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The impact of marine reserves on nekton diversity and community composition in subtropical eastern Australia

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ABSTRACT

The effectiveness of marine reserve protection on the biodiversity of aquatic assemblages (i.e. nekton) in subtropical eastern Australia was examined within two small (<6 km^2) marine reserves and four non-reserve areas. The two marine reserves, and their corresponding non-reserves, were located in different geographical locations within Moreton Bay (north and south) and sites were surveyed with multiple hauls of a seine net. Species richness, evenness, density and mean size of the inshore communities were compared between the reserves and non-reserves. No statistical significant difference was detected in species richness between the areas however species evenness was significantly lower in the only non-reserve site impacted by commercial net fishing. Mean size of nekton was found to be significantly greater in the marine reserves compared to non-reserves but no statistical significant difference was found in the density of nekton between the study sites. Multivariate analysis revealed differences in community composition, particularly between the geographical locations where areas were impacted by different types of fishing pressure (recreational v commercial). These results highlight the impact commercial fishing can have on entire nekton assemblages, not just on targeted species. Our study demonstrates that the small marine reserves in Moreton Bay are protecting marine biodiversity and are thus at least partially achieving their management objective (to enhance the zone's marine biodiversity).

1. Introduction

Marine reserves are promoted as an ecosystem management tool which have the capacity to maintain biodiversity by protecting all components of the ecosystem including the protection of unexploited species, habitats and thus the structure and function of ecological communities (Roberts and Hawkins, 2000; Palumbi, 2002; Sobel and Dahlgren, 2004).

Moreton Bay in subtropical eastern Australia is located along a coastal zone of one of the fastest growing regions in Australia. Although it represents only 3% of the Queensland coastline it has the highest recorded abundance and diversity of cetaceans and dugongs in Australia (Chilvers et al., 2005), over 750 fish species have been recorded in the area (Johnson, 1999), it produces 12% of the total volume of commercial fisheries landings (value of $14.3 million in 2003) (CHRIS, 2003)

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and accounts for one-third of the recreational fishing effort in the state (Williams, 2002). Less than 1% of the Moreton Bay Marine Park is currently protected in no-take marine reserves. No-take marine reserves (hence forth referred to simply as marine reserves) are a management technique, which aim to reduce the impacts to marine biodiversity through strict controls on extractive activities (i.e. no extractions permitted) within the designated areas. Under Marine Parks legislation the management objective of the marine reserves in Moreton Bay is 'to provide for the permanent preservation of the zone's biological diversity and natural condition to the greatest possible extent, while allowing the public to appreciate and enjoy the undisturbed nature of the area' (EPA, 1999). Biodiversity is defined as the natural diversity of native wildlife, together with the environmental conditions necessary for their survival, and includes – regional diversity, ecosystem diversity, species diversity and genetic diversity as per the *Nature Conservation Act 1992*.

Species-specific differences between marine reserves and non-reserve 'fished' areas in Moreton Bay have been demonstrated by Pillans et al. (2005) on the exploited mud crab *Scylla serrata* and on a range of exploited finfish species including yellowfin bream *Acanthopagrus australis*, dusky flathead *Platycephalus fuscus* and barred grunter bream *Pomadasys kaakan* (Pillans, 2006). However, no studies have been carried out on the effect of marine reserves in Moreton Bay on overall marine biodiversity to determine if they are meeting their broader biodiversity management objective. In comparison, numerous studies have been carried out worldwide on the effects of marine reserves on marine biodiversity (for example see Jennings et al., 1996; Jennings and Polunin, 1997; Edgar and Barrett, 1999; Lipej et al., 2003; Fraschetti et al., 2005).

Biodiversity has been considered to be one of the main variables that indicates ecosystem health (Magurran, 2004) and the most common metrics of diversity used in studies to assess impacts to marine biodiversity include: species richness, species diversity, species evenness and species dominance (Gray, 2000; Magurran, 2004). Species richness (the number of species) is the most widely used variable in community studies, however it has a series of limitations including being very dependent on sample size (sampling effort). Using this index alone can lead to a loss of information about the relative abundance between the different species (Magurran, 2004). As a complement to species richness, the relationship between the relative abundance of the different species (species evenness) has been incorporated into the analysis of community structure (for example Simpson's index, Smith and Wilson's evenness index) (Magurran, 2004). Evenness is defined as how evenly the total number of individuals of a community are divided between the different species (Magurran, 2004). It has been suggested that multiple measures of diversity should be used in a single study to gain an understanding of how communities are impacted by change (Rice, 2000). Therefore both species richness and species evenness were used in this study followed by a range of multivariate analysis to assess impacts of marine reserve protection on community composition in subtropical eastern Australia.

One of the main objectives of implementing conservation marine reserves is to maintain biodiversity (Ward et al., 2001; Halpern and Warner, 2003) which may lead to potential benefits such as increases in measures of species diversity within reserve boundaries. However, the empirical evidence demonstrating these benefits to marine biodiversity (i.e. increases in species richness and/or diversity) are variable. Increases in species richness/diversity within protected areas is documented in a number of studies: New Caledonia (Wantiez et al., 1997), the Seychelles (Jennings et al., 1996), Florida (Johnson et al., 1999), tropical Australia (Ley et al., 2002) and the Mediterranean (Claudet et al., 2006). However, declines (or no change) in species richness within protected areas is documented in studies in the Philippines (Russ and Alcala, 1989), Fiji (Jennings and Polunin, 1997), Tasmania (Edgar and Barrett, 1999) and Slovenia (Lipej et al., 2003). One of the main problems with the interpretation of community patterns in protected areas is the fact that the expected effect (theoretical) of the protection is not specified.

For example in this study we predict that a successful reserve would generate the following ecological patterns in the nekton community: higher abundance (relative density) and mean size of nekton in the marine reserves compared to the non-reserve 'fished' areas; a more evenly distributed community without a reduction in species richness. An increase in species richness could not be ecologically interpreted as a success of the reserve unless previous historic knowledge of the community indicates that no exotic species have colonized the community. However, an increase in species evenness as a consequence of a reduction in rare species could be interpreted as a failure of the reserve. Intermediate values of these variables in an area that is not protected but is close to the reserve (i.e. boundary non-reserve) would be interpreted as a beneficial effect of the reserve.

As losses of marine biodiversity are highest in coastal areas (Gray, 1997) and Moreton Bay is a coastal area heavily impacted by anthropogenic pressure, it was the objective of this study to assess whether or not the existing marine reserves in Moreton Bay (which were implemented to protect marine biodiversity) were meeting their management objective. Our study aims to assess the effectiveness of two small coastal marine reserves in Moreton Bay by measuring and comparing the diversity and community composition of nekton (i.e. fish and invertebrates) within reserve and non-reserve 'fished' areas. Specifically, we compared a number of biodiversity measures including species richness and evenness of nekton communities between the two reserves and four non-reserves. Total catch of nekton was also compared between reserve and non-reserves to determine if protection had a substantial effect on fish numbers (density) and size. Multivariate analysis using density and habitat/environment data was then used to test for differences between the sites and to identify if protection was the driving force in the patterns detected.

This is the first study to assess the impacts of marine reserves on nekton diversity and community composition in subtropical eastern Australia. The results will demonstrate whether the existing marine reserves are meeting their broader biodiversity management objective, and will provide important baseline information for the future monitoring and establishment of marine reserves in the region.

2. Materials and methods

2.1. Study sites

Surveys were carried out in the Moreton Bay Marine Park (27°25′S, 153°20′E) which is a semi-enclosed subtropical bay covering an area of approximately 3400 km^2 (EPA, 1999) (Fig. 1). The Marine Park itself is more than 35 km wide in the north, tapering to less than 5 km wide at the southern extremity. The two reserve sites surveyed in this study were Tripcony Bight reserve (5.7 km^2 in area) located in the northern extent of the Marine Park and Willes Island reserve (1.9 km^2 in area) located in southern Moreton Bay (Fig. 1c). The marine reserves were established in 1997 and are complete no-take areas managed under one government department (Queensland Parks and Wildlife Service, QPWS). See Pillans et al. (2005) for a detailed description of the biophysical features and management regimes within the study sites.

