2006 Scholars Report

Edited by: Thomas A. Okey and Sandy Keys

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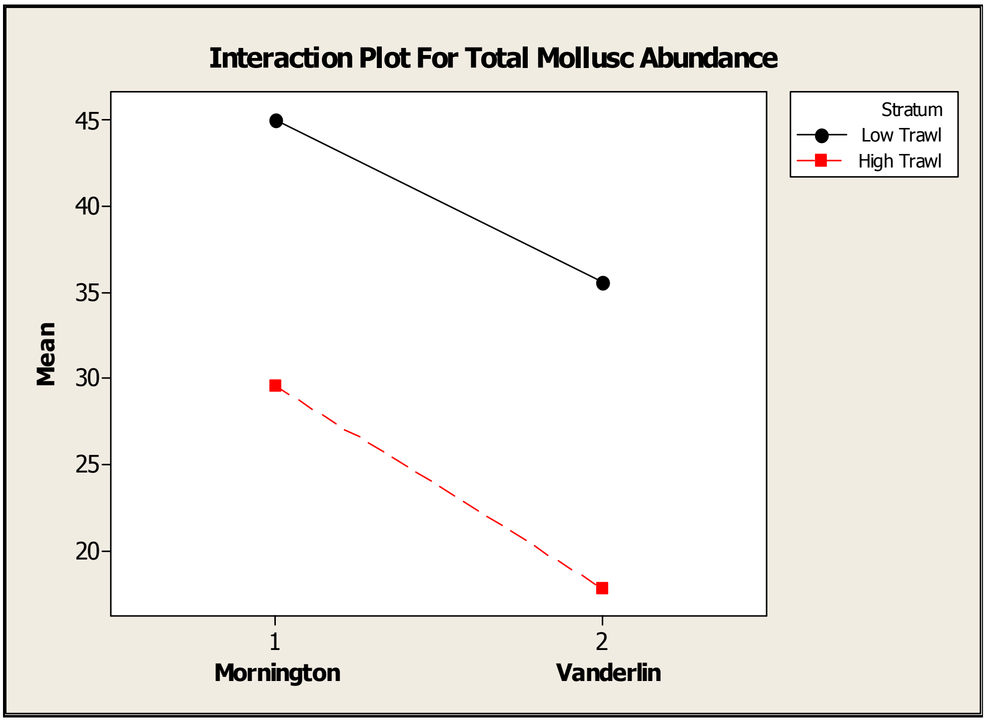

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Important notice

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Preface

Vacation scholarships were offered by CSIRO Division of Marine and Atmospheric Research to enrolled undergraduate students for an eight week period between December 2005 and January 2006 to provide students with the opportunity to undertake an appropriate research project in a field of interest to them and the Division. Applicants were required to have completed not less than three years of a full-time undergraduate course, but those who had already commenced an Honours program were not eligible.

Despite this narrow eligibility requirement, many high quality applicants applied for the several scholarship positions proposed by Division staff. During these eight weeks, the Vacation Scholars were supervised by members of the research staff and most were located at the Cleveland, Queensland laboratory. The present report is a compendium of scholarly contributions by three of these vacation scholars. These contributions demonstrate not only the quality of these particular Vacation Scholars, but also the large science pay-off from the small investment in this Vacation Scholarship program. Each of these contributions fit into larger research programs in the Division, which more broadly benefit the Australian public. These students thus played key supporting roles in these broader projects.

It is often thought that involving students in important project work is too often not worth the investment of time and public funds. There is therefore a common hesitancy, even a strong reluctance among some, to give students the opportunity to be involved. The contributions in this report prove this notion wrong, especially considering that these contributions document the results of merely eight weeks of work by undergraduate students over the holiday season. It must also be considered that the work reported here represents only part of the work that these vacation scholars conducted during their scholarship tenure.

At the conclusion of the 2005-2006 Vacation Scholarship period, these three scholars were asked to present their scholarship work orally during a special seminar held at the CSIRO Marine and Atmospheric Research Laboratory at Cleveland, Queensland. All of the scholars made high caliber presentations, which were well received by the Cleveland Laboratory staff and which initiated much thought provoking discussion.

These Australian scholars have also demonstrated their respective potentials as productive scientists and their ability move into positions from which they can help meet Australia’s growing marine science and stewardship needs.

On behalf of CSIRO’s Division of Marine and Atmospheric Research and on behalf of the Australian public, we thank these authors for their contributions and we wish them the best of luck in their future endeavors.

We thank John Gunn, Deputy Chief of CSIRO Marine and Atmospheric Research, for his strong support of this program and his attention to its execution. We also thank Julie Husin, Senior Recruitment Consultant, for guiding and administering this program. The work described here was built on the work of many individuals, but we specifically mention Dr. Neil Gribble here since his Northern Great Barrier Reef model was the basis of the adapted Torres Strait model. Toni Cannard provided key document production support and editorial assistance.

Thomas A. Okey
Lihir Island, Papua New Guinea, December 2006

Sandy Keys
Brisbane, Australia, December 2006

Patterns of infaunal biodiversity in relation to trawling intensity in the Gulf of Carpentaria: A preliminary exploration

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Introduction

The Gulf of Carpentaria is a shallow (<70m) epicontinental sea that is one of the least biologically characterised marine systems in the world, making it a location of considerable international geological and ecological significance (Bustamante, 2003). Few studies have specifically assessed the diversity and abundance of the soft-sediment infaunal assemblages. Alongi (1989) suggested that infaunal assemblages inhabiting the tropics had examples of both high and low diversity, abundance and biomass because of the wide range of environmental conditions experienced there. A higher frequency of climatic disturbance and more patchy food supply, compared with temperate regions, results in mosaics of resilient infaunal assemblages that are characterised by proportionately more small opportunistic, surface deposit-feeding infauna (Alongi, 1989; Long & Poiner, 1994). This argument was further supported in an empirical study by Long and Poiner (1994) examining infaunal assemblages of the Gulf of Carpentaria. They found that the infaunal abundance and biomass were similar to those within other tropical continental shelves, regulated by physical factors that correlate with sediment grain size and depth, and that scavengers/carnivores and deposit feeders numerically dominated the Gulf, with the most dominant species comprised mainly of opportunistic or second-stage colonizing taxa. With increasing anthropogenic pressure, management of this complex biological system presents a major challenge given the lack of knowledge of ecosystem structure and function and the diversity of environmental gradients and habitats therein.

The principal demersal fishery in the Gulf of Carpentaria, established in the late 1960s, is the $120 million Northern Prawn Fishery (NPF) (Figure 1). The NPF is one of Australia's most valuable Commonwealth fisheries, targeting tiger prawns, king prawns, endeavour prawns and banana prawns, with catch in recent years averaging 8000 metric tonnes (CSIRO, 2003; 2004). The commercial fishery catch is made up of eight prawn species, but three of them: banana prawns (*Penaeus merguiensis*), the brown tiger prawn (*Penaeus esculentus*) and the grooved tiger prawn (*Penaeus semisulcatus*) make up almost 80% of the annual average catch (Wang & Die, 1996; Somers, 1994). Trawling for banana prawns operates mainly in the eastern waters of the Gulf of Carpentaria and occurs only during the day using a balloon trawl, which is a large net that has a wide opening and hardly touches the bottom (Hutchings, 1990). Trawling for the other prawn species occurs at night and is concentrated near coastal seagrass beds in the southern and western areas of the Gulf of Carpentaria. The trawling nets for capture of these prawns are rigged so that they drag along, raised 15cm off the seabed, with a tickler chain dragged along the bottom, in front of the net (Hutchings, 1990). The fishery begins trawling for banana prawns at the beginning of the fishing season around mid March-April and then progressively changes to tiger prawn fishing as banana prawn catch rates decline. Currently the banana prawn fishery lasts around three-four weeks, and the tiger prawn fishery lasts eight months, but is interrupted by a one-month trawl closure in July (Wang & Die, 1996; Hutchings, 1990).

There are various means by which fishing and trawling gear affects the seafloor: directly through scraping and ploughing; sediment resusupension; removal, or scattering of non-target benthos and indirectly through the dumping of bycatch, post-fishing mortality of damaged or disturbed organisms and long-term changes to community structure (Jones, 1992). Watling and Norse (1998) argued that with the possible exception of agriculture, bottom trawling and dredging may be the most physically damaging and widespread of all anthropogenic disturbance. It has been argued that the disturbances associated with benthic trawling have widespread impacts on soft sediments and the communities that inhabit them (Rijnsdorp et al., 1998). The extent to which the abundance and diversity of organisms and the overall community structure are affected by trawling is dependent on the duration and incidence of the physical disturbance, and the intrinsic capacity of the population or assemblage to endure, and recover from the disturbance (Underwood, 1989; Lindegarth et al., 2000).

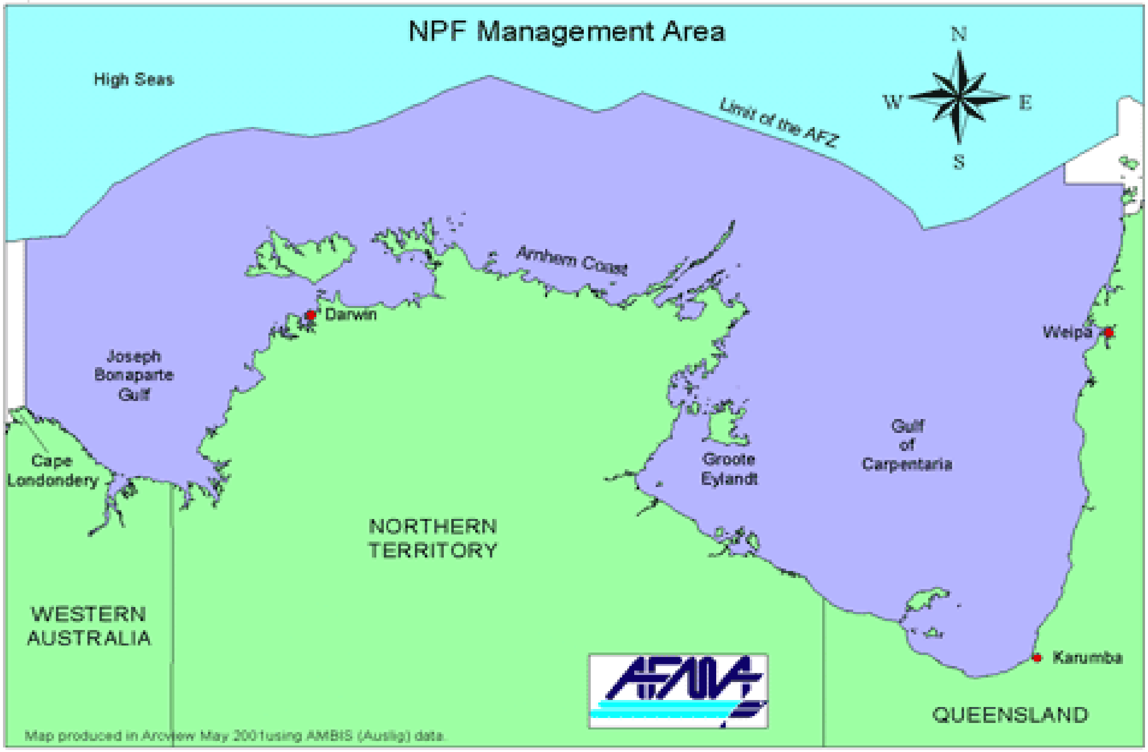


Figure 1: The Northern Prawn Fishery occupies an area of 771,000 square kilometres off Australia's northern coast. It extends from the low water mark to the outer edge of the Australian fishing zone along some 6000 kilometres of coastline between Cape York in Queensland and Cape Londonderry in Western Australia (Source: CSIRO, 2004)

A plethora of studies have been conducted to assess trawling impacts (eg: Dayton et al., 1995(review); Engel & Kvitek, 1998; Jennings & Kaiser 1998; Frid and Clark, 1999; Frid et al., 2000; Jennings et al., 2002) with mixed results. Demonstrating a significant impact of trawling on the benthos is difficult due to the natural disturbance regime and the high variability of the benthos, temporally and spatially (Drabsch et al., 2001). The effects of trawling on infauna are more noticeable where levels of natural disturbance are low than where they are high because the community is already adapted to disturbance regimes (Jennings et al., 2001; Kaiser & Spencer, 1996; Jennings and Kaiser, 1998; Brylinsky et al., 1994). Recovery of the community following trawling also depends on the natural disturbance regime. In one study conducted on sites with mobile substratum and strong currents, experimental trawling changed the community composition but within six months there was no significant difference between trawled and control sites (Kaiser & Spencer, 1996; Jennings et al., 2001). But in more stable sediments, recovery is slower. Tuck et al. (1998) studied trawling impacts in a previously untrawled muddy area and impacts were still apparent 18 months later and similarly Collie et al. (1997) showed that recovery from scallop dredging took at least two years (Jennings et al., 2001). This means that the effects of small-scale disturbance events, even when frequent, can be repeatedly masked by the background of large-scale disturbance, or that the scale of disturbance can be small enough to allow a rapid recolonisation such that larges-scale effects are never apparent (Hall et al., 1993; Kaiser et al., 1998). However, Kaiser et al. (1998) argues that presumably there exists a threshold scale and frequency of disturbance events at which lasting ecological effects may occur, even against a backdrop of natural disturbance. The impacts from continued trawling of the benthos may exceed this threshold, leading to long-term changes in community structure and function.

According to Jennings et al. (2001), to describe the effects of trawling disturbance at a fishery level, it is necessary to study the real fisheries where disturbance occurs at large scales over long time periods (Collie et al., 1997; Thrush et al., 1998; Kaiser et al., 2000). Studies of the benthos within three broad regions of the Gulf of Carpentaria subjected to different levels of trawling intensity were undertaken over the period February-March 2005. As part of a CSIRO Summer Scholarship in 2006 the aim of this preliminary sub-sample was to investigate the effects of otter trawling over two (out of three) quantifiable gradients of disturbance on the infaunal benthos of two (out of three) regions within the Gulf of Carpentaria. According to Drabsch et al. (2001) infaunal assemblages are good indicators of disturbance because of their sensitivity to habitat alterations and importance to the structure and function of marine systems. Their activities within the marine system affect important characteristics of the sediment, increasing the water and oxygen content, topographic detail, microbial activity and erosion resistance, decreasing compaction, and altering the grain size (Hall, 1994; Drabsch et al., 2001). Infauna are also taxonomically and functionally diverse with numerous other species within complex food webs depending directly or indirectly on them (Bilyard, 1987; Drabsch et al., 2001).

We seek to address whether there is any quantifiable difference between the two disturbance gradients of otter trawling for prawns on the infaunal assemblages of the Gulf of Carpentaria through univariate analyses of abundance and diversity.

