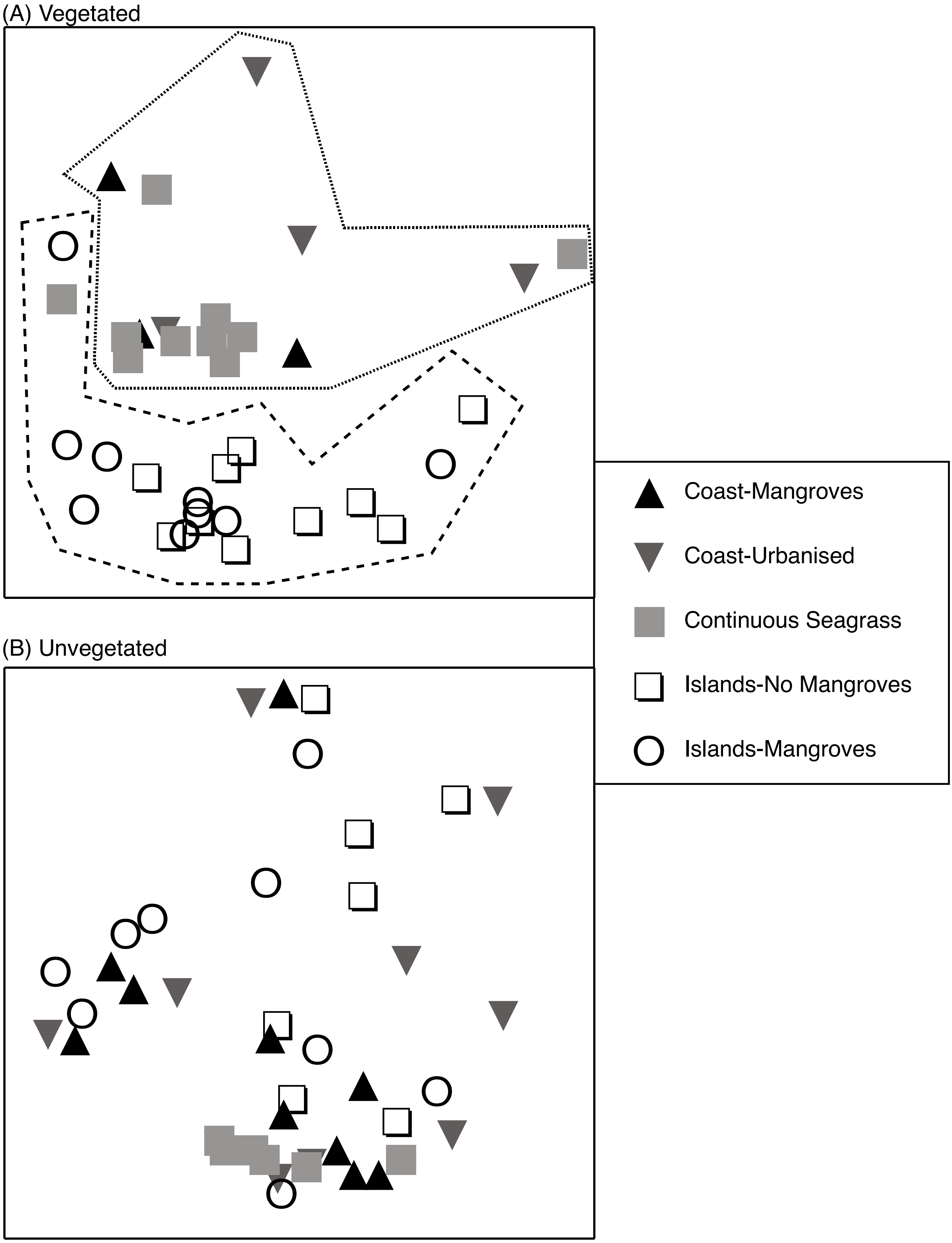


Figure 7.5: nMDS ordinations of untransformed data on the composiition of the fish assemblage utilising (A) intertidal seagrass beds and (B) unvegetated mudflats in five different mosaics in southern Moreton Bay. Data are the site averages for n = 4 replicate seines. Dashed lines represent significantly different groupings of mosaics in ANOSIM.

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Appendix 1

Staff

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2. Ms Bronwyn Cameron – Research Assistant (full time)
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