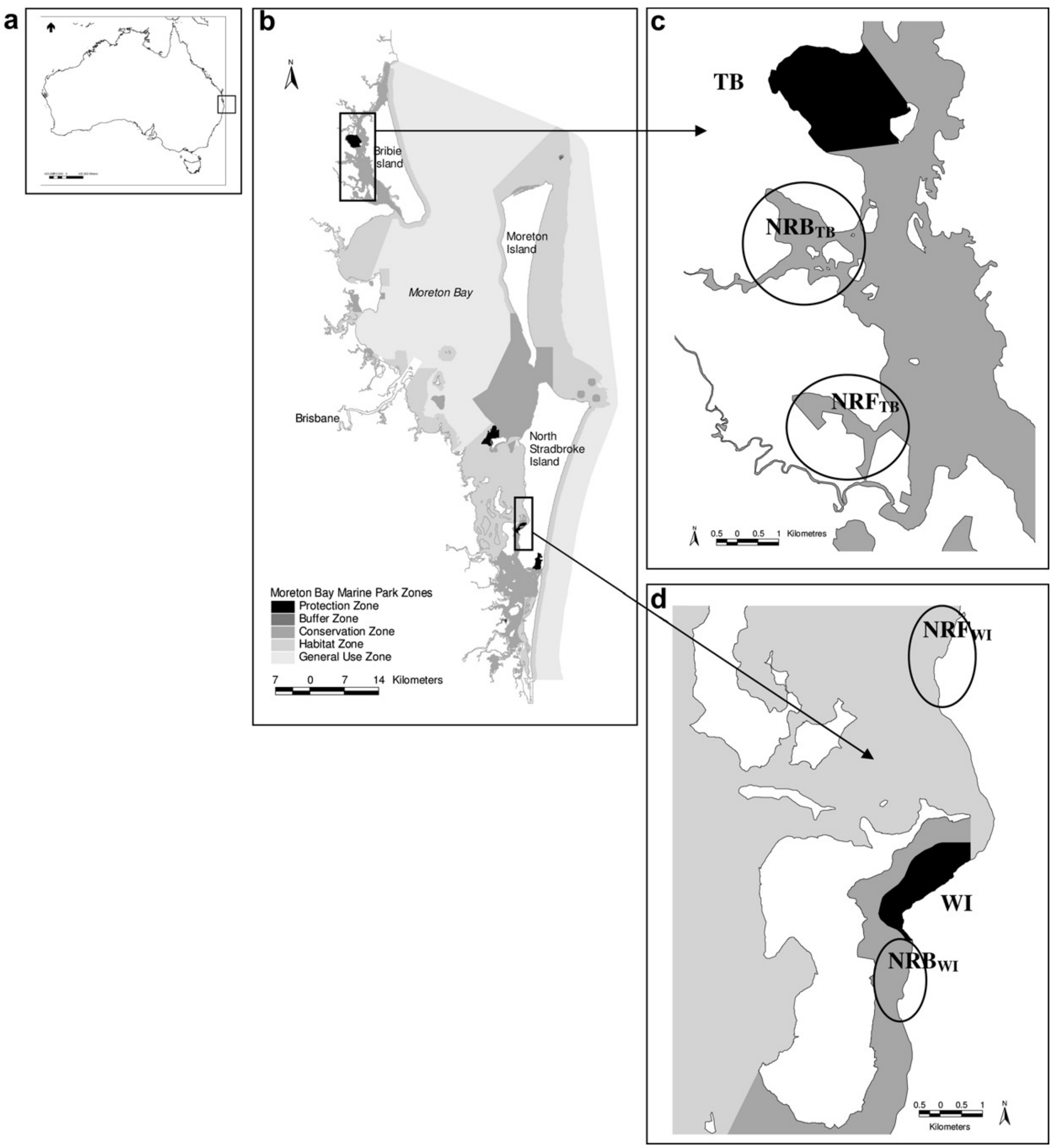

Fig. 1 – (a) Map of Australia, insert shows location of the study area in subtropics. (b) Map of the Moreton Bay Marine Park with zoning scheme, inserts show two marine reserves surveyed in this study. (c) Map of northern study sites: Tripcony Bight reserve (TB) and associated non-reserve sites (NRB$_{TB}$ = Boundary non-reserve, NRF$_{TB}$ = Far non-reserve) (d) Map of southern study sites: Willes Island reserve (WI) and associated non-reserve sites (NRB$_{WI}$ = Boundary non-reserve, NRF$_{WI}$ = Far non-reserve).

2.2. Survey design

As no baseline data was available before the implementation of the marine reserves a true before-after-control-impact (BACI) experimental design could not be carried out (Underwood, 1994). Three different areas were surveyed in this study depending on their management status (protected/unprotected) which included two marine reserves (R), two boundary non-reserves (NRB) and two far non-reserves (NRF). Each marine reserve was compared to two similar non-reserve (fished) areas and non-reserves were chosen on the basis of their proximity to the reserve (boundary or far), habitat similarities (such as habitat and substrate type and vegetation cover) and hydrodynamic conditions (such as flow rates, depth and water chemistry). In terms of their proximity to the reserves the boundary non-reserves were located no more than 2 km north or south of the marine reserve boundary and the far non-reserves were located no more than 7 km north or south of the reserve boundary (Fig. 1c). The two marine reserves surveyed in this study were located in different geographical locations in Moreton Bay (North, South). Tripcony Bight reserve and the corresponding non-reserves were located in the North and Willes Island reserve and the corresponding non-reserves were located in the South (Fig. 1b). Non-reserves located in the North were subject to recreational fishing and the non-reserves located in the South were impacted by both commercial and recreational fishing.

2.3. Field surveys (nekton)

Sampling within fully protected reserves needs to be non-destructive and at the same time needs to maximize the amount of data collected over a range of species (Edgar and Barrett, 1997). As our study areas were located within inshore coastal environments, which are commonly very turbid environments with poor underwater visibility, the use of underwater visual census including baited underwater video techniques, were not possible. We used seine netting to estimate the relative density and size structure of nekton within inshore habitats such as seagrass beds and sand and mud flats. Other common netting techniques, such as gill-netting and trammel netting, were not permitted in this study due to QPWS permit conditions.

Sampling was carried out using a 50 m seine net (dimensions 3 m wide and 12 mm square-mesh netting) with a codend which sampled an area of at least 625 m^2 per haul. Sampling was carried out approximately two hours before the low tide (during the day) and continued throughout the low tide period until the rising tide flooded the shoreline. To ensure sampling was standardized throughout the study seine netting was carried out on the same tidal cycle and moon phase for each of the study sites and hauls were carried out by the same two people for the duration of the study. Total catch was sorted and each animal was identified to species, counted, measured (using total length, cm TL) and released.

Netting was carried out in each study area during summer and winter seasons from 2002 to 2003. Replicated hauls (number dependent on north or south location) were carried out randomly within each of the study sites but at least 100 m apart to minimize pseudoreplication. For example, eight replicated hauls were carried out within each of the study areas located in the North (i.e. Tripcony Bight and two non-reserves) resulting in 96 hauls over the two years of this study. Whereas six replicated hauls (due to the smaller size of the study areas) were carried out within the study sites located in the South (i.e. Willes Island reserve and two non-reserves) resulting in 72 hauls throughout the duration of this study. Haul position was marked on a GPS to ensure that the same sites were resampled throughout the study.

2.4. Habitat surveys

After each haul of the seine net a 30 m tape was set perpendicular to the shore and coverage of submergent vegetation and substrate were assessed by placing a 0.25 m^2 quadrat at every 5 m interval along the transect line (adapted from English et al., 1997). At times the visual census technique had to be adjusted due to the turbidity of the water (poor visibility) and physical sub-samples of the benthic vegetation were necessary to estimate the percentage cover. Salinity, water temperature and average depth of each haul were also recorded at each transect.