Materials and Methodology

2.1 Site and Survey Description

The survey design was developed around a natural experiment (i.e.: the existing human "manipulation" of different levels of disturbance on the benthos) to examine the effects of trawling on the infaunal assemblages of the Gulf of Carpentaria. The three key sampling regions of Mornington Island, Vanderlins Islands and Groote Eylandt were chosen to maximize the degree of environmental variation present in the Gulf of Carpentaria and the selection and delineation of these regions was completed by analysing previous environmental and fisheries data (Figure 2). Fishing intensity was determined by examining AFMA's Vessel Monitoring System (VMS) data for five previous years (1999-2003)(Figure 3). A "High Trawl" average had >24hours of trawling a year, with >6hours in four out of the five years and >6hours in 2003. A "Medium Trawl" average had >6hours of trawling a year, with >2hours in four out of five years and >2hours in 2003. A "Low Trawl" average had <1 hour of trawling a year. There were a total of 126 sampling stations and to examine the effects of commercial trawling the 42 sample locations within each geographic region were distributed randomly within each of the three trawl intensity strata. Locations were adjusted where necessary to eliminate extreme clumping or avoid untrawlable ground.

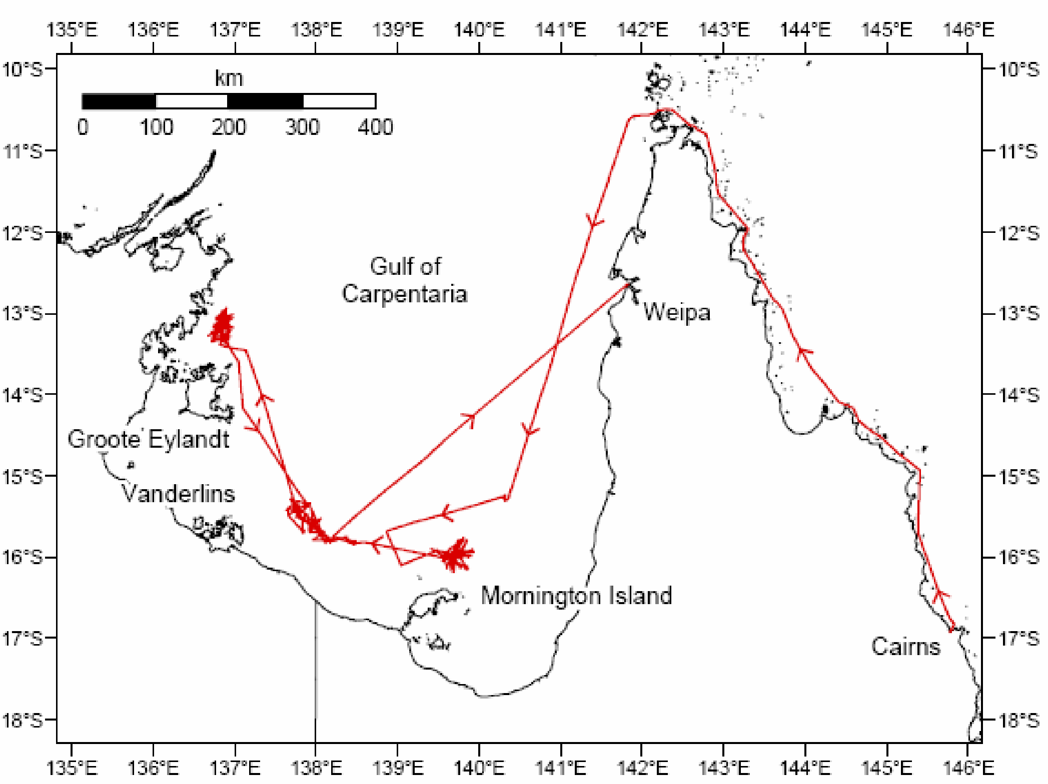


Figure 2: The voyage track within the Gulf of Carpentaria

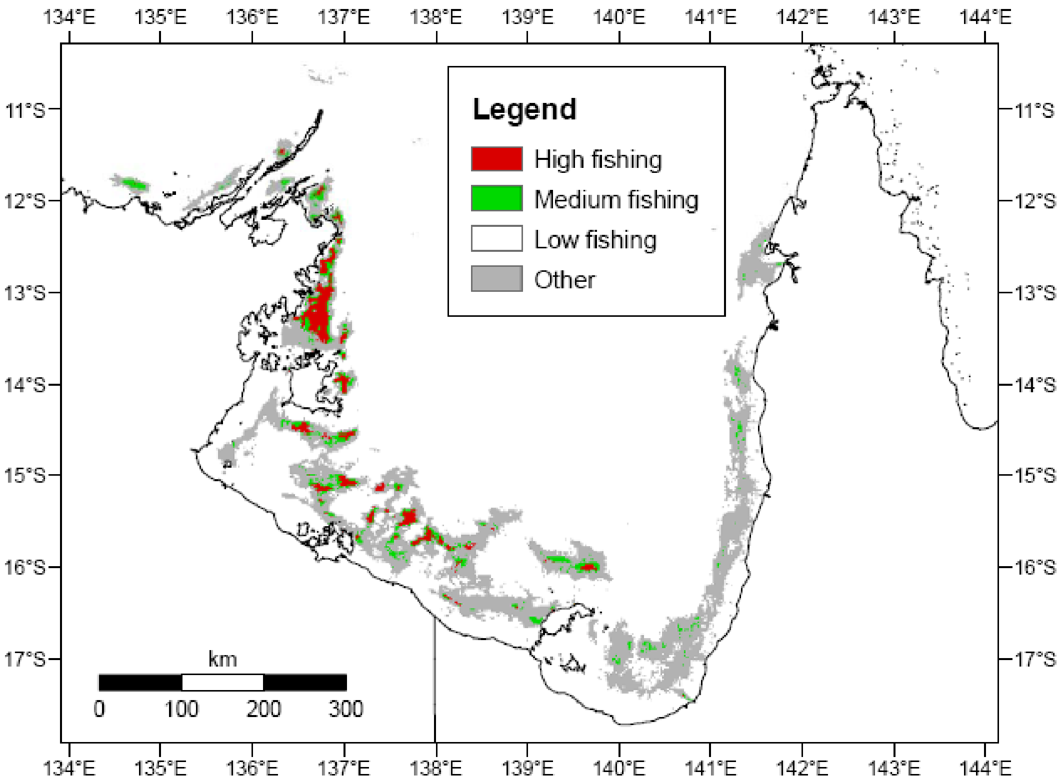


Figure 3: The levels of fishing strata within the Gulf of Carpentaria

The Scientific expedition using the RV *Southern Surveyor* was carried out between 23rd February and 21st March 2005. Benthic samples were collected using a 0.05m^2 box core and placed into sample storage bins (Nally bins). Each sample was visually inspected to determine that the sample was reasonably undisturbed and the penetration depth was measured. Sample contents were then transferred from sample storage bins to a 500 µm sieve where they were immersed in water and washed by fine spray to remove fine sediment. Residual material was washed into appropriately sized, pre-labelled heavy-duty zip-lock bags and a 10% phosphate-buffered formalin solution stained with rose-bengal was decanted into the bags before the bag was sealed and placed into a pre-labelled drum for later analysis. Prior to species analysis samples were reopened and the 10% phosphate-buffered formalin solution removed and replaced by a 70% ethanol solution.

As part of this preliminary exploration, five samples were selected haphazardly-from each high and low trawl intensity stratum at both Mornington and Vanderlins (Figure 4 a,b). All samples were swirled and sieved using a 500 µm mesh sieve. Identification of all organisms to species level is a major time and cost restraint (Warwick, 1993; Clarke & Warwick, 1994; Somerfield & Clarke, 1995). Many studies on macro-benthic disturbance have shown that very little information is lost by working at a taxonomic level higher than species (i.e.: family) (Alves et al., 2003). Thus, due to time constraints, animals were identified to only Order and Family level. Polychaetes were identified to Family using Wilson et al. (2003) and Beesley et al. (2000); Crustaceans, Molluscs, Echinoderms and other Annelida were identified to Order using Jones and Morgan (1994), Ruppert and Barnes (1994) and Smith and Carlton (1975).

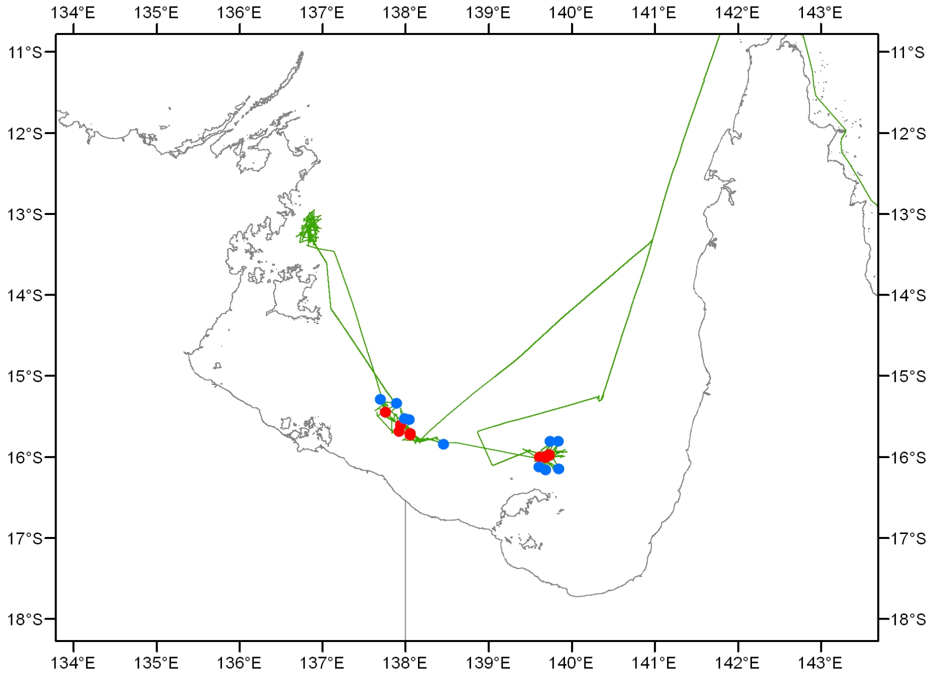


Figure 4a: Map of voyage track with location of the twenty High (red) and Low (blue) Trawl operations

2.2 Data Analysis

Data analysis was conducted using univariate statistics. The number of taxa and individual animals per core are widely used indicators of disturbance and were examined here. A diversity index was not used because they lack sensitivity (Warwick & Clarke, 1991, T. Okey pers. comm.) and different indices can give conflicting results (Hurlbert, 1971). Two-way ANOVA using Minitab 14 was employed to test for a significant interaction between Stratum x Trawling Effect in the total number of species and abundance for each major phylum group. The interaction between Region x Trawling Effect was also of interest to determine the impact of trawling between the two locations. The traditional alpha value of 0.05 was used to define significance in all analyses as ANOVA is robust to all but serious heterogeneity, and a more conservative alpha value increases the chance of Type II errors, which are more detrimental in impact studies (Underwood, 1993; Drabsch et al., 2001). A regression fitted line graph of polychaete abundance against crustacean abundance, and numerous interaction plots were created using Minitab 14.

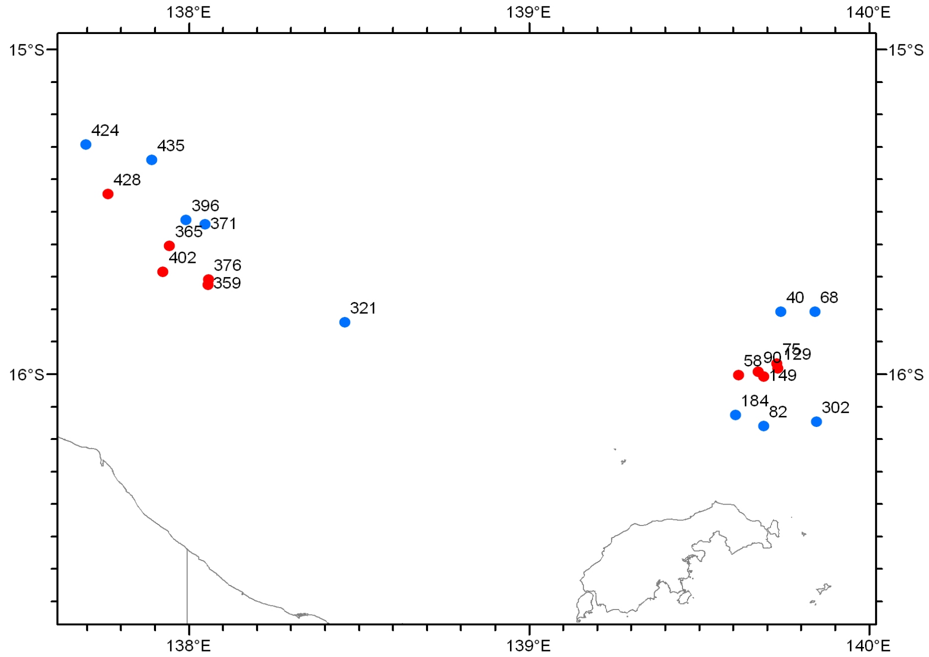


Figure 4b: Zoomed image of 4a with the twenty individual operation numbers.

Results

In all, 2833 infaunal invertebrates were sorted from box core samples taken from the twenty operations. A total of 883 polychaetes (602:Mornington and 281:Vanderlins), a total of 1007 crustaceans and a total of 943 (578:Mornington and 365:Vanderlins) "other phylum" were taken from the twenty operations. The coefficients of variation of the total number of species and abundance per operation were significantly different between both stratum (*p=0.036*) and region (*p=0.044*). The high intensity trawled operations (41.10) had on average a greater coefficient of variance than low intensity (34.07) and Mornington (39.53), on average had a greater coefficient of variance than Vanderlins (32.78).

Figures 5, 6, 7 and 8 display the descriptive statistics for all the interactions in this study design. Crustaceans numerically dominated the infauna (36%) followed by polychaetes (31%). The region of Mornington appeared to have a much higher abundance and diversity of phylum than Vanderlins but there was a large amount of variation in the phylum results for Mornington compared to Vanderlins. The Low Trawl operations within each region appear to have greater abundances and diversity than the High Trawl operations. The Mornington Low Trawl operations had the most abundant and diverse taxa. The most abundant individual operations were 68 (M.LT), 82 (M.LT), 184 (M.LT) and 129 (M.HT). The most diverse individual operations were 82 (M.LT), 129 (M.HT) and 184 (M.LT).

A positive correlation was found between total polychaete mean abundances and crustacean mean abundance with an r^2 value of 68%, implying that overall the biophysical conditions were suitable for the coexistence of both phylum (Figure 9).