2.5. Data analysis

The main components of biodiversity are species richness (number of species) and evenness (how similar species are in their abundances) which are the measures commonly used to assess anthropogenic impacts on marine systems (see Clarke and Warwick, 1994; Gray, 2000; Magurran, 2004). It has been suggested that multiple measures of diversity should be used in a single study to gain an understanding of how communities are impacted by change (Rice, 2000). Therefore, a number of diversity measures and univariate and multivariate analysis were employed in this study.

2.5.1. Univariate statistics

Species richness, evenness, density and mean size: Four dependent variables were analyzed (species richness, evenness, nekton density and mean size) and data were tested for normality (Kolmogorov–Smirnov test) and homogeneity of variance (Levine's Test). In the case of species richness and fish density, data were square root transformed to break the dependence between the mean and the variance of the distributions, whilst mean size data were log transformed.

A four-factor repeated measurements ANOVA was performed on each of the dependent variables with Location treated as a random factor (2 levels = North, South); Status treated as a fixed factor (3 levels = Reserve, Boundary non-reserve, Far non-reserve) with 6–8 replications (hauls) depending on Location; Season treated as a repeated factor (2 levels = Summer, Winter) and Year treated as a repeated factor (2 levels = 2002, 2003). When a significant difference was found in the ANOVA analysis a Least Significant Difference (LSD) multiple comparison test was used to determine which means were different. All univariate statistics were performed using Statistica 7.0 (StatSoft, 2004).

The index E (Bulla, 1994) was used to calculate the evenness of the different samples:

$E = (O - (1/s))/(1 - (1/s))$

where s = number of species, $O = \sum$ minimum between p_i and $1/p_i$ = relative abundance (frequency) of species i. This index was selected because it is relatively independent of sample size, is linear over gradients and is equally sensitive to changes between dominant or rare species (Smith and Wilson, 1996). It has also been suggested as the more adequate index to use when the mechanisms that drive the community assemblage is unknown (Mouillot and Wilson, 2002).

2.5.2. *Multivariate analysis*

Non-metric Multi Dimensional Scaling (nMDS) was used to assess the ordination of the sampling units (hauls) per sampling period in each reserve. Relative abundance of fish species were used as variables and seine net hauls as replicates (Bray Curtis similarity was used as the metric for this analysis). One-way Analysis of Similarity (ANOSIM) was used to test whether the ordination was significant in each sampling period per site, including the post hoc multiple comparisons. Similarity percentages (SIMPER) were used to examine individual contributions by species to any observed patterns in the ordinations (results shown in Appendix A).

A RELATE test for correlating similarity matrices were carried out using the fish abundance similarity matrix (Bray Curtis similarity) and the substrate/environmental variables matrix (Euclidian normalised distance). Following Dietz (1983) suggestion a Kendal nonparametric correlation coefficient was used for the ranked correlation. 999 permutations where used to obtain the significance level of the sample statistic to determine whether the differences in the community structure could have been explained by the differences in the habitat structure. The nMDS, ANOSIM, RELATE and SIMPER analyses were performed using Primer 5.0 (Clarke and Warwick, 1994).

3. Results

3.1. *Total catch*

A total of 168 hauls were carried out during summer and winter seasons from 2002 to 2003 resulting in a total catch of 65708 individuals including 90 fish species and 11 invertebrate species (Appendix B).

3.1.1. *Univariate analysis*

3.1.1.1. *Species richness*. ANOVA results revealed no statistical significant difference in species richness for the factors location, status or year (Table 1). A significant difference was detected in the factor season and for the interaction between season ($p < 0.001$) and status (0.01). Post hoc tests for the interaction between season x status revealed that in summer the reserves were lower in species richness than the boundary non-reserves and boundary non-reserves were also found to be higher in species richness than the far non-reserves during this season. In winter the only significant difference detected was higher species richness in the reserves compared to the far non-reserves (Fig. 2a and b).

3.1.1.2. *Evenness (Bulla, E)*. There were significant differences in evenness detected in the interaction between the factors location × status ($p < 03.02$), the factor year ($p < 0.001$) and between the factors season × status ($p < 0.01$). No significant difference was detected in the factors location, status or season (Table 1). Post hoc tests for the interaction between location x status revealed that only the far non-reserve in the south (NRF_{WI}) was significantly lower than all other sites (Fig. 3a). Evenness was also found to be significantly higher in the northern sites during summer than southern sites during

Table 1 – Univariate analysis

Variable	Effect	df	MS	F	p	Post hoc tests
Species richness	Location	1	0.02	0.12	ns	
	Status	2	0.55	2.46	ns	
	Season	1	16.45	48.27	0.001	
	Season × Status	2	1.69	4.97	0.01	Summer R < NRB, NRB > NRF; Winter R > NRF
	Year	1	0.20	0.78	ns	
Evenness (Bulla)	Location	1	0.03	2.91	ns	
	Status	2	0.01	1.73	ns	
	Location × Status	2	0.04	4.15	0.02	NRF_{WI} < all other sites
	Season	1	0.003	0.24	ns	
	Season × Location	1	0.09	7.17	0.01	Summer TB sites > WI sites; Summer WI sites < Winter WI sites
	Year	1	0.34	30.15	0.001	
Density	Location	1	7.76	0.28	ns	
	Status	2	25.57	0.93	ns	
	Season	1	115.20	2.23	ns	
	Year	1	237.54	4.94	0.03	
Mean size	Location	1	0.04	0.34	ns	
	Status	2	0.51	3.91	0.02	R > NRF; NRB > NRF
	Season	1	2.45	13.98	0.001	
	Year	1	0.08	0.52	ns	

ANOVA results for species richness, evenness (Bulla, E), density and mean size of nekton caught within two marine reserves (Tripcony Bight and Willes Island) and four non-reserves. Location (North TB, South WI); Status (Reserve R, Boundary non-reserve NRB, Far non-reserve NRF; Season (Summer S, Winter W); Year (2002, 2003). Non-significant ($p > 0.05$) interaction terms have been removed from the model.

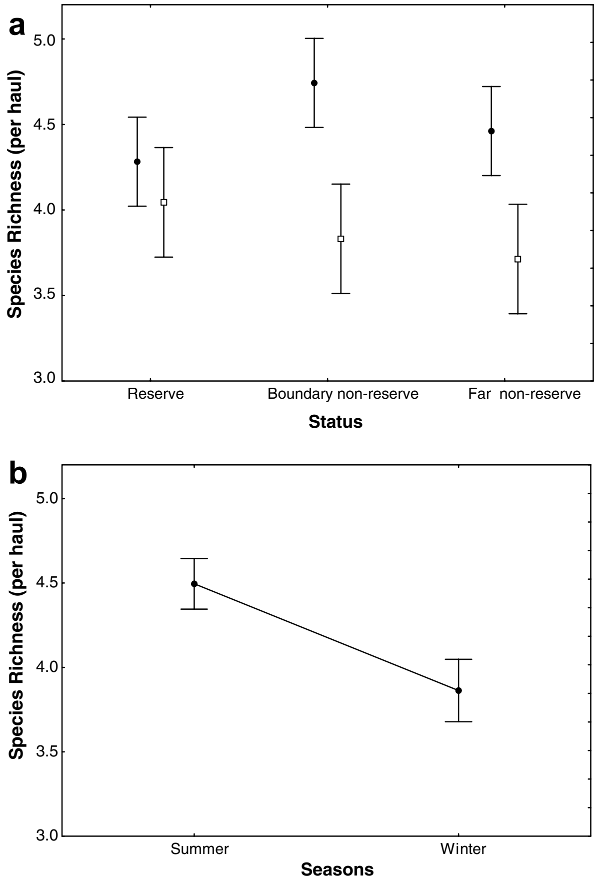

Fig. 2 – Species richness (per haul) + 95%CI of nekton caught within two marine reserves (Tripcony Bight and Willes Island) and four non-reserves for (a) Status (Reserve, Boundary non-reserve, Far non-reserve) × Season (Summer (●) and Winter (□)) and (b) Season (Summer, Winter).

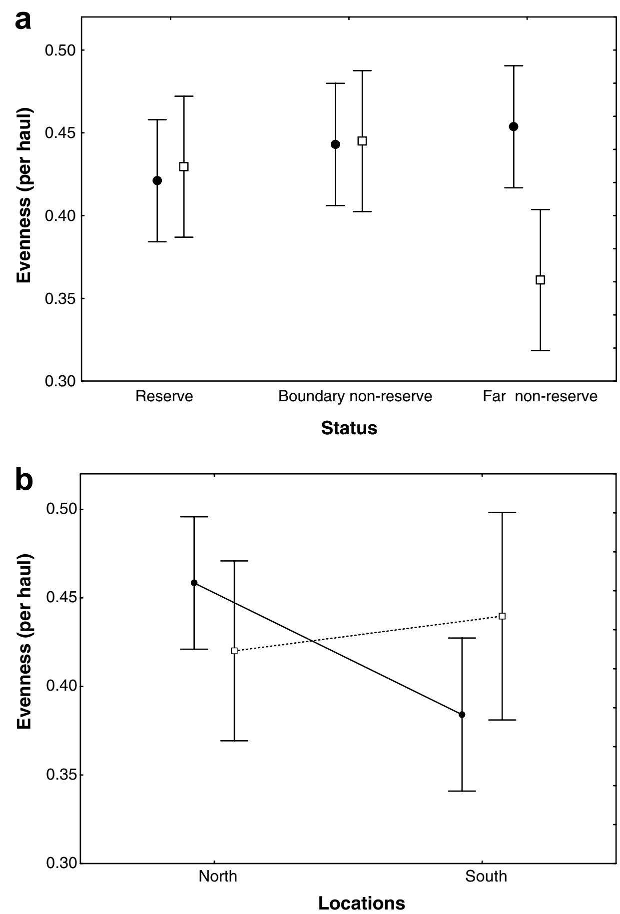

Fig. 3 – Evenness (Bulla, E) (per haul) + 95%CI of nekton caught within two marine reserves (Tripcony Bight and Willes Island) and four non-reserves for (a) Status (Reserve, Boundary non-reserve, Far non-reserve) × Location (North (●) and South (□)) and (b) Location (North, South) × Season (Summer (●) and Winter (□)).

the same season. Evenness within the southern sites was found to be significantly higher in winter compared to summer (Fig. 3b).

3.1.1.3. *Nekton density and mean size*. There was no difference detected in density of nekton between the factors location, status or season, however there was a significant difference detected for the factor year ($p < 0.03$) (Table 1) (Fig. 4). ANOVA results for mean size revealed a significant difference in mean size of nekton for the factors status ($p < 0.02$) and season ($p < 0.001$). No significant difference was detected for the factors location or year (Table 1). Post hoc tests revealed mean size was significantly greater in the reserves compared to the far non-reserves and mean size was significantly greater in the boundary non-reserves compared to the far-non-reserves (Fig. 5a and b).

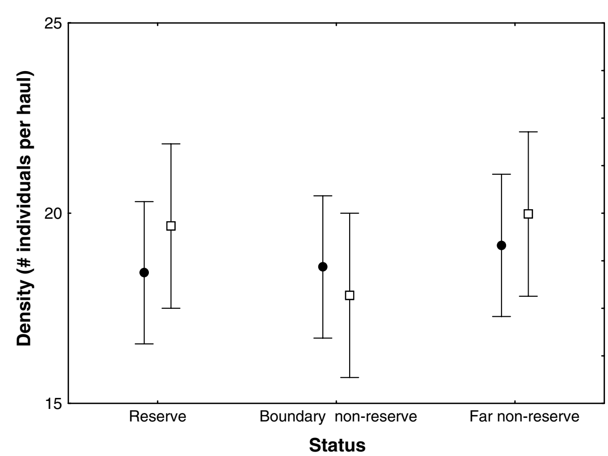

Fig. 4 – Density (# individuals per haul) + 95%CI of nekton caught within two marine reserves (Tripcony Bight and Willes Island) and four non-reserves for Status (Reserve, Boundary non-reserve, Far non-reserve) × Location (North (●) and South (□)).

3.1.2. *Multivariate analysis*

3.1.2.1. *Community structure*. ANOSIM was used to test whether the factor location affected the MDS ordination significantly. In the northern study areas all ordinations were

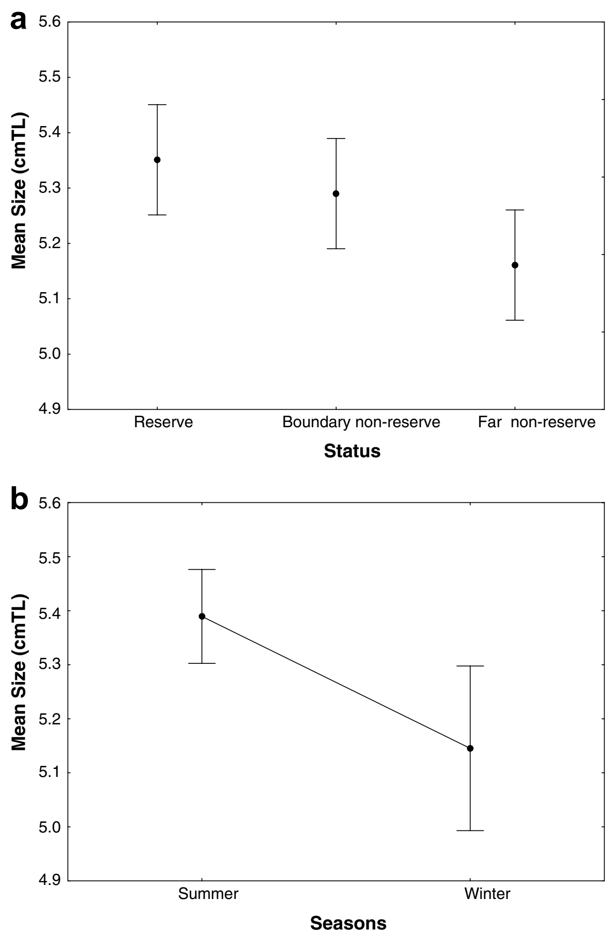

Fig. 5 – Mean size (cm TL) + 95%CI of nekton caught within two marine reserves (Tripcony Bight and Willes Island) and four non-reserves during a. Status (Reserve, Boundary non-reserve, Far non-reserve) and b. Season (Summer, Winter).

found to be significant even though no clear clustering of the sites was observed (Table 2) (Fig. 6). For all the seasons the habitat ordination correlated with the species ordination (RELATE analysis) (Table 3). In the southern areas all species based ordinations were found to be significant, and clear clustering of the sites was observed, particularly for the far non-reserve located in the south (N). This site was clustered together for all time periods and clearly separated from all other sites, particularly in Winter 2003. In the southern areas no correlation was detected with the habitat/environmental matrices RELATE analysis (Table 3). In terms of the pairwise comparisons for the factor status the boundary non-reserves were found to be significantly different from the reserves in all comparisons with the exception of the southern sites in 2002. The far non-reserves were found to be significantly different to the reserves in all cases. Similarly in all but one case the boundary non-reserves were found to be significantly different to the far non-reserves (Table 2).