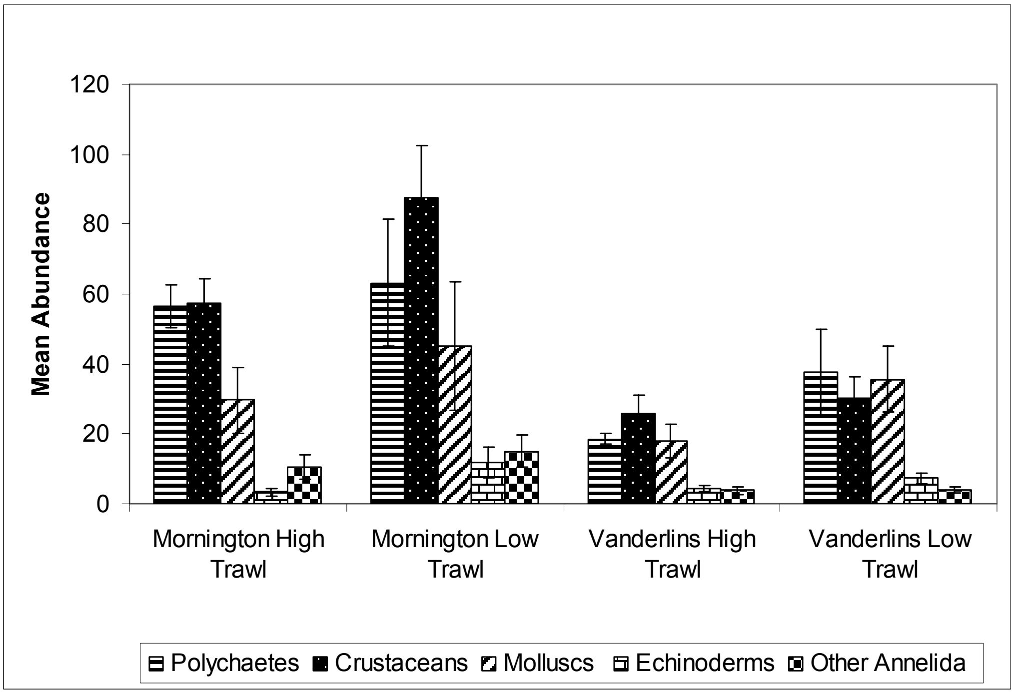


Figure 5: Mean (with Standard Error) abundance of taxa in relation to region and stratum

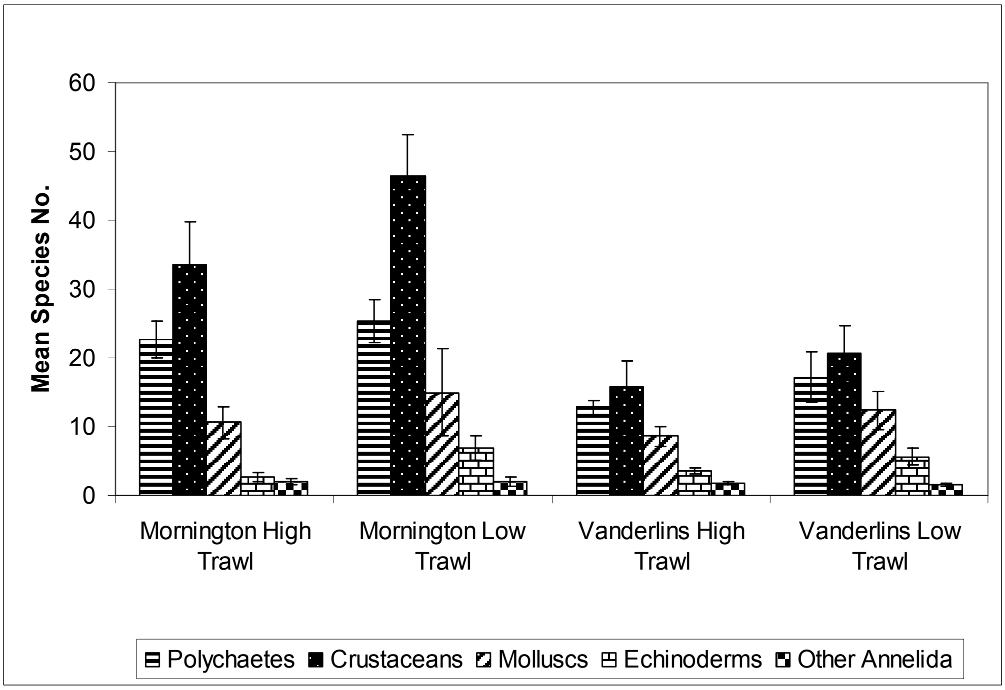


Figure 6: Mean (with Standard Error) diversity of taxa in relation to region and stratum

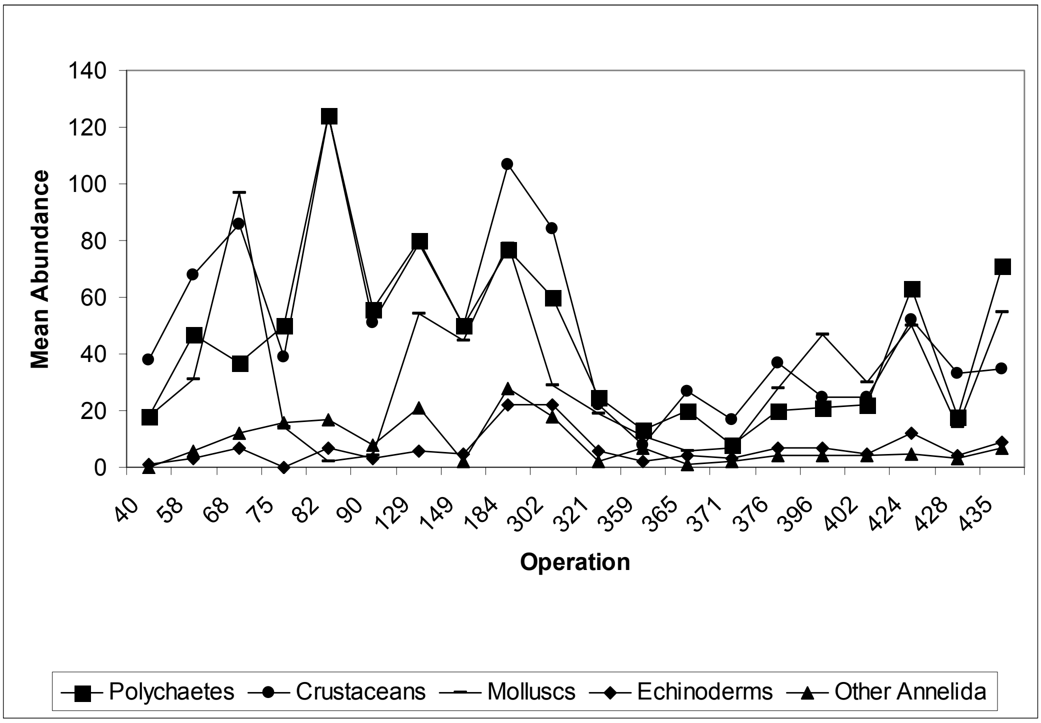


Figure 7: Mean abundance of taxa for all operations

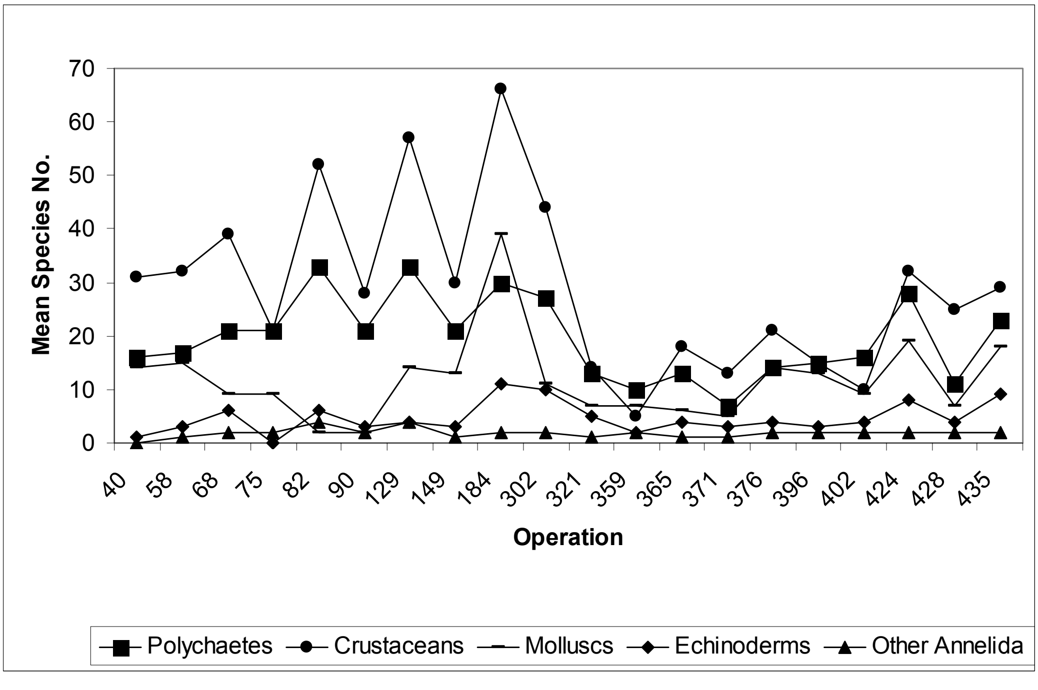


Figure 8: Mean diversity of taxa for all operations

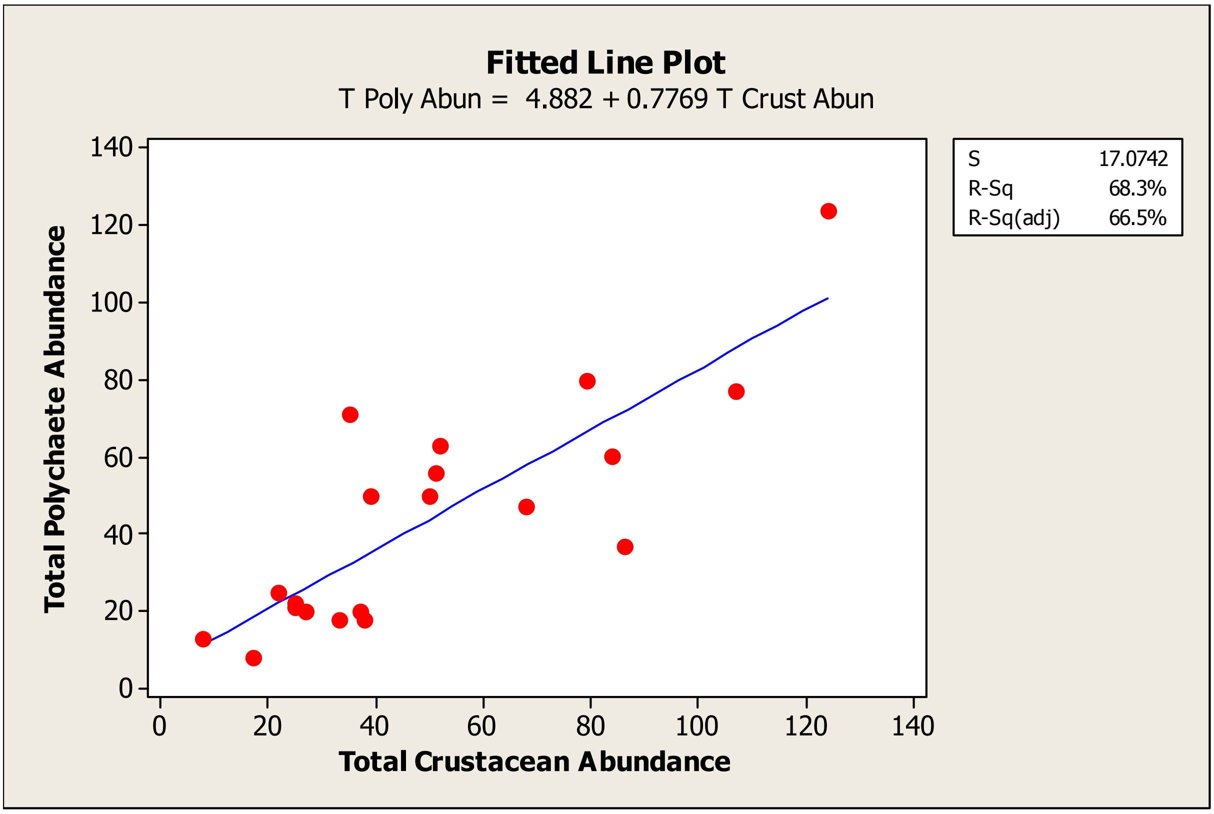


Figure 9: Fitted line plot displaying a positive correlation between total polychaete and crustacean mean abundance

Capitellidae numerically dominated the polychaetes (16%), with the top ten most abundant polychaete families representing 75% of all individuals (*Figure 10 and 11*). At Mornington, *Capitellidae*, *Spionidae*, *Nephtyidae, Lumbrineridae* and *Terebellidae* were found in greater abundances in High Trawl operations compared to Low Trawl, but at Vanderlins only *Lumbrineridae* was found in greater abundances. *Gammaridea amphipoda* numerically dominated the Crustaceans (28%), followed by *Tanaidacea* (18%), *Decapoda* (17%), *Ostracoda* (15%) and *Cumaceans* (8%)(*Figure 12 and 13*). At Mornington, *Isopods*, *Tanaids*, and *Caprellidea amphipoda* were found in greater abundances in High Trawl operations compared to Low Trawl, which was similar to Vanderlins with *Isopods*, *Tanaids*, and *Calanoida Copepods* found in greater abundances.