4. Discussion

Our nekton surveys within the two marine reserves and non-reserve 'fished' areas in Moreton Bay revealed that the existing marine reserves were not higher in species richness or evenness compared to non-reserve sites, however a significant increase in mean size of nekton was detected in the marine reserves – a common benefit (or management intention) of implementing marine reserves (see empirical studies within the following reviews Ward et al., 2001; Halpern, 2003; Sobel and Dahlgren, 2004). Furthermore, differences in community composition were also revealed between the protected and unprotected sites, particularly between the geographical locations where sites were impacted by different types of fishing pressure. Each biodiversity measure used in this study to assess the impacts of marine reserves on nekton diversity and community composition will now be discussed.

4.1. *Species richness*

In general, we found that the marine reserves in Moreton Bay did not have a significant difference in species richness than the non-reserve 'fished' areas. Biodiversity (i.e. species

Table 2 – Multivariate analysis

Location/Time	*R*	*p*	Status	Status	Statistic	*p*	Location/Time	*R*	*p*	Status	Status	Statistic	*p*
North S02	0.219	0.003*	R	B	0.227	0.006	South S02	0.136	0.04*	R	B	0.028	ns
North S02			R	F	0.353	0.007	South S02			R	F	0.209	0.03
North S02			B	F	0.079	ns	South S02			B	F	0.211	0.01
North W02	0.201	0.005*	R	B	0.214	0.02	South W02	0.361	0.001*	R	B	0.103	ns
North W02			R	F	0.186	0.03	South W02			R	F	0.457	0.01
North W02			B	F	0.177	0.04	South W02			B	F	0.524	0.002
North S03	0.361	0.001*	R	B	0.441	0.003	South S03	0.426	0.001*	R	B	0.335	0.01
North S03			R	F	0.334	0.002	South S03			R	F	0.548	0.002
North S03			B	F	0.316	0.001	South S03			B	F	0.496	0.002
North W03	0.211	0.002*	R	B	0.331	0.002	South W03	0.565	0.001*	R	B	0.38	0.01
North W03			R	F	0.178	0.04	South W03			R	F	0.757	0.002
North W03			B	F	0.14	0.03	South W03			B	F	0.613	0.002

ANOSIM results (R values and significance levels) and multiple comparisons of nekton density within two marine reserves (Tripcony Bight and Willes Island) and four non-reserves over two years from 2002 to 2003. Location = North, South; Time = Season (Summer, Winter)/Year (2002, 2003); Status = R = Reserve, *B* = Boundary non-reserve, *F* = Far non-reserve.

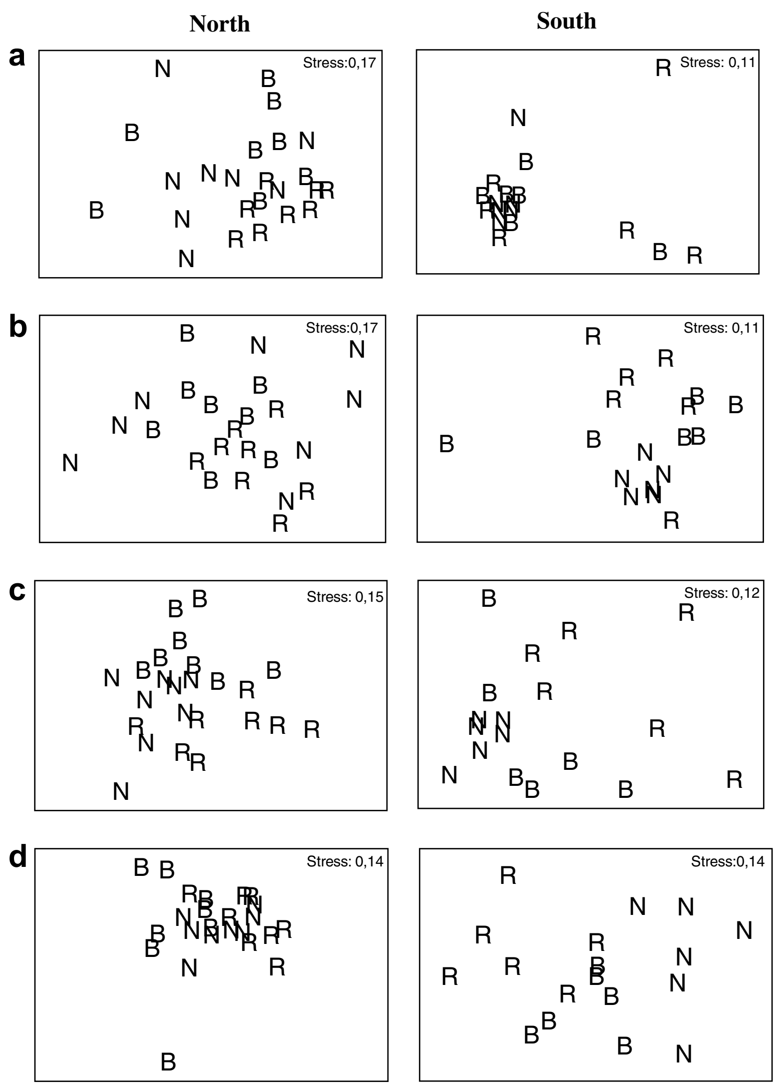

Fig. 6 – MDS ordination of density of nekton from two marine reserves (Tripcony Bight and Willes Island) and four non-reserves during time periods of: (a) Summer 2002, (b) Winter 2002, (c) Summer 2003, (d) Winter 2003 for status *R* = Reserve, *B* = Boundary non-reserve and *N* = Far non-reserve (note that N denotes the far non-reserve site in this ordination only).

Table 3 – Multivariate analysis

Location	Season/Year	Rho	*p*
North	S02	0.432	0.024
North	W02	0.358	0.041
North	S03	0.401	0.016
North	W03	0.298	0.033*
South	S02	0.184	0.251
South	W02	0.058	0.147
South	S03	0.002	0.134
South	W03	0.152	0.091

RELATE Test correlated matrices. Correlation between the ordination of the sites depending on the nekton community and the ordination of the sites depending on the habitat (substrate and vegetation cover) and environmental variables (depth, temperature, salinity) for the two marine reserves (Tripcony Bight and Willes Island) over two years from 2002 to 2003. Location = North, South; Season/Year = Summer and Winter (2002, 2003), * denotes significant value.

richness and diversity) within marine reserves is expected to increase due to the cessation of fishing compared to fished areas (McClanahan, 1994; Jennings et al., 1996; Wantiez et al., 1997; Halpern, 2003; Worm et al., 2006). However, there are contrasting empirical results: in a number of studies species richness was found to decline or not change in the protected areas compared to fished sites, as shown in Fiji (Jennings and Polunin, 1997), Tasmania (Edgar and Barrett, 1999), Slovenia (Lipej et al., 2003) and Italy (Fraschetti et al., 2005). Furthermore, Jennings and Polunin (1997) suggested that diversity does not have to consistently change due to cessation of fishing. In many situations, removal of fishing pressure may cause a decline in species richness because of unpredictable ecosystem changes or because particular species become dominant and exclude others (Barkai and Branch, 1998; Edgar and Barrett, 1999; Ward et al., 2001).

These contrasting results could be due simply to a lack of statistical power to detect differences in biodiversity between areas. This can be caused by large natural variability between sites and the metric(s) used in measuring biodiversity (Price, 2002), as well as the lack of independence between species richness and sampling effort. Additionally, reduction in species richness is a very slow process (unless a big disturbance occurs) therefore species richness is known to be a weak variable for detecting changes in communities (Smith and Wilson, 1996; Magurran, 2004). In our study we did detect differences in species richness between the study areas, but only in regards to seasonal differences. Higher species richness was recorded in summer compared to winter and seasonal patterns in species richness were also detected between the reserves and non-reserves. We expect that these results were due to the seasonal variation commonly exhibited in catches of nekton in subtropical Australia (Blaber and Blaber, 1980; Laegdsgaard and Johnson, 1995).

4.2. Evenness

Using the evenness index, Bulla (E) (Bulla, 1994), we found that species evenness was significantly different between the study areas, with significantly less evenness in the southern far non-reserve (NRF_{WI}) compared to all other sites. It is important to note that each of the marine reserves in this study were located in different geographical locations (north and south) and as such each of the non-reserves experienced different levels of fishing pressure. Tripcony Bight reserve, which is located in Pumicestone Passage, was closed to all forms of commercial fishing in October 1995 becoming the first 'recreational only' fishing estuary in Queensland (O'Neill, 2000). Therefore the non-reserves (NRB_{TB}, NRF_{TB}) associated with Tripcony Bight reserve were impacted by recreational fishing only, in comparison to the non-reserves (NRB_{WI}, NRF_{WI}) associated with the Willes Island reserve which were open to both commercial and recreational fishing.

The non-reserve (NRF_{WI}) which had a much lower species evenness than all other study sites was also the only non-reserve in this study to be impacted by commercial net fishing (gill and tunnel netting). Evenness is an additional critical aspect of biodiversity which is often neglected in ecological studies (Giller et al., 2004). Anthropogenic impacts, such as fishing, can bring about significant changes in evenness without associated changes in richness (Giller et al., 2004). Although the non-reserve (NRF_{WI}) is the only site impacted by commercial net fishing in this study and therefore we cannot draw a strong conclusion, we suggest that the commercial fishing at this non-reserve could be the reason for the dramatic decline in evenness. This highlights the wider impacts commercial fishing (particularly netting) can have on nekton assemblages within shallow-water ecosystems, not just on targeted species, which warrants further investigation in Moreton Bay.

Furthermore, this result could be interpreted as a beneficial effect of the reserve as we predicted that a successful reserve would have a more evenly distributed community without a decrease in species richness. This result suggests that the reserves in Moreton Bay are providing more resilience to the community as communities with high evenness have been described as being more resilient than those with low evenness (Mouillot and Wilson, 2002; Magurran, 2004).

4.3. Density and mean size

Our surveys revealed that the density of nekton within the marine reserves was not significantly different to nekton in non-reserves in Moreton Bay. Studies looking at wider community effects within marine reserves have shown that the density and biomass of non-target species generally do not differ between reserve and fished sites (Williamson et al., 2004; Fraschetti et al., 2005). We did not expect to find significant increases in density of nekton assemblages within the reserves after just five years of protection. We would however, expect rapid increases in the density and mean size of exploited species within the reserves, as shown by Halpern (2003), Pillans et al., 2005) on mud crabs *S. serrata* and Pillans (2006) on various finfish species within the two marine reserves in Moreton Bay. We did not find any differences in the density or mean size of fisheries species between the reserves and non-reserve sites in this study. This could be mainly due to the mean size of fisheries species in this study being less than legal-size, which are not usually targeted by fishers.

However, results of our study did reveal greater mean size of nekton between the marine reserves and far non-reserves. Interestingly, no significant difference in mean size was detected between the reserves and boundary non-reserves. This result could be interpreted as a beneficial effect of the reserves as boundary non-reserves could gain larger nekton through the process of 'spillover' due to their close proximity to the reserves, as we predicted. Seasonal and annual differences in density and mean size of nekton were also detected in this study. For example, greater mean size of nekton was recorded in summer compared to winter seasons. Seasonal differences in catches and mean size of nekton were expected to differ in this study due to seasonal variability in subtropical populations, which could be related to species life-cycles (such as seasonal migrations and shift in habitat requirements etc.) (Blaber and Blaber, 1980).

4.4. Community composition

Differences were detected in the community composition of nekton between the reserves and non-reserves in this study. Patterns revealed by the multivariate analysis demonstrated significant differences in the community structure between the geographical locations. The lack of clear separation between the reserve and non-reserves in the ordination of the northern sites (i.e. Tripcony Bight reserve and non-reserve sites), together with the significant correlation between the habitat/environmental data and nekton community, suggests that the differences in the nekton community could be explained by the habitat/environmental analysis. We propose that the protection provided by the Tripcony Bight reserve has not generated a change in the nekton community structure between the reserve and non-reserves at this location. This result can be interpreted four ways: (1) there is no difference in community composition between the marine reserve and non-reserves at this location, (2) the pressure produced by the recreational fishing at this location is not big enough to generate a significant change in the nekton community structure, (3) the change generated by reserve protection is gradual and five years after implementation may not be enough time to detect this change, or (4) there is an impact but the system is well-mixed so the scale of the study areas (reserves and non-reserves) are too small to see a local impact.

In contrast, clear separation between the southern sites (Willes Island reserve and non-reserve sites) was detected in the MDS, but no correlations were found between this pattern and the habitat/environmental analysis. This suggests that the difference revealed in the southern sites must be attributed to a different variable (factor and/or cause) than in the northern sites. Considering that commercial fishing (particularly netting and trawling) commonly removes more biomass from the marine environment than recreational fishing and these methods target a wide range of species, it is not surprising to find a significant drop in evenness for the far non-reserve site (NRF_{WI}) which is impacted by commercial netting. This pattern could also be attributed to non-target species responding to the cessation of fishing within the marine reserves (Ward et al., 2001) and non-target species responding to fishing disturbance in the non-reserves which could be consistent with the generalized model of disturbance i.e. increased dominance of small, fast-growing species, and general reductions in species diversity and evenness (Hall, 1999; Blanchard et al., 2004). We propose that the changes in the nekton community composition between the southern sites could be a consequence of the protection afforded by the Willes Island reserve.

Our results also highlight the consequences commercial fishing can have on whole communities, not just on targeted species. This observation is consistent with dramatic changes in community structure (shift in trophic levels) due to cessation of commercial fishing have been documented in marine reserves in New Zealand (Babcock et al., 1999; Shears and Babcock, 2002) and on the Great Barrier Reef (Graham et al., 2003).

4.5. *Reserve size*

Although we did not include reserve size as one of the variables tested in our study, it is important to highlight that size of marine reserves can influence the efficacy of marine reserves, particularly those which aim to maintain and enhance marine biodiversity. The small marine reserves in Moreton Bay did provide protection to community composition however no increases in diversity were detected, despite Tripcony Bight reserve being three times larger in size than the Willes Island reserve. Even though a number of factors could have attributed to this result, such as surrounding fishing pressure, lack of statistical power and time necessary to detect such changes, it is unlikely that the small reserves in Moreton Bay are capable of enhancing biodiversity (apart from mean size). It is now well documented that larger marine reserves almost always contain more species than smaller marine reserves as larger reserves can hold larger populations of more species and can include a variety of representative habitats (Halpern, 2003; Hastings and Botsford, 2003; Sale et al., 2005). For a conservation reserve to be effective it needs to be large enough to sustain the populations of interest (Parnell et al., 2005; Salomon et al., 2006) and as such we expect that the reserves in Moreton Bay are too small to have a substantial positive impact on biodiversity.

5. Conclusions

Our surveys of the marine biodiversity within the two marine reserves in Moreton Bay are the first of their kind in subtropical eastern Australia. They provide important baseline data for managers and planners of the Marine Park. Although the marine reserves in Moreton Bay did not show statistically significant increases in species richness or density we did detect a significant increase in evenness and mean size of nekton in the marine reserves compared to non-reserves. Based on our earlier predictions we consider that because the marine reserves have: a more evenly distributed community without decreases in species richness, a greater mean size of nekton within the reserves and boundary non-reserves, and a different community structure than non-reserves, our results demonstrate that the small marine reserves in Moreton Bay are successful in protecting some aspects of marine biodiversity. However, the effects were subtle and larger marine reserve systems are recommended as they commonly protect a broader range and coverage of habitats and can therefore contain more species increasing the potential for marine reserves to be self-sustaining. Our results also highlight the importance of using multiple measures of diversity and community structure to detect changes in protected areas, as without the use of the metric evenness and the multivariate analysis used in our study we would not have detected any statistically significant differences between the marine reserves and non-reserve sites in Moreton Bay.