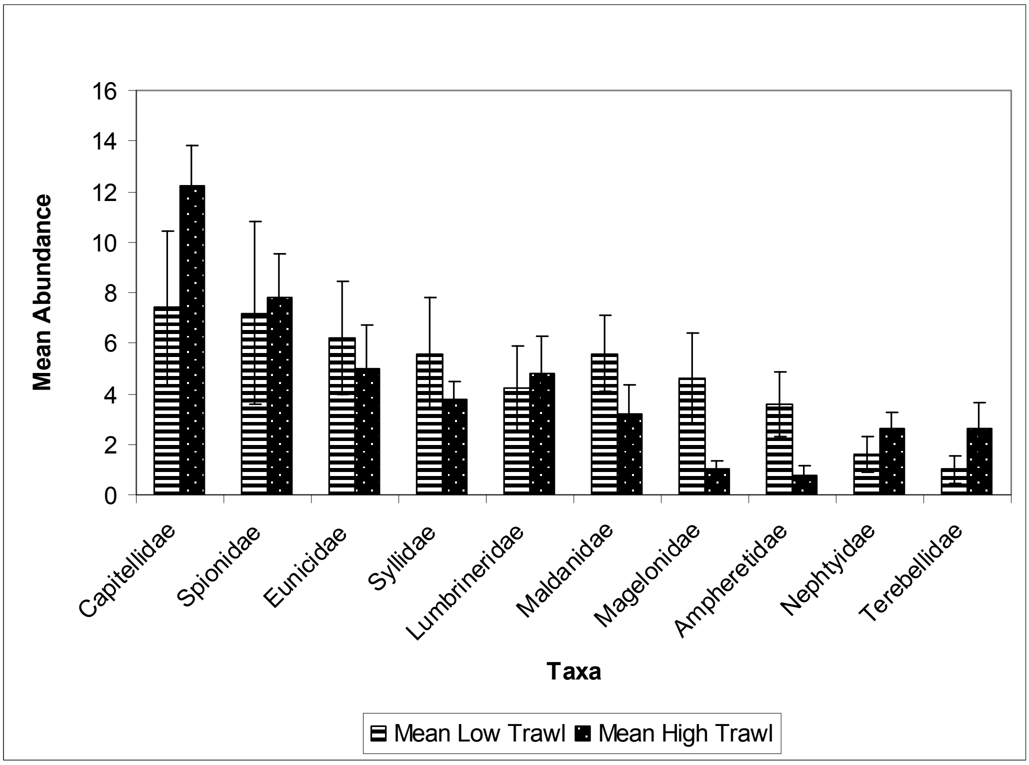


Figure 10: Top ten most abundance polychaete families at Mornington

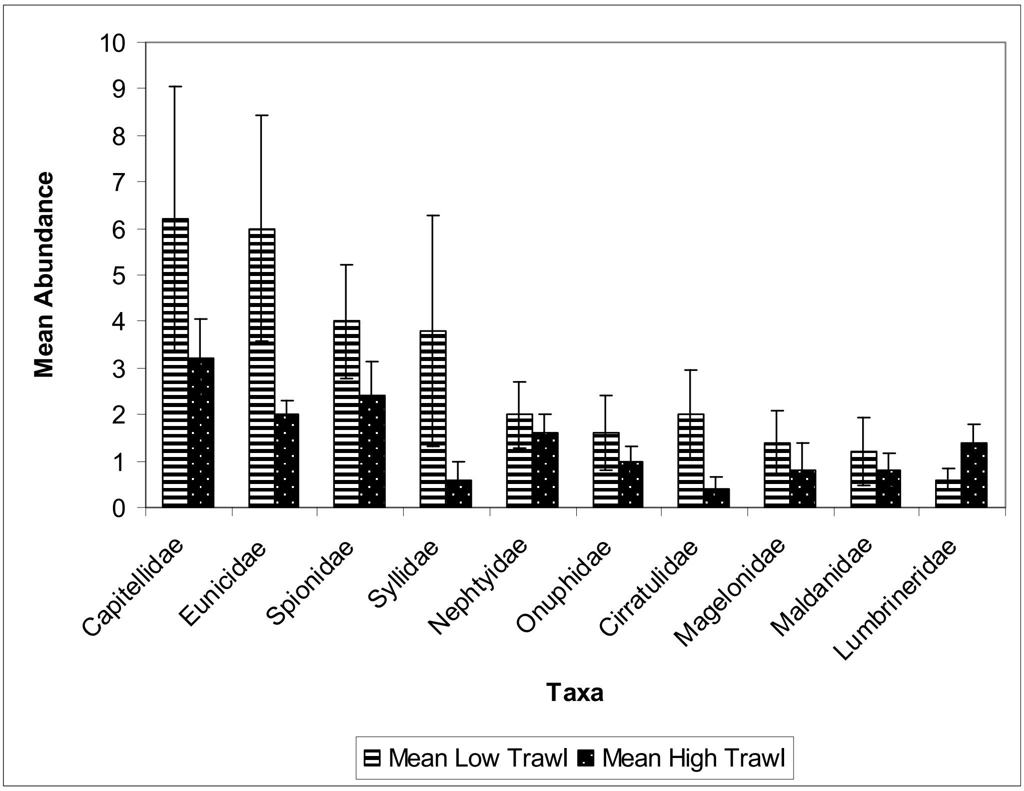


Figure 11: Top ten most abundance polychaete families at Vanderlins

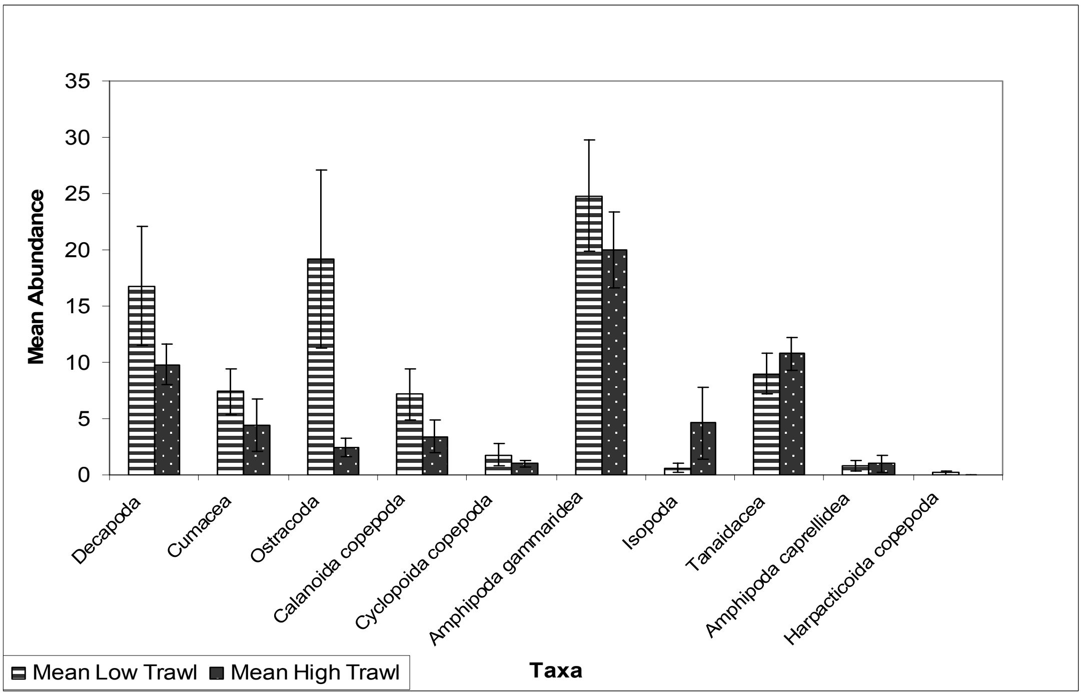


Figure 12: Mean crustacean abundance at Mornington with the effect of stratum

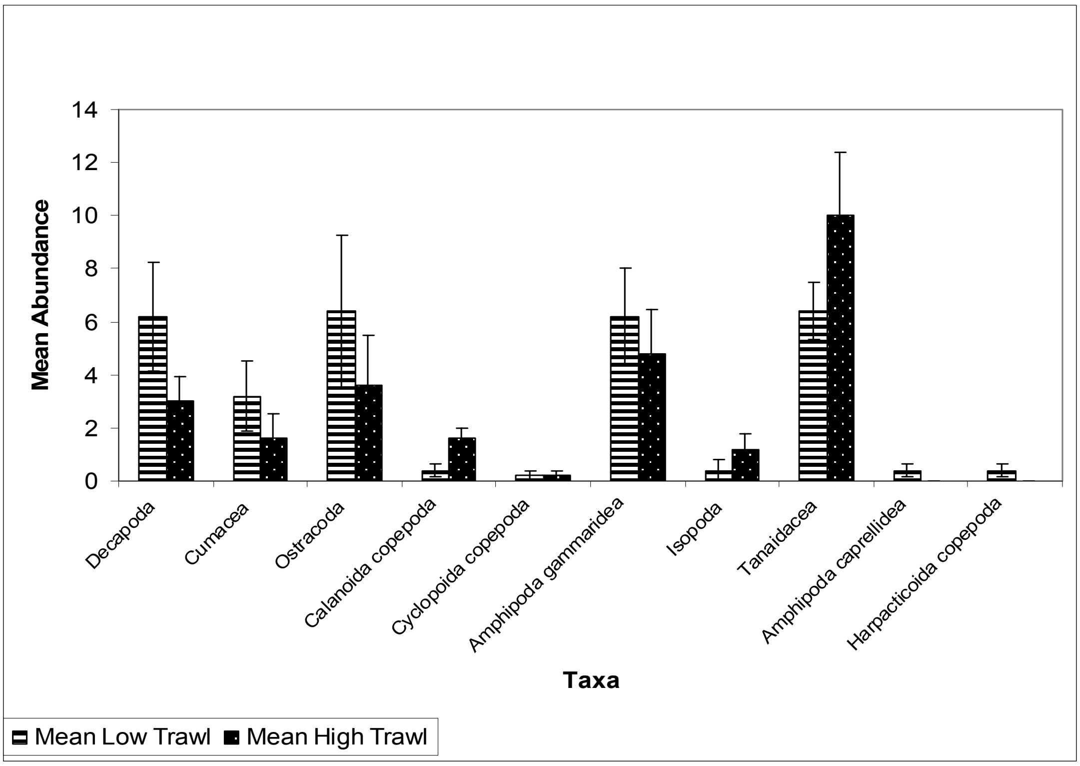


Figure 13: Mean crustacean abundance at Vanderlins with the effect of stratum

The distribution of individuals among taxa was very uneven with the top ten polychaete families at Mornington and Vanderlins accounting for 65% and 67% of all polychaete infauna respectively. *Capitellidae* was the most diverse at Mornington accounting for 12% of all polychaete infauna and *Eunicidae* was the most diverse at Vanderlins with 13% (*Figure 14 and 15*). For the crustaceans, Decapod species accounted for 25% of all crustacean infauna, followed by Gammaridea amphipoda 24% and *Tanaids* 15% (Figure 16 and 17).