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Appendix A

Multivariate analysis. Results of SIMPER analysis on the density of nekton caught within two marine reserves (Tripcony Bight and Willes Island) and four non-reserves over two years from 2002 to 2003. Location = North, South; Time periods = Summer and Winter (2002, 2003); Status = R: Reserves, B: Boundary non-reserve, F: Far non-reserve. Only includes species that contributed up to and including 70% of the total catch in each of the study sites.

Appendix A – *continued*

Location	Time	Status	Average similarity	Species	Average abundance	Average similarity	Contribution (%)	Cumulative (%)
North	S02	R	54.43	*Pelates sexlineatus*	48.14	36.76	67.59	67.59
				Gerres subfasciatus	7.5	4.04	7.43	75.02
		B	32.46	*Pelates sexlineatus*	19.04	8.94	27.55	27.55
				Ambassis marianus	14.5	5.88	18.1	45.66
				Gerres subfasciatus	12.37	3.75	11.56	57.22
				Siganus fuscescens	4.17	1.76	5.42	62.64
				Selenotoca multifasciata	3.6	1.61	4.97	67.61
				Monacanthus chinensis	4.0	1.45	4.47	72.08
		F	36.86	*Gerres subfasciatus*	17.3	9.91	26.88	26.88
				Pelates sexlineatus	15.14	7.89	21.41	48.28
				Sillago maculata	9.33	3.83	10.39	58.67
				Siganus fuscescens	6.28	2.64	7.15	65.83
				Hyporhampus r.ardelio	5.34	2.12	5.74	71.57
North	W02	R	39.48	*Gerres subfasciatus*	38.23	23.41	59.31	59.31
				Sillago maculata	9.26	2.98	7.55	66.86
				Stolephorus indicus	6.55	2.97	7.53	74.39
		B	31.68	*Gerres subfasciatus*	12.62	7.24	22.84	22.84
				Stolephorus indicus	12.56	4.06	12.81	35.65
				Arrhampus sclerolepis	6.23	3.85	12.16	47.81
				Mugil georgii	16.63	3.8	12.01	59.82
				Sillago maculata	4.81	2.8	8.83	68.66
				Herklotsichthys castelnaui	10.39	2.56	8.1	76.75
		F	17.93	*Pelates sexlineatus*	19.71	4.78	26.63	26.63
				Gerres subfasciatus	14.97	3.49	19.45	46.08
				Tetractenos hamiltoni	10.61	2.0	11.15	57.24
				Atherinomorus ogilbyi	14.67	1.75	9.75	66.99
				Arrhampus sclerolepis	8.05	1.14	6.35	73.33
North	S03	R	46.56	*Pelates sexlineatus*	12.78	9.25	19.88	19.88
				Herklotsichthys castelnaui	8.81	5.4	11.6	31.48
				Monacanthus chinensis	5.6	4.28	9.2	40.67
				Ambassis marianus	10.78	3.41	7.32	47.99
				Atherinomorus ogilbyi	7.58	3.14	6.75	54.74
				Monodactylus argenteus	8.49	2.96	6.35	61.09
				Penaeus esculentus	4.05	2.93	6.29	67.38
				Hyporhampus r. ardelio	8.49	2.79	5.98	73.36
		B	54.24	*Pelates sexlineatus*	15.22	12.59	23.21	23.21
				Metapenaeus bennettae	16.38	9.62	17.73	40.94
				Siganus fuscescens	9.89	6.27	11.56	52.5
				Monacanthus chinensis	5.33	4.06	7.49	59.99
				Tetractenos hamiltoni	5.85	3.79	6.99	66.98
				Gerres subfasciatus	5.19	3.54	6.53	73.52
		F	53.68	*Gerres subfasciatus*	15.3	12.93	24.09	24.09
				Pelates sexlineatus	12.32	9.11	16.98	41.07
				Herklotsichthys castelnaui	9.13	6.78	12.63	53.7
				Siganus fuscescens	8.94	5.29	9.85	63.54
				Monacanthus chinensis	4.62	3.72	6.93	70.48
North	W03	R	48.42	*Pelates sexlineatus*	41.92	30.52	63.03	63.03
				Gerres subfasciatus	9.52	6.0	12.38	75.42
		B	31.86	*Gerres subfasciatus*	20.4	13.66	42.89	42.89
				Sillago maculata	9.57	4.73	14.84	57.74
				Pelates sexlineatus	11.48	3.8	11.91	69.65
				Ambassis marianus	13.45	3.03	9.53	79.18
		F	49.02	*Pelates sexlineatus*	32.7	21.29	43.42	43.42
				Gerres subfasciatus	16.36	10.56	21.55	64.97
				Sillago maculata	8.8	4.85	9.9	74.87
South	S02	R	28.36	*Pelates sexlineatus*	30.12	12.44	43.86	43.86
				Herklotsichthys castelnaui	18.25	3.82	13.48	57.34

(continued on next page)

Appendix A – *continued*

Location	Time	Status	Average similarity	Species	Average abundance	Average similarity	Contribution (%)	Cumulative (%)
				Atherinomorus ogilbyi	21.14	3.43	12.08	69.41
				Monacanthus chinensis	4.89	2.67	9.43	78.84
		B	47.61	*Pelates sexlineatus*	34.92	25.83	54.25	54.25
				Penaeus esculentus	7.25	4.87	10.24	64.49
				Monacanthus chinensis	6.67	3.86	8.1	72.59
		F	65.94	*Pelates sexlineatus*	56.37	45.47	68.96	68.96
				Hyporhampus r. ardelio	16.68	7.19	10.9	79.86
South	W02	R	44.82	*Pelates sexlineatus*	26.98	15.04	33.56	33.56
				Gerres subfasciatus	13.93	7.15	15.95	49.51
				Mugil georgii	13.01	6.98	15.56	65.07
				Monacanthus chinensis	8.61	5.42	12.1	77.17
		B	42.3	*Gerres subfasciatus*	18.09	12.13	28.69	28.69
				Pelates sexlineatus	16.69	9.34	22.08	50.76
				Monacanthus chinensis	10.76	6.72	15.89	66.65
				Tetractenos hamiltoni	5.78	2.92	6.91	73.57
		F	72.52	*Pelates sexlineatus*	47.11	41.03	56.58	56.58
				Monacanthus chinensis	20.26	15.8	21.79	78.37
South	S03	R	49.6	*Gerres subfasciatus*	21.32	12.38	24.95	24.95
				Pelates sexlineatus	16.28	11.31	22.8	47.75
				Mugil georgii	8.68	4.99	10.06	57.81
				Arrhampus sclerolepis	7.22	3.29	6.63	64.45
				Sillago maculata	4.58	3.29	6.63	71.07
		B	57.21	*Pelates sexlineatus*	31.86	27.21	47.57	47.75
				Monacanthus chinensis	9.21	7.38	12.89	60.46
				Hyporhampus r. ardelio	11.36	5.74	10.03	70.49
		F	50.65	*Atherinomorus ogilbyi*	31.69	19.34	38.19	38.19
				Monacanthus chinensis	20.15	12.01	23.72	61.91
				Hyporhampus r. ardelio	13.6	7.19	14.2	76.11
South	W03	R	41.55	*Leiognathus decorus*	16.51	7.94	19.1	19.1
				Pelates sexlineatus	9.63	6.77	16.29	35.39
				Gerres subfasciatus	8.95	4.82	11.59	59.13
				Herklotsichthys castelnaui	9.48	2.23	5.37	64.5
				Penaeus esculentus	2.82	1.89	4.54	69.03
				Metapenaeus bennettae	3.71	1.75	4.22	73.26
		B	50.19	*Pelates sexlineatus*	20.43	16.32	32.53	32.53
				Monacanthus chinensis	13.62	9.65	19.23	51.76
				Centropogon australis	7.28	4.74	9.45	61.2
				Penaeus esculentus	8.71	3.86	7.69	68.9
				Gerres subfasciatus	9.13	3.18	6.34	75.24
		F	72.3	*Pelates sexlineatus*	38.3	31.17	43.12	43.12
				Hyporhampus r. ardelio	16.04	12.76	17.65	60.77
				Monacanthus chinensis	8.46	6.07	8.4	69.16
				Gerres subfasciatus	8.66	5.9	8.17	77.33

Appendix B

Summary of total catch of nekton caught in two marine reserves (Tripcony Bight and Willes Island) and four non-reserves from seine net surveys carried out during summer and winter from 2002 to 2003. Species recorded in terms of their presence or absence within the reserves (R) and non-reserves (NR = boundary and far non-reserves combined) depending on their location in Moreton Bay (North and South).