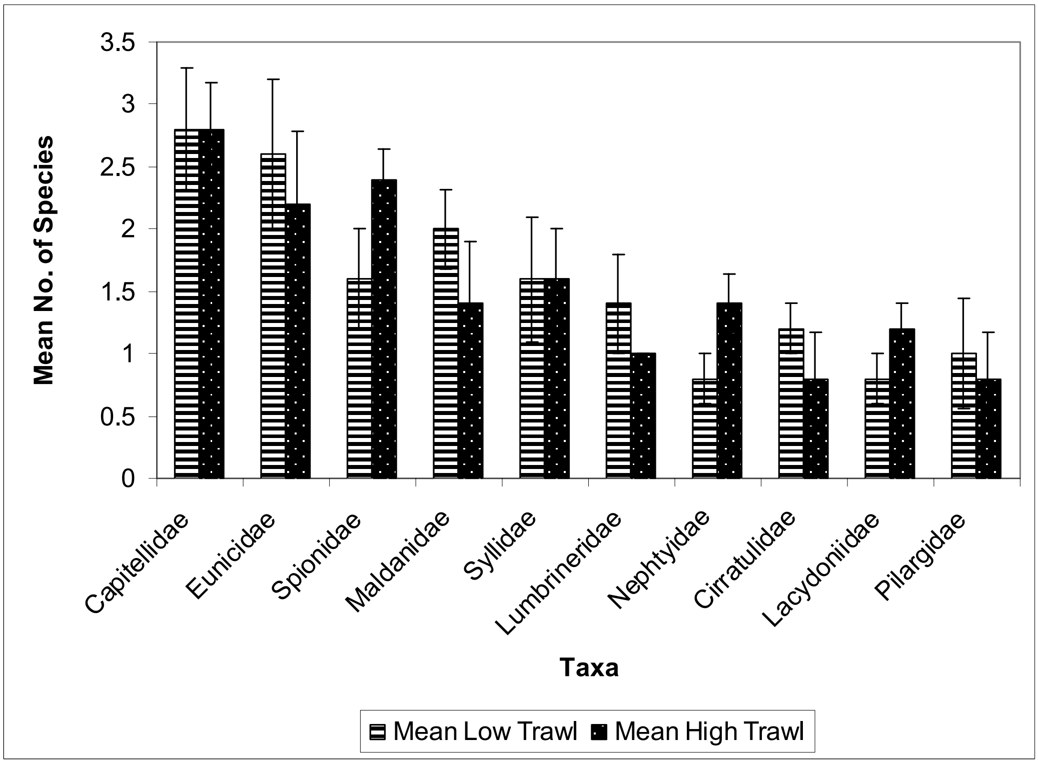


Figure 14: Top ten most diverse polychaete families at Mornington

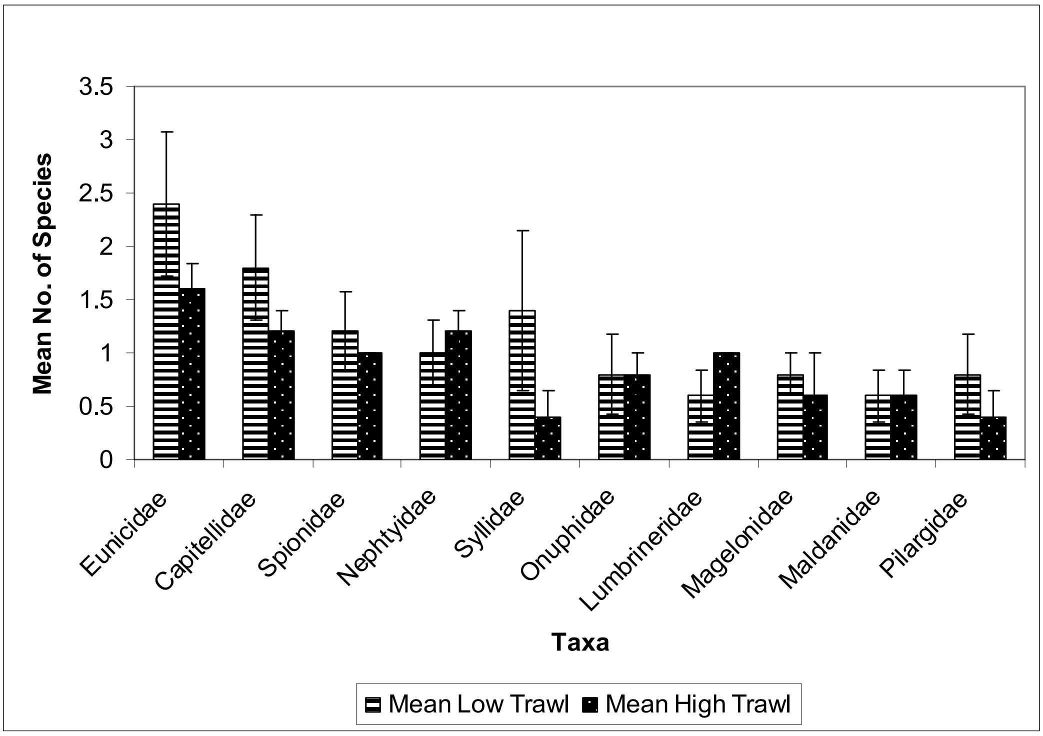


Figure 15: Top ten most diverse polychaete families at Vanderlins

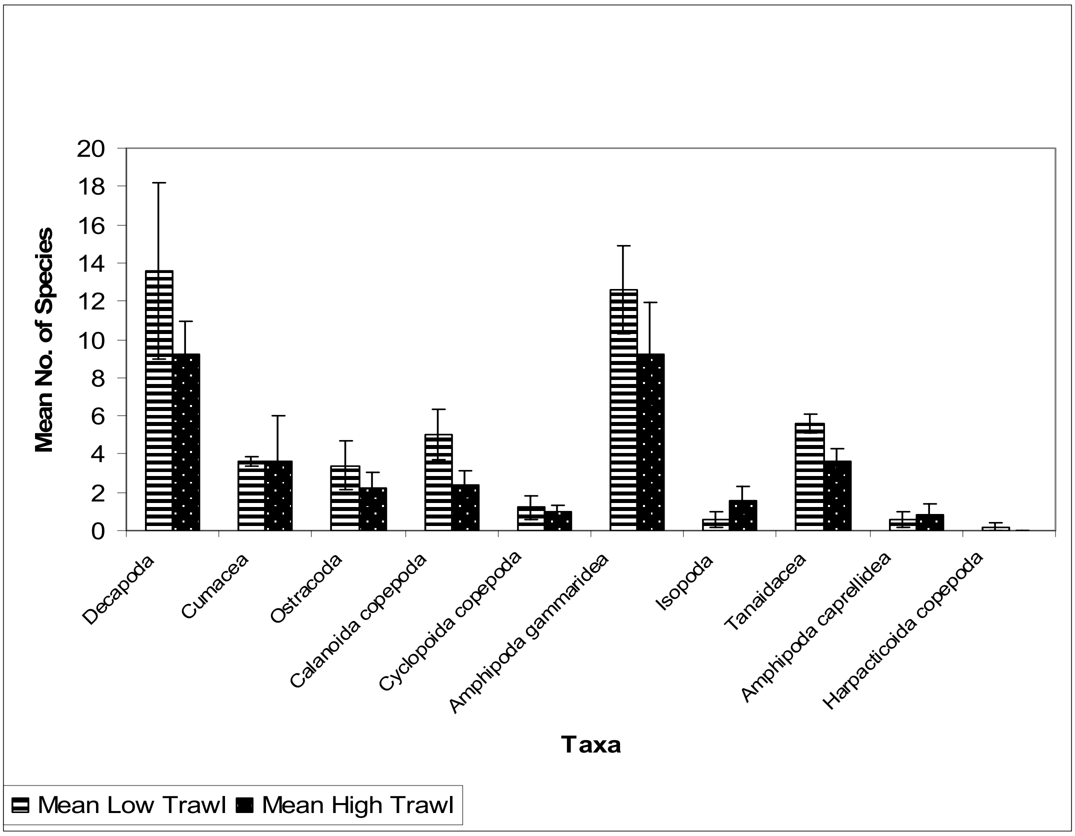


Figure 16: Mean number of crustacean species per Order at Mornington with the effect of stratum

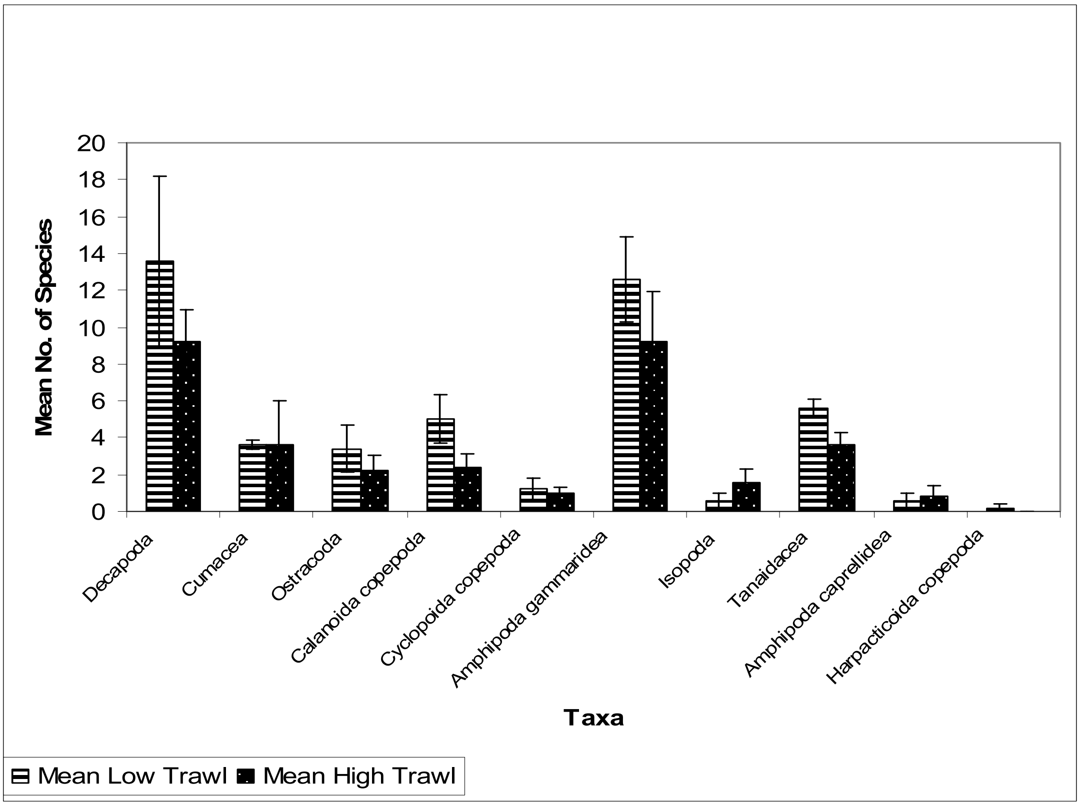


Figure 17: Mean number of crustacean species per Order at Vanderlins with the effect of stratum

Two-Way ANOVA (Table 2) found no significant difference between species abundance/diversity and stratum (High vs. Low) except with the Echinoderm abundance and diversity. Large-scale variation in the form of natural disturbance regimes appeared in general to swamp any variation that could have been attributed to trawling intensity (stratum). There was however a significant difference between polychaete and crustacean abundance/diversity between regions (Vanderlins vs. Mornington). The interaction between Region x Stratum was not significantly different.

Table 2: Two-Way ANOVA infaunal abundance and diversity results for the selected univariate variables

Two-Way ANOVA Source	F	p (>0.05)	Two-Way ANOVA Source	F	p (>0.05)
Polychaete Abundance x Stratum	1.25	0.280 (N.S)	*Polychaete Species x Stratum*	1.63	0.220 (N.S)
Polychaete Abundance x Region	7.72	0.013 (S)	*Polychaete Species x Region*	10.19	0.006 (S)
Stratum x Region	0.29	0.595 (N.S)	*Stratum x Region*	0.08	0.780 (N.S)
Crustacean Abundance x Stratum	3.71	0.072 (N.S)	*Crustacean Species x Stratum*	3	0.103 (N.S)
Crustacean Abundance x Region	24.55	0.000 (S)	*Crustacean Species x Region*	18.4	0.001 (S)
Stratum x Region	2.13	0.164 (N.S)	*Stratum x Region*	0.62	0.443 (N.S)
Mollusc Abundance x Stratum	2.06	0.170 (N.S)	*Mollusc Species x Stratum*	1.21	0.288 (N.S)
Mollusc Abundance x Region	0.84	0.373 (N.S)	*Mollusc Species x Region*	0.38	0.546 (N.S)
Stratum x Region	0.01	0.919 (N.S)	*Stratum x Region*	0.01	0.937 (N.S)
Echinoderm Abundance x Stratum	5.77	0.029 (S)	*Echinoderm Species x Stratum*	7.23	0.016 (S)
Echinoderm Abundance x Region	0.51	0.484 (N.S)	*Echinoderm Species x Region*	0.01	0.932 (N.S)
Stratum x Region	1.29	0.272 (N.S)	*Stratum x Region*	0.91	0.354 (N.S)

Figure 18, 19, 20 and 21 illustrate the interaction between polychaete, crustacean, mollusc and echinoderm abundance with stratum and region. All follow a similar pattern, with Low Trawl operations having a higher mean abundance than High Trawl operations which is most pronounced within the mollusc results. Mornington on average has a higher abundance of taxa than Vanderlins, except for echinoderms in High Trawl operations. The difference between regions is most pronounced in the crustacean mean abundance. Further illustrations of mean abundance against individual operations are available in Appendix A.

Figure 22, 23, 24 and 25 illustrate the interaction between polychaete, crustacean, mollusc and echinoderm mean number of species with stratum and region. All follow a similar pattern, with Low Trawl operations having a higher mean number of species than High Trawl operations which is most pronounced within the mollusc results. Mornington on average has a higher number of species than Vanderlins, except for echinoderms in High Trawl operations. The difference between regions is most pronounced in the crustacean mean number of species. Further illustrations of mean number of species against individual operations are available in Appendix (A-H).

Figure 18

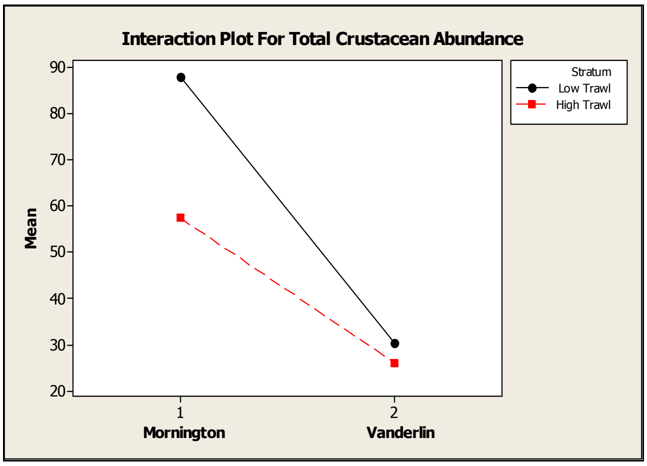


Figure 19

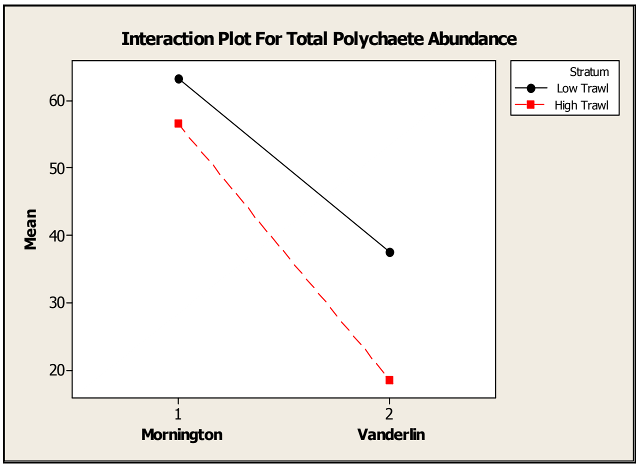


Figure 20

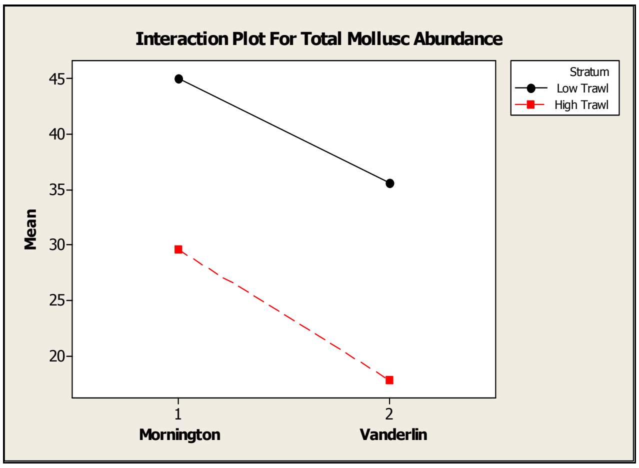
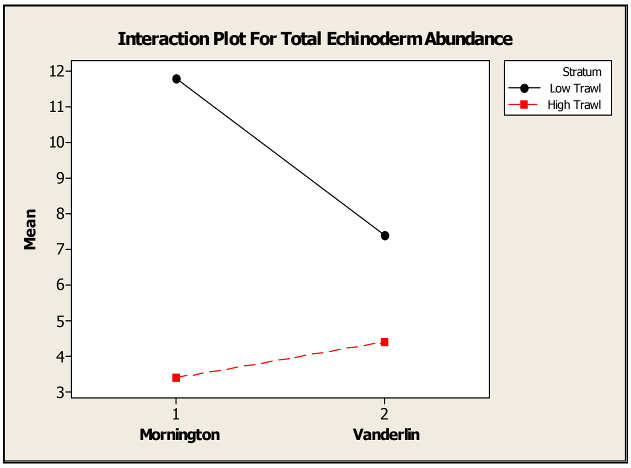


Figure 21



Figures 18, 19, 20 and 21: Interactions between abundance and stratum and region for crustaceans, polychaetes, molluscs and echinoderms respectively**.**

Figure 22

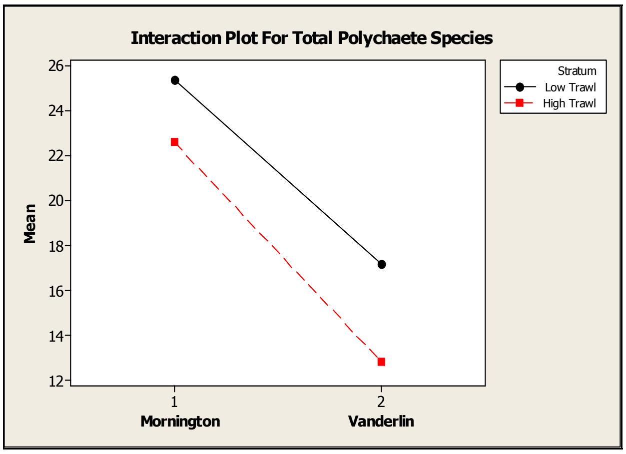


Figure 23

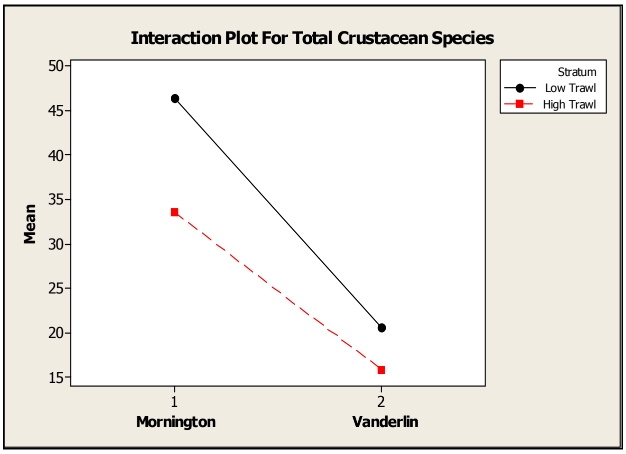


Figure 24

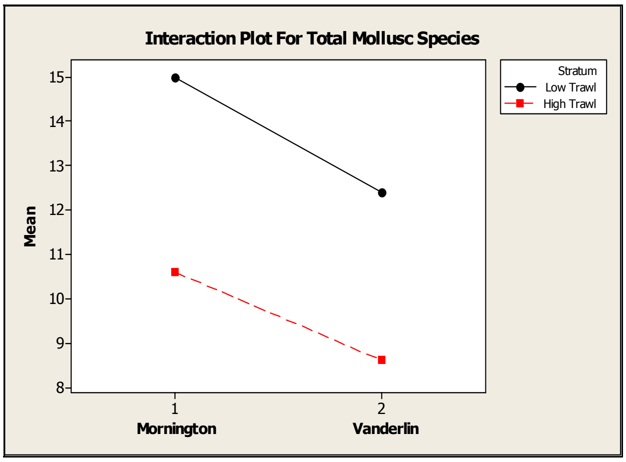
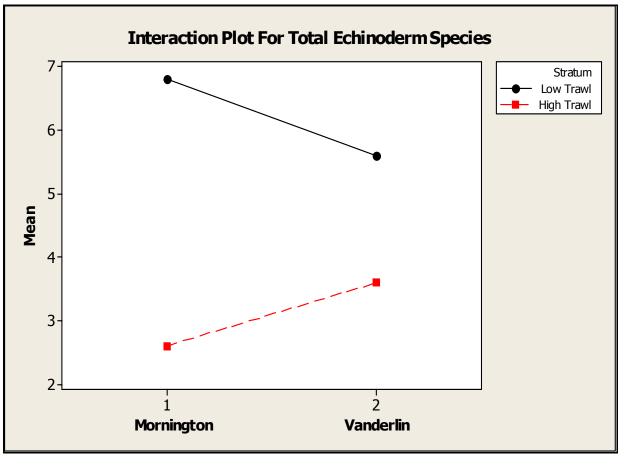


Figure 25



Figures 22, 23, 24 and 25: Interactions between mean number of species and stratum and region for crustaceans, polychaetes, molluscs and echinoderms respectively.

Discussion

Benthic communities within shallow (<100m) continental shelf communities experience continual disturbance at various scales (Hall, 1994). Large-scale natural disturbances, such as seasonal storms and daily scouring by tidal currents, form a backdrop against, which other, smaller disturbances occur, such as predation interactions (Kaiser, 1998). Hall et al. (1993) concluded that while it was possible to detect short-term effects of predator disturbance, large-scale effects never manifested because they were either masked by a background of large-scale disturbances or small-scale disturbances permit such a rapid recolonisation that large-scale effects never become apparent (Kaiser, 1998). Therefore, benthic communities that inhabit frequently disturbed environments are less likely to exhibit any long-term changes in their structure and function in response to trawling intensity (Shepherd, 1983; Kaiser & Spencer, 1996, Kaiser et al., 1998; Kaiser, 1998; Posey et al., 1996). Kaiser (1998) argues, that seemingly the scale and frequency of physical disturbance events can increase to a point where lasting ecological effects are observed even against a backdrop of natural disturbance but within this preliminary subsample there was little evidence to suggest that this threshold was exceeded for the Gulf of Carpentaria infaunal assemblages.

While it is probable that the highly variable natural disturbance regime of the Gulf of Carpentaria masked any difference between stratum it is also possible that the small sample size (n=5) might not have been adequate to capture real differences. Kaiser (2003) in an examination of disturbance effects on the benthos concluded that detection was scale-dependent in relation to the size of samples collected and the type of sampling device used. At smaller spatial scales (such as in this study), others have also not detected disturbance effects on benthic species diversity (Kaiser et al., 1994; Thrush et al.1995; Currie & Parry, 1996; Kaiser et al., 2000). For example, in this study no statistically significant difference was observed between crustaceans and stratum, but a statistically significant difference was found between echinoderms and stratum. Kaiser (2003) argues that crustacean such as decapods are often patchily distributed across the seabed so at smaller scales of sample size, small-scale variation in the distribution of decapod fauna may be increased such that the statistical power to detect responses to fishing decreases. Small echinoderms such as ophiuroids however are probably sampled adequately at small scales (Kaiser, 2003), which may explain why statistically significant difference was found in their abundance and diversity between stratum.

It is also possible that the lack of stratum difference was due to the light otter trawl gear, which is used within the Northern Prawn Fishing grounds. Poiner et al. (1998) in the Great Barrier Reef region found that a single pass with an otter trawl did not significantly affect the epifauna, and severe damage was only detected after repeated trawling of the same ground. In a meta-analysis conducted by Collie et al. (2000) they found that otter trawling had the least significant impact on fauna compared with other gears. The otter doors that hold the wings of the otter trawl open cause the most damage when trawling, but these only constitute a small proportion of the total width of the gear (Kaiser et al., 2002). Additionally, the AFMA VMS data did not include 2004's data and it is also possible that with a small sample size some of the classified "High Trawl" operations were not trawled for as many hours as in previous years. This difference in effort would have a confounding effect on the data and explain to some degree the non-significant difference between stratum, because the communities would have been able to recover from the disturbance. According to Collie et al. (2000) sandy sediment communities are able to recover within 100 days implying that they can withstand 2-3 trawling incidents a year without changing markedly.

In the study done by Long and Poiner (1994) in the Gulf of Carpentaria they characterised the sediment of south-east Mornington as sand and muddy sands, whereas the western Vanderlins was mud and sandy muds. They also found that trends in abundance and species richness were correlated with Gulf-wide trends in sediment type and depth, with highest abundances and species richness occurring in the eastern and south-eastern areas of the Gulf (Mornington), and the lowest in the western and north-western Gulf (Vanderlins)(Long & Poiner, 1994). That result was analogous to this study, with the region of Mornington having a significantly higher abundance and species richness than Vanderlins. Hall (1999) argues that this is a result of trawling disturbance varying with sediment type, as the gear penetrates deeper into soft muds than sands (Drabsch et al., 2001).

The coefficients of variation of the total number of species and abundance per operation were significantly different between both stratum (p=0.036) and region (p=0.044). The high intensity trawled operations (41.10) had on average a greater coefficient of variance than low intensity (34.07) and Mornington (39.53), on average had a greater coefficient of variance than Vanderlins (32.78). This was a similar result to Kaiser and Spencer (1996) who in comparing trawled and untrawled sites within two regions with different sediment types found that the coefficient of variance was greater in the mobile sediments compared to the more stable sediments and greater in the trawled sites compared to the untrawled sites. Lending credence to the hypothesis that heterogeneity increases with physical disturbance (Thistle, 1981; Caswell & Cohen, 1991; Warwick & Clarke, 1993; Kaiser & Spencer, 1996).

In terms of notable taxa trends, in the study competed by Long and Poiner (1994) they identify three polychaete families that dominated their phylum abundance and diversity counts: *Capitellidae*, *Eunicidae* and *Terbellidae*. All of which were either deposit feeders or scavengers/carnivores. Likewise the top ten polychaete species which dominated in this study were either deposit feeders or scavengers/carnivores which are indicative of disturbance. According to Frid et al. (2000), the dead and injured fauna left on the sea floor, and the addition to the benthos of offal and dead by-catch, increases opportunities for mobile scavengers/predators; with continual disturbance benefiting opportunistic fauna. Of the most abundant and diverse polychaete families in this study most are opportunistic and mobile scavengers/predators, but were found in roughly equal numbers in both high and low stratum and to some degree between regions. They were also found in high abundance with crustaceans and a positive correlation between polychaete abundance and crustacean abundance was found, indicating that perhaps the Gulf conditions are suitable for the proliferation of both groups in significant abundance. One point of note though when examining the results is that due to the lack of time and taxonomic expertise, species were sorted into a Family and Order level and from these the number of species were conservatively estimated which may have led to a overestimation of true species numbers.

Conclusion

This study identified that Mornington was characterised by more unstable sediments and natural variability than Vanderlins and had a significantly greater abundance and diversity of species as analogous to the Long and Poiner (1994) study. But despite having a slightly greater coefficient of variance, High Trawl operations were not significantly different to the Low Trawl operations. This was either due to the large amount of variability between samples as a factor of natural disturbance regimes; the impact of light trawl gear and possible confounding of fishing effort data with subsequent recovery of communities; or due to the small sample size (n=5) which might not have represented the community adequately enough to detect a significant difference. Indeed the large variability between major phylum abundances/diversity for both regions and stratum is often recognised in small sample sizes.

Many present-day studies are constrained to examine an already considerably altered environment from which certain species have been extirpated (Kaiser et al., 2002). Despite the many efforts to deduce whether fishing activities are having a significant effect on benthic assemblages we are unable to discern the original composition of the fauna because the data gathered prior to the start of bottom fishing is nonexistent (Kaiser et al., 2002). Therefore, experimental designs must make do with comparing different intensities of fishing disturbance on already altered patches of benthos at large spatial and temporal scales to discern whether a significant difference exists. It is only through further empirical studies that the Northern Prawn Fishery in the Gulf of Carpentaria can be managed sustainably for the future.

“Such is of the course of deeds that move the wheels of the world: small hands do them because they must, while the eyes of the great are elsewhere.”

J.R.R. Tolkien

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The RV Southern Surveyor

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A preliminary Torres Strait Ecopath model: Adaptation of the northern Great Barrier Reef model

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Summary

Trawling effects on penaeid prawn communities in the Northern Great Barrier Reef were analysed by the Northern Great Barrier Reef Ecopath model by combining a generalised ecosystem template of coral reefs and in-depth annual cross-shelf surveys of the far northern Great Barrier Reef. Functional groups described in the Northern Great Barrier Reef Ecopath are based on those described for a Caribbean Reef coral system, with groups based on diet composition, body size and lifestyle (Gribble, 2003). The cross-shelf surveys provided much of the information, including the biomass of target prawns and discards, as well as fish and other non-fish taxa biomasses for the model (Gribble, 2003). Diet composition, production and consumption rates, and all data not obtained from the Great Barrier Reef surveys for use in the Northern Great Barrier Reef Ecopath model were calculated from information derived from Gulf of Carpentaria prawn predation literature (Brewer et al. 1991; Salini et al. 1990, 1992 and 1998), FISHBASE (Froese & Pauly, 1999), relevant Ecopath models (Opitz, 1996) and literature on specific Great Barrier Reef species (Gribble, 2003).

The purpose of the preliminary food web of the Torres Strait is to provide a means for examining the complexity of modelling secondary effects of benthic biomass removal on the ecosystem. The area of interest corresponds with the area where most prawn trawling effort is concentrated. Modifying the Great Barrier Reef prawn Ecopath model for the Torres Strait area results in utilising many parameters already calculated for the Great Barrier Reef, although site-specific data would be preferred. The functional groups in the Great Barrier Reef model were used as a basis for the Torres Strait food web, with additional groups relevant to the area added (Table 1). Parameters were calculated for the Torres Strait where data was available, as documented in the following sections, with all other parameters taken from the Northern Great Barrier Reef model and other tropical reef models (Table 1).

Model inputs

Primary Producers and Detritus

Benthic autotrophs
Benthic autotrophs represent the seagrass, algae and coral algal symbiont populations of the Northern Great Barrier Reef, with their combined average biomass being the largest of any other functional group in the area (Gribble, 2003). The Torres Strait food web however, has seagrass as a separate functional group, which is included in more detail below, due to Dugong dependence on seagrass for food and their importance as prawn nursery grounds. Benthic autotrophs in the Torres Strait food web therefore, represent algae and coral algal symbionts. Common algae in the Torres Strait include those from the genre *Halimeda, Caulerpa* and *Sargassum*, with the *Halimeda* genre widespread across the entire Torres Strait showing little or no substrate preference (Pitcher et al., 1992). The *Caulerpa* genre shows some preference for habitat with rubble, whereas *Sargassum* appears restricted to hard

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substrata due to the need of a solid attachment for the holdfast (Pitcher et al., 1992). As well as providing food and shelter for many organisms, as with the other algae genre, *Halimeda* also has the potential to contribute substantially to sediment carbonate due to abundant skeletons (Pitcher et al., 1992).

Table 1: Basic parameters for the Torres Strait food web using Ecopath

	GROUP NAME	BIOMASS (t·km^{-2})	PRODUCTION / BIOMASS (year^{-1})	CONSUMPTION / BIOMASS (year^{-1})
1	Cephalopods	0.328	4.590	17.550
2	Large groupers	0.035	0.370	2.300
3	Scombrids/jacks	2.024	0.720	8.900
4	Sea birds	0.015	5.400	80.000
5	Large sharks/rays	0.557	0.240	4.900
6	Small schooling fish	3.122	2.250	20.050
7	Large fish carnivore	1.780	0.960	10.960
8	Large schooling fish	0.600	1.246	12.700
9	Rock lobster*	0.680	0.450	7.400
10	*P.longistylus*	0.640	7.570	37.900
11	Other prawns	0.251	1.100	20.000
12	*P.esculentus*	0.177	7.570	37.900
13	Small fish omnivore	2.226	2.350	12.800
14	Dugong*	0.947	0.100	36.500
15	Sea turtles (large)	0.007	0.900	3.500
16	Crustaceans	2.741	3.100	20.000
17	*M.endeavouri*	0.142	7.570	37.900
18	Holothurians*	0.003	0.770	3.360
19	Echinoderms	8.404	1.500	6.000
20	Trochus*	5.670	2.500	14.000
21	Pearl shell*	0.009	1.350	23.000
22	Benthic molluscs/worms	10.972	2.900	10.000
23	Zooplankton	3.216	40.000	165.000
24	Sessile animals	30.950	0.800	12.000
25	Fish herbivore	7.116	2.730	37.450
26	Decomposer/microfauna	6.000	120.000	400.000
27	Phytoplankton	7.515	70.000	
28	Benthic autotrophs	175.515	13.250	
29	Seagrass*		9.000	
30	Detritus/discards	3.836		
31	Detritus	40.000		

* Functional groups added to the Northern Great Barrier Reef model relevant to Torres Strait

Biomass estimates of benthic autotrophs in Torres Strait were taken from estimates of the Northern Great Barrier Reef. Although seagrass is a separate functional group within the Torres Strait food web its proportionally biomass was not removed from the benthic autotroph group as there has been a likely underestimation of benthic autotroph biomass in the Northern Great Barrie Reef through sampling of only lagoon and inter-reef habitats, excluding the reef proper (Gribble, 2001). Additionally, a separate seagrass biomass in the Northern Great Barrier Reef was not stated for the Great Barrier Reef model. Thus, the biomass of benthic autotrophs in the Torres Strait food web was left as 175.109 t/km² (Gribble, 2003). **Production** estimates were also taken from the Northern Great Barrier Reef and used directly in the Torres Strait food web as 13.25 (Gribble, 2003).

Phytoplankton

Phytoplankton, minute free-floating aquatic plants, had both biomass and production estimates taken from the northern Great Barrier Reef model, for lack of more site-specific data. Estimates for the northern Great Barrier Reef came from Sorokin (1994), with **biomass** being 7.515 t/km² and **production** being 70 (Gribble, 2003). Low biomass and high production rates of phytoplankton can support large biomasses of higher trophic groups, making them important contributors to the ecosystem (Okey and Mahmoudi, 2002).

Seagrass

Seagrass in the Torres Strait is unevenly distributed; it is usually very sparse and varies in density over short distances (Pitcher et al., 1992). A survey of the Torres Strait in 1989 identified two species groups, *Halophila* species and strap-like forms including *Cymodocea serrulata, C. rotunda, Halodule uninervis, Thalassia hemprichii,* and *Enhalus acaroids* (Pitcher et al., 1992). Seagrass beds of the Torres Strait are vitally important in supporting the largest population of threatened dugong in the world. Seagrass beds also provide nursery grounds for many organisms, including juvenile Penaeid prawns. No **biomass** or **production rate** estimates for seagrass were available instead; a rough **production rate** of 9 was estimated (T. Okey pers. comm.) based on the West Florida Shelf model, with biomass left blank.

Detritus

Detritus provides energy and nutrients for many marine organisms, and can be supplied from a variety of sources. No available data on detritus **biomass** in the Torres Strait could be found, so the biomass estimate of 40 t/km² from the northern Great Barrier Reef was used (Gribble, 2003).

Detritus/Discards

Bycatch, labelled detritus/discards in the Torres Strait food web and the Northern Great Barrier Reef model, affects the ecology of an area in two main ways; first, through the removal of various species from the environment and secondly, through the provision of extra food for scavengers through discards (TSPZJA, 2005). Detritus/discards, or trawling bycatch, for the northern Great Barrier Reef consists mostly of small bottom omnivorous and herbivorous fish, as well as a small proportion of adult turtles (Gribble, 2001). The Torres Strait Bycatch Action Plan has identified bony fishes as the largest group caught as bycatch by prawn trawlers of the Torres Strait, similar to trawling of the northern Great Barrier Reef (Table 2: TSPZJA, 2005). The similar methods and prawn trawling gear utilised in the northern Great Barrier Reef and the Torres Strait, as well as the use of bycatch reduction devices and turtle exclusion devises in both fisheries means the range of organisms caught as bycatch would be similar. As such, the Torres Strait food web model used the **biomass** of 3.836 t/km² from the northern Great Barrier Reef Ecopath model.

Table 2: Bycatch species and percent of total catch for the Torres Strait prawn-trawling fishery

BYCATCH SPECIES	PERCENT OF TOTAL CATCH
Bony fish	52-69
Crabs	8
Scallops	3
Sharks and rays	1-3
Squid, tropical rock lobster, Moreton bay bugs, sponges and turtles	1
Sea snakes	0.1-0.2

Source: TSPZJA (2005)

Microbes

Decomposer/microfauna

The decomposer/microfauna functional group, incorporating single celled heterotrophs such as bacteria, ciliates and foraminiferans, is important in reintroducing dead organic matter back into the trophic cycle of reef systems (Opitz 1996). **Biomass** (6 t/km²), **production** (120/year) and **consumption** (400/year) rates for this functional group were taken directly from the Great Barrier Reef model (Gribble, 2001; 2003), although the model did not state how these values were calculated.

Plankton

Zooplankton

Zooplankton provide an important link between primary producers and higher trophic organisms through the channelling of phytoplankton and benthic autotroph energy upwards in the trophic cycle. Consisting mostly of protozoans and juvenile stages of mesozooplankton, zooplankton are also a great source of food for larval fish in spawning areas and schooling fish. The northern Great Barrier Reef Model (Sorikin, 1994) supplied **biomass** (3.216 t/km²), **production** (40), **consumption** (165) and **diet composition** (Table 3) values for the Torres Strait food web due to a lack of more relevant data.

Table 3: Diet composition of zooplankton in the northern GBR Ecopath model

PREY	PROPORTION OF DIET
Phytoplankton	0.200
Benthic autotrophs	0.800
SUM	1.000

Source: Gribble (2003)

Macro-invertebrates

Pearl shell

Pearl shells were once an important fishery in the Torres Strait and as such, are given their own functional group separate from the benthic molluscs and worms group. Although the fishery collapsed in the 1960's, consequent low biomass in the area combined with managements hope to conserve present populations for optimising farming operations (TSPZJA, 2006), provided the reason for forming a separate pearl shell group. Direct interactions with prawn trawling may be minimal, however pearl shells may be a bycatch species.

The 1989 benthic habitat surveys provided valuable density estimates of pearl shells greater than 100 mm close to the prawn trawling area of the Torres Strait (Pitcher et al., 1992). Density values were multiplied by the 1 kg average weight of pearl shells (Darren Dennis, pers. comm.) and then extrapolated to include the entire prawn fishing ground area, giving a **biomass** estimate of 0.0042 t/km². However, when balancing the model, predation upon pearl shells was greater then the biomass could support. Even after diverting some of the bivalve predation to the benthic molluscs and worms group, predation was too high. As a solution, pearl shell biomass was increased to 0.009 t/km², which may account for an underestimation of density due to inclusion of only individuals greater than 100 mm in the survey.

No data was available for the other basic parameters required by Ecopath, with infaunal bivalve values from the west Florida Shelf used instead (Okey and Mahmoudi, 2002). Infaunal bivalve values averaged small and large bivalves, including both scallops and infaunal bivalves in the functional group, which may make **production rate** (1.35), **consumption rate** (23) and **diet composition** (Table 4) inaccurate.

Table 4: Diet composition of infaunal bivalves in the West Florida Shelf Ecopath model

PREY	PROPORTION OF DIET
Phytoplankton	0.290
Benthic autotrophs	0.300
Detritus/discards	0.190
Detritus	0.200
SUM	1.000

Source: Okey and Mahmoudi (2002)

Benthic molluscs/worms

The benthic molluscs/worms functional group was incorporated in the Torres Strait food web directly from the northern Great Barrier Reef model (Gribble, 2003, 2001), including estimates of **biomass** (10.972 t/km²) from benthic dredge data (Poiner et al., 1998), **production rate** (2.9), **consumption rate** (10) and **diet composition** (Table 5). However, the functional group includes worms, gastropods and bivalves and may be too aggregated for the purpose of examining secondary effects of trawling on the ecosystem. This is particularly true due to the great variations in lifestyle and trawling vulnerability of organisms in the group. Additionally, the aggregation of these groups caused too much predation for the biomass to support, resulting in reduction of predation rates to create a balanced model.

Table 5: Diet composition of benthic molluscs/worms in the northern GBR Ecopath model

PREY	PROPORTION OF DIET
Crustaceans	0.010
Echinoderms	0.002
Benthic molluscs/worms	0.065
Zooplankton	0.013
Sessile animals	0.050
Decomposer/microfauna	0.170
Phytoplankton	0.086
Benthic autotrophs	0.179
Detritus	0.425
SUM	1.000

Source: Gribble (2003)

Mega-invertebrates

Sessile animals

Sessile animals including porifera, cnidarian, bryozoan and ascidiacea, are a broad group that should be split to capture important processes and interactions when modelling secondary effects of trawling on the Torres Strait ecosystem. Such a broad group resulted in biomass levels too low to support the high predation rates in the model. Splitting the group and directing predation rates to more specific groups may reduce this problem. Having said that, the group has remained aggregated for the Torres Strait food web due to the lack of specific data needed by Ecopath. As such, **biomass** (30.95 t/km²) sourced from benthic dredge samples (Poiner et al., 1998), **production rate** (8), **consumption rate** (12) and **diet composition** (Table 6) were all taken from the northern Great Barrier Reef model (Gribble, 2001, 2003).

Table 6: Diet composition of sessile animals in the northern GBR Ecopath model

PREY	PROPORTION OF DIET
Zooplankton	0.025
Decomposer/microfauna	0.019
Phytoplankton	0.220
Benthic autotrophs	0.651
Detritus	0.085
SUM	1.000

Source: (Gribble, 2003)

Echinoderms
Echinoderms for the northern Great Barrier Reef model include asteroidea, ophiuroidea, echinoidea, and holothurians, with crinoids excluded following the logic of Opitz (1993) as cited in Gribble (2001). The Torres Strait food web used the northern Great Barrier Reef parameter estimations for **biomass** (8.404 t/km²), **production rate** (1.5), **consumption rate** (6) and **diet composition** (Table 7). However, site-specific information is necessary for accurate values due to the separation of holothurians from the echinoderm functional group of the food web, as well as the specific purpose of the Torres Strait model to examine secondary effects of trawling.

Table 7: Diet composition of echinoderms in northern GBR Ecopath model

PREY	PROPORTION OF DIET
Cephalopods	0.003
Crustaceans	0.002
Echinoderms	0.059
Benthic molluscs/worms	0.020
Zooplankton	0.053
Sessile animals	0.050
Decomposer/microfauna	0.149
Phytoplankton	0.003
Benthic autotrophs	0.554
Detritus	0.107
SUM	1.000

Source: Gribble (2001)

Holothurians
The East Torres Strait beche-de-mer fishery is important to Torres Strait islanders as a source of income however, in recent years the most valuable and highly fished species have been protected due to overexploitation (Skewes et al., 2004). The important role holothurians play in recycling nutrients back into the system, as well has their economic importance to islanders formed the basis of separating holothurians from the echinoderm functional group.

Production rate in mass-balanced models such as Ecopath represent the total mortality of each group. The Caribbean model calculated total mortality of a fished holothurian population at 0.6, double that of an unfished population (Pauly, 1993). The fished holothurian value was initially used in the Torres Strait food web however, during the model balancing process the production rate of holothurians in Torres Strait was increased from 0.6 to 0.77. The increased production rate was based on available site-specific natural mortality estimates in the range 0.6 to 1.0, with 0.8 stated as a reasonable overall estimate of natural mortality for all species combined (Skewes et al., 2004). Given the assumption that total mortality is twice that of natural mortality (Pauly et al., 1993), the production rate of Torres Strait holothurians may be even higher than the 0.77 stated in the Torres Strait food web.

No site-specific data was available for **consumption rate** of holothurians, so a value of 3.36 was used from the Northern Great Barrier Reef model, which obtained its information from the Caribbean coral reef (Gribble, 2003; Opitz, 1996).

The most recent survey of holothurian stock size, distribution and fishery status in Torres Strait provided detailed information regarding **biomass** in the area for use in the food web. Mean density (147.12 per ha) multiplied by the mean weight (0.00126395 tonnes) of all commercial species in the Eastern Torres Strait beche-de-mer fishery gives a total average biomass of 0.002 t/km² (Skewes et al., 2002). **Diet composition** was available for only one target species of the Torres Strait beche-de-mer fishery, *Stichopus tremulus* (Table 8) and is assumed to be similar to all other commercial species (Hudson et al., 2004).

Table 8: Diet composition of a commercial holothurian in the Torres Strait (*Stichopus tremulus*)

PREY	PROPORTION OF DIET
Phytoplankton	0.01
Benthic autotrophs	0.13
Detritus	0.86
SUM	1.00

Source: Hudson et al. (2004)

Trochus

Trochus was historically important between 1920 and 1959 and again in the 1980's but declining overseas markets has resulted in the small, commercial and subsistence fishery of today. Despite the low participation levels in this fishery, it has been included as its own functional group.

Production (2.5) and **consumption rates** (14) of trochus were unavailable; consequently, values were taken from the small gastropod functional group of the Caribbean coral reef system (Opitz, 1993). Calculations of **biomass** (5.6 t/km²) for trochus involved estimating the average weight of trochus (Table 9) and then multiplying that average by the average 3.4 trochus per 120 m² density from surveys of the Torres Strait Bourke Isles (CMR, 1999; Long et al., 1993). **Diet composition** was also unavailable for trochus of the Torres Strait, with those values taken from the small gastropod functional group of the Galapagos Ecopath model (Okey et al., 2004).

Table 9: Total numbers, total weight and individual weight of trochus caught in the Bourke Isles, Torres Strait

REEF	TOTAL NUMBERS	TOTAL WEIGHT (TONES)	WEIGHT OF AN INDIVIDUAL (TONES)
Ashmore	21767.00	4.40	0.0002
Browse	0.00	0.00	0.0000
Cartier	222.00	0.00	n/a
Hibernia	0.00	0.00	0.0000
Scott Nth	0.00	0.00	0.0000
Scott Sth	718.00	0.10	0.0002
Seringapatam	0.00	0.00	0.0000
Total	22706.00	4.60	Average =0.0002

Source: adapted from CMR (1999)

Table 10: Diet composition of small gastropods in the Galapagos Ecopath model

PREY	PROPORTION OF DIET
Echinoderms	0.033
Benthic molluscs/worms	0.039
Sessile animals	0.002
Benthic autotrophs	0.456
Detritus	0.470
SUM	1.000

Source: Okey et al. (2004)

Crustaceans

Crustaceans in the northern Great Barrier Reef model, excluding prawns, were incorporated into the Torres Strait food web with **biomass** (2.741 t/km²), **production rate** (3.2) and **consumption rate (**20), and **diet composition** (Table 11) values used directly from the Great Barrier Reef model. Crustacean biomass for the northern Great Barrier Reef was derived from benthic dredge and fish-trawl sampling data, with production and consumption values based on literature relating to prawn predation in the Gulf of Carpentaria and previous Ecopath models (Gribble, 2004, 2001).

Table 11: Diet composition of crustaceans in the northern GBR Ecopath model

PREY	PROPORTION OF DIET
Cephalopods	0.002
Small schooling fish	0.005
Small fish omnivore	0.002
Crustaceans	0.030
Echinoderms	0.030
Benthic molluscs/worms	0.050
Zooplankton	0.120
Sessile animals	0.095
Fish herbivore	0.001
Decomposer/microfauna	0.055
Phytoplankton	0.113
Benthic autotrophs	0.200
Detritus	0.297
SUM	1.000

Source: Gribble (2003)

Prawns

Prawn **biomass, production, consumption** and **diet composition** (Table 12) values were all estimated from prawn surveys, literature on prawn predation in the Gulf of Carpentaria and previous Ecopath models (Gribble, 2003, 2001). Both prawn trawl gear and dredge gear were utilised to survey all elements of the community including those prawns that stay buried in the substrate and those that emerge into the water column (Gribble, 2001). Biomass estimates for prawn species that occurred in both the northern Great Barrier Reef model and the Weipa model were very similar. However, production and consumption rates varied substantially bringing attention to the fact that site-specific data may be necessary for the Torres Strait.

Table 12: Diet composition of prawns in the northern GBR Ecopath model

	PROPORTION OF DIET			
PREY	**Other Prawns**	***P.longistylus***	***P.esculentus***	***M.endeavouri***
Cephalopods	0.002	0.000	0.000	0.000
large fish carnivore	0.005	0.000	0.000	0.000
Other prawns	0.000	0.000	0.007	0.000
Small fish omnivore	0.023	0.000	0.000	0.000
Crustaceans	0.020	0.050	0.080	0.100
Echinoderms	0.070	0.000	0.000	0.000
Benthic molluscs/worms	0.200	0.190	0.120	0.000
Zooplankton	0.090	0.080	0.163	0.100
Sessile animals	0.040	0.200	0.140	0.100
Decomposer/microfauna	0.055	0.100	0.000	0.100
Phytoplankton	0.035	0.000	0.000	0.000
Benthic autotrophs	0.025	0.000	0.000	0.000
Detritus/discards	0.203	0.180	0.258	0.400
Detritus	0.232	0.200	0.232	0.200
Sum	1.000	1.000	1.000	1.000

Source: Gribble (2001)

Rock lobster

Rock lobsters are a separate functional group in the Torres Strait food web due to the important commercial value of the fishery in the area. Site specific data for rock lobsters of Torres Strait was only available for the calculation of biomass, with **production** and **consumption** rates taken from the Spiny lobster of Caribbean coral reefs (Opitz, 1996), and **diet composition** (Table 13) from the lobster fisheries of the Galapagos (Martínez, 2000). **Biomass** calculations of rock lobsters within the study area came from Ye et al. (2004). Two populations of rock lobster were identified, the recruiting 1+ year class and the fished 2+ year class, each making up a different proportion of the entire population (Ye et al., 2004). Rock lobsters have not been separated into two-year classes in the model, therefore biomass estimates are averaged across the population. Within a 19045 km^2 area, the report estimates an average 7.25 million 1+ lobsters and 1.75 million 2+ lobsters.

Ye et al., (2004) surveyed rock lobsters in the western Torres Strait in May/June of 2002 collecting measurements of total weight, tail length and equivalent carapace length. Each frequency distribution for the measurements displayed two distinct peaks that appear to represent the two age-classes. Comparing the size-frequency distribution of carapace length with a graph of spatial and temporal differences in the size of each year class confirms that the peaks are depicting age-classes. The peak of each age class in the total weight frequency distribution provided average weights which, when combined with the average number of lobsters of each age class, allowed total biomass to be calculated at 0.68 t/km^2.

Table 13: Diet composition of lobsters in the Galapagos Ecopath model

PREY	**PROPORTION OF DIET**
P.longistylus	0.090
Other prawns	0.013
P.esculentus	0.078
M.endeavouri	0.060
Echinoderms	0.038
Pearl shell	0.002
Benthic molluscs/worms	0.055
Sessile animals	0.300
Benthic autotrophs	0.103
Detritus	0.261
SUM	1.000

Source: Martínez (2000)

Cephalopods

Cephalopod **biomass** (0.328 t/km²) in the northern Great Barrier Reef was estimated by summating the benthic dredge and fish-trawling data of the region used to examine different mechanisms of cephalopod communities (Gribble, 2001; Poiner et al., 1998). **Production rate** (4.59), **consumption rate** (17.55) and **diet composition** (Table 14) came from previous Ecopath models, such as that of the Caribbean coral reef system (Gribble, 2003). Lack of data for the Torres Strait resulted in the northern Great Barrier Reef values being used here.

Table 14: Diet composition of cephalopods in the northern GBR Ecopath model

PREY	**PROPORTION OF DIET**
Cephalopods	0.100
Small schooling fish	0.065
Large fish carnivore	0.020
Large schooling fish	0.030
Other prawns	0.010
Small fish omnivore	0.100
Crustaceans	0.090
Benthic molluscs/worms	0.310
Zooplankton	0.245
Detritus/discards	0.030
SUM	1.000

Source: Gribble (2003)

Fish

Fish species lists from the Northern Great Barrier Reef survey and from the Caribbean Reef system were compared for the northern Great Barrier Reef model, with matching species or species performing the same function, aggregated based on diet, body size and lifestyle (Gribble, 2001). Fish populations are likely to be similar in the Torres Strait, so functional groups were not altered, with **biomass, production** and **consumption rates**, and **diet** (Table 15) matrixes taken directly from the Northern Great Barrier Reef model. Size distinctions of large and small fish are based on large fish being greater than 30 cm maximum size (Gribble, 2001).

Table 15: Diet composition of all fish groups in the northern GBR Ecopath model

	PROPORTION OF DIET					
PREY	**Large sharks/rays**	**Small schooling fish**	**Large fish carnivores**	**Large schooling fish**	**Small fish omnivore**	**Fish herbivore**
Cephalopods	0.024	0.000	0.005	0.000	0.000	0.000
Large groupers	0.001	0.000	0.000	0.000	0.000	0.000
Scombrids/jacks	0.030	0.000	0.020	0.001	0.000	0.000
Sea birds	0.003	0.000	0.000	0.000	0.000	0.000
Large sharks/rays	0.010	0.000	0.000	0.000	0.000	0.000
Small schooling fish	0.075	0.004	0.030	0.040	0.002	0.000
large fish carnivore	0.150	0.000	0.005	0.000	0.000	0.000
Large schooling fish	0.005	0.000	0.012	0.002	0.000	0.000
P.longistylus	0.015	0.000	0.000	0.000	0.005	0.000
Other prawns	0.005	0.001	0.000	0.000	0.000	0.000
P.esculentus	0.035	0.000	0.000	0.000	0.010	0.000
Small fish omnivore	0.084	0.000	0.010	0.000	0.015	0.000
Crustaceans	0.060	0.005	0.020	0.005	0.045	0.000
M.endeavouri	0.035	0.000	0.000	0.000	0.007	0.000
Echinoderms	0.021	0.000	0.130	0.000	0.060	0.000
Benthic molluscs/worms	0.100	0.000	0.070	0.100	0.200	0.001
Zooplankton	0.000	0.974	0.130	0.720	0.108	0.004
Sessile animals	0.011	0.006	0.070	0.000	0.040	0.000
Fish herbivore	0.020	0.000	0.259	0.000	0.016	0.000
Decomposer/ microfauna	0.000	0.002	0.000	0.000	0.000	0.000
Phytoplankton	0.000	0.002	0.010	0.000	0.060	0.001
Benthic autotrophs	0.000	0.006	0.199	0.015	0.216	0.988
Seagrass	0.000	0.000	0.000	0.000	0.000	0.000
Detritus/discards	0.300	0.000	0.000	0.117	0.000	0.000
Detritus	0.016	0.000	0.030	0.000	0.216	0.006
Sum	**1.000**	**1.000**	**1.000**	**1.000**	**1.000**	**1.000**

Source: Gribble (2001)

Seabirds

Although seabirds are likely to have little, if any, direct involvement in exploring the effects of trawling on removal of benthic biomass, they are included in the food web due to their predation of trawling bycatch, which is another aspect of management. Observations have identified crested terns (*Sterna bergii*), frigate birds (*Frigata ariel, F. minor*) and brown boobies (*Sula leucogaster*) to be the main sea bird populations of the Torres Strait that may be affected by trawling operations due to discarding of bycatch (TSPZJA, 2005). **Biomass** (0.015 t/km²) values of seabirds in the northern Great Barrier Reef came from the Caribbean coral reef system and **production rate** (5.4), **consumption rate** (80) and **diet composition** (Table 16) were sourced from here also (Gribble, 2001).

Table 16: Diet composition of seabirds in the northern GBR Ecopath model

PREY	PROPORTION OF DIET
Scombrids/jacks	0.100
Small schooling fish	0.200
large fish carnivore	0.090
Large schooling fish	0.150
P.longistylus	0.005
Other prawns	0.007
P.esculentus	0.020
Small fish omnivore	0.041
M.endeavouri	0.030
Zooplankton	0.050
Fish herbivore	0.002
Detritus/discards	0.305
SUM	1.000

Source: Gribble (2001)

Reptiles

Sea turtles

Biomass (0.007 t/km²) of sea turtles for the northern Great Barrier Reef came from the Caribbean coral reef system, stated to be consistent with information gathered from the Great Barrier Reef surveys (Gribble, 2001). Estimations of **production** (0.9), **consumption** (3.5) **and diet composition** (Table 17) also came from the Caribbean coral reef system due to a lack of such information in the northern Great Barrier Reef. Initial searching has found data lacking for the Torres Strait as well, resulting in the use of northern Great Barrier Reef values for the Torres Strait food web.

Table 17: Diet composition of sea turtles in the northern GBR Ecopath model

PREY	PROPORTION OF DIET
P.esculentus	0.010
Crustaceans	0.050
M.endeavouri	0.055
Echinoderms	0.030
Benthic molluscs/worms	0.040
Sessile animals	0.010
Benthic autotrophs	0.805
SUM	1.000

Source: Gribble (2001)

Mammals

Dugong

Dugong occur along the Australian coast from Shark Bay in Western Australia to Moreton Bay in Queensland, with Torres Strait supporting the largest and potentially most important population (Marsh et al., 2004a). The significance of this population in generating an important traditional fishery of meat and oil for Torres Strait islanders is the reason for including dugong as a functional group. Unfortunately, **production** and **consumption rates** of Dugong are unknown and these values had to come from the Manatee populations of the West Florida Shelf model (Okey and Mahmoudi, 2002). Manatees and dugong belong to the order Sirenia with both feeding on seagrasses. Their similarities are potentially so close that the production rate of 0.1 year $^{-1}$ and the consumption rate of 36.5 year $^{-1}$ for Manatees would be similar to those of Dugong.

Marsh et al., (2004a) carried out aerial surveys of most of Torres Strait to assess Dugong distribution and abundance, gathering information used to estimate biomass for the model. The survey broke the region up into "blocks" for assessment, each covering different areas and assessed at varying intensities. Using Arcview 3.1, the area surveyed in the '2B survey block' overlapping with the area of interest in the model was calculated at 9.22 km². The entire area of interest is however 9.716 km², meaning that some of it was unsurveyed by Marsh et al., (2004a, Figure 1).

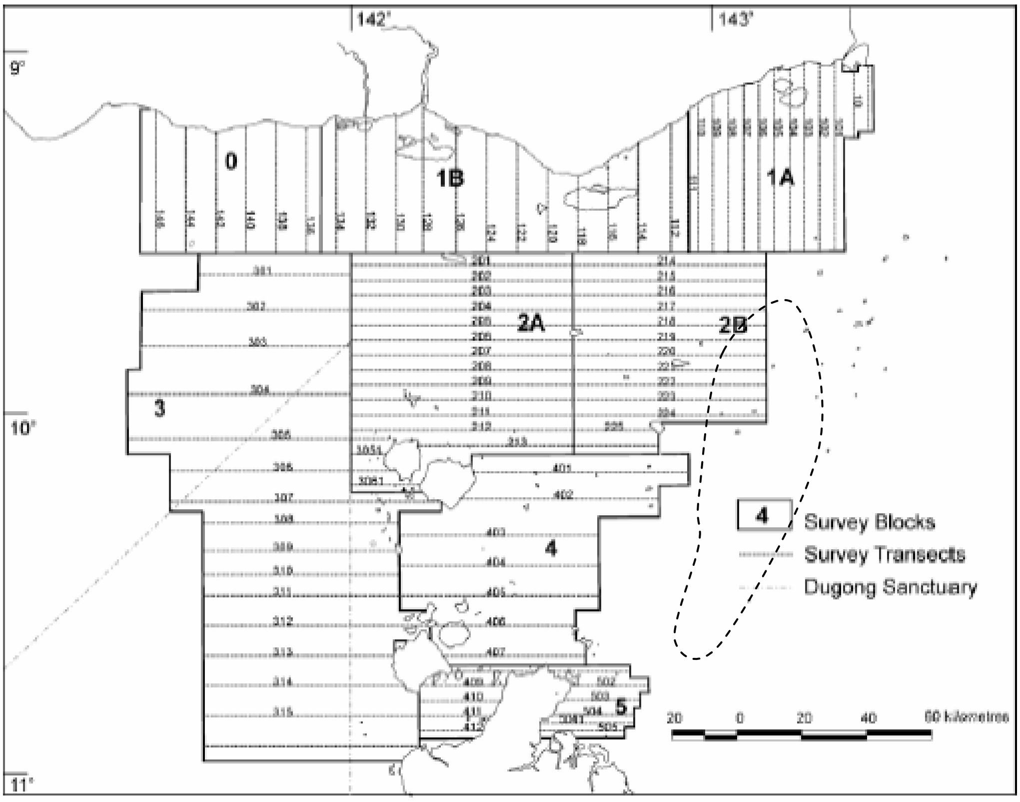

Figure 1: Map of region surveyed by Marsh et al. (2004a) showing survey blocks and the area of interest for the Ecopath model (dashed line)

With only 8.43% of survey block 2B surveyed; it is likely that the two dugong sighting in the overlap area may only be a fraction of what was there. The estimate of dugong abundance below may be more accurate for the overlap area.

2 dugong * 0.0843 (% of 2B surveyed) = 23.72 dugong

Extrapolated dugong numbers from the overlap area to the entire area of interest gives an estimated 24.9 dugong, each with an average weight of 370 kg (Gaus et al., 2004). Calculations of **biomass** using the estimated dugong abundance and the average dugong weight gave a result of 0.9472 t/km² for the area of interest. There are suggestions of large-scale dugong movements within and between even very large survey regions, like those of the Torres Strait, a factor that may alter the accuracy of the biomass estimate (Marsh et al., 2004a).

Dugong predominantly feed on the seagrass *Halophila ovalis*, as well as seagrasses from the genera *Halodule* however; there is evidence that they also include small amounts of algae in their diet (Marsh et al., 2004b). When seagrass beds dieback or are completely lost, some dugongs have been found to consume unusually high proportions of algae and the less preferred seagrass species (*Thalassia, Cymodocea* and *Enhalus*) (Marsh et al., 2004b). Under normal conditions, consumption of algae may be incidental or almost non-existent. Due to the minuscule amounts of algae likely consumed by dugong in the Torres Strait, any changes in abundance of either population would have little or no affect on the other. For this reason, algae were not included in dugong **diet** making seagrass the only prey item of Torres Strait dugong.

Fisheries

Prawn trawling

The Torres Strait prawn fishery operates out of the eastern part of the Torres Strait and is considered the most valuable commercial fishery in the area. Commercial species of the fishery include *Metapenaeus endeavouri, Penaeus esculentus*, and *