Common name	Species	North		South	
		R	NR	R	NR
Eastern anchovy	*Thryssa aestuaria*	×			×
Australian anchovy	*Engraulis australis*			×	
Indian anchovy	*Stolephorus indicus*	×	×	×	×
Yellow perchlet	*Ambassis marianus*	×	×	×	×

Appendix B – *continued*

Common name	Species	North		South	
		R	NR	R	NR
Two-eyed cardinalfish	*Apogon nigripinnis*				×
Pink-breasted siphonfish	*Siphamia rosiegaster*		×	×	×
Striped Silver siphonfish	*Apogon fasciata*	×	×	×	×
Blue-faced whiptail	*Pentapodus paradiseus*				×
Great barracuda	*Sphyraena barracuda*		×	×	
Pick-handled barracuda	*Sphyraena jello*			×	×
Blue-spotted sabretooth blenny	*Petroscirtes lupus*	×	×	×	×
Brown blubberlip	*Plectorhinchus gibbosus*		×		
Yellowfin bream	*Acanthopagrus australis*	×	×	×	×
Diamond fish (butterbream)	*Monodactylus argenteus*	×	×	×	×
Butterfly fish (stripey)	*Microcanthus strigatus*	×	×	×	×
Barred estuarine shrimp	*Palaemon serrifer*	×	×		
Striped catfish	*Plotosus lineatus*		×		
Estuary cod	*Epinephelus coioides*		×	×	
Smooth-handed crab	*Pilumnopeus serratifrons*		×	×	×
Mud crab	*Scylla serrata*	×	×	×	×
Sand crab	*Portunus pelagicus*	×	×	×	×
Cuttlefish	*Sepia sp.*		×	×	×
Double-ended pipefish	*Syngnathoides biaculeatus*	×	×	×	×
Estuarine stinkfish	*Callionymus macdonaldi*			×	×
Large-toothed flounder	*Pseudorhombus jenynsii*	×	×	×	×
Dusky flathead	*Platycephalus fuscus*	×	×	×	×
Fringe-eyed flathead	*Cymbacephalus nematophthalmus*	×	×	×	×
Smooth flutemouth	*Fistularia commersonii*		×	×	×
Barred fortesque	*Centropogon australis*	×	×	×	×
Fortesque	*Bathyaploactis curtisensis*		×	×	
River garfish	*Hyporhampus regularis ardelio*	×	×	×	×
Snub-nosed garfish	*Arrhampus sclerolepis*	×	×	×	×
Bar-tailed goatfish	*Upeneus tragula*		×	×	×
Goby	*Amoya sp.*	×	×	×	×
Shoulder-spot goby	*Acentrogobius caninus*				×
Grassy sweetlip	*Lethrinus laticaudis*		×	×	×
Large-scaled grinner	*Saurida undosquamis*				×
Gudgeon	*Psammogobius biocellatus*		×	×	×
Happy moments	*Siganus fuscescens*	×	×	×	×
White-spotted rabbitfish	*Siganus canaliculatus*			×	
Ogilby's hardyhead	*Atherinomorus vaigiensis*	×	×	×	×
Southern herring	*Herklotsichthys castelnaui*	×	×	×	×
Giant herring	*Elops hawaiensis*	×			
Hairy pipefish	*Urocampus carinirostris*	×	×		×
Fan-bellied leatherjacket	*Monacanthus chinensis*	×	×	×	×
Six-spine leatherjacket	*Meuschenia trachylepis*	×	×	×	×
Red lionfish	*Pterois volitans*		×	×	
Crocodile longtom	*Tylosurus crocodilus*	×	×	×	×
Stout longtom	*Tylosurus gavialoides*				×
Luderick	*Girella tricuspidata*	×		×	×
Moses perch	*Lutjanus russelli*	×	×	×	×
Mangrove jack	*Lutjanus argentimaculatus*			×	
Diamond-scale mullet	*Liza vaigiensis*		×	×	×
Fan-tailed mullet	*Liza subviridis*		×	×	×
Poddy mullet	*Valamugil georgii*	×	×	×	×
Sea mullet	*Mugil cephalus*	×	×	×	×
Tiger mullet	*Liza argentea*			×	
Nudibranch	*Dendrodoris rainfordi*			×	
Blue-ringed octopus	*Hapalochlaena fasciata*			×	
Blue-spotted parrotfish	*Leptoscarus vaigiensis*		×	×	
Blue-barred parrotfish	*Scarus ghobban*				×
Yellowtail pike	*Sphyraena obtusata*	×	×	×	×
Black-naped ponyfish	*Leiognathus decorus*	×	×	×	×
Long-spined porcupinefish	*Cyclichthys jaculiferus*			×	
Bay prawn	*Metapenaeus bennettae*	×	×	×	×

(continued on next page)

Appendix B – *continued*

Common name	Species	North		South	
		R	NR	R	NR
Slender seagrass shrimp	*Latreutes pymoeus*	×			×
Tiger prawn	*Penaeus esculentus*	×	×	×	×
Double-spotted queenfish	*Scomberoides lysan*	×	×	×	×
Queenfish	*Scomberoides sp.*				×
Australian butterfly ray	*Gymnura australis*		×		
Blue-spotted mask ray	*Dasyatis kuhlii*	×	×	×	×
Cowtail ray	*Pastinachus sephen*	×			
Eastern shovelnosed ray	*Aptychotrema rostrata*	×	×		×
Estuary stingray	*Dasyatis fluviorum*	×	×	×	×
White-spotted stingray	*Himantura toshi*	×	×	×	
Brown whipray (species A)	*Himantura* sp. A				×
Reticulate whipray	*Himantura uarnak*		×		
Pig-eye shark	*Carcharinus amboinensis*	×			
Common silverbiddy	*Gerres subfasciatus*	×	×	×	×
Whipfin silverbiddy	*Gerres filamentosus*		×		
Striped butterfish	*Selenotoca multifasciata*	×	×	×	
Seahorse	*Hippocampus procerus*			×	×
Broad squid	*Photololigo etheridgei*	×	×	×	×
Surgeon fish	*Acanthurus sp.*			×	×
Tailor	*Pomatomus saltatrix*		×		
Tarwhine	*Rhabdosargus sarba*	×	×	×	×
Seven-fingered threadfin	*Polydactylus multiradiatus*	×			
Stars and stripes pufferfish	*Arothron hispidus*	×	×	×	×
Common toadfish	*Tetractenos hamiltoni*	×	×	×	×
Banded toadfish	*Marilyna pleurosticta*	×	×	×	×
Narrow-lined toadfish	*Arothron manilensis*		×	×	×
Weeping toadfish	*Torquigener pleurogramma*		×	×	
Big-eye trevally	*Caranx sexfasciatus*				×
Diamond trevally	*Alectis indicus*				×
Giant trevally	*Caranx ignobilis*	×		×	×
Onion trevally	*Carangoides caeruleopinnatus*	×			
Yellowfin tripodfish	*Tripodichthys angustifrons*	×	×	×	×
Eastern trumpeter	*Pelates sexlineatus*	×	×	×	×
Golden-lined whiting	*Sillago analis*	×	×	×	×
Summer whiting	*Sillago ciliata*	×	×	×	×
Winter whiting	*Sillago maculata*	×	×	×	×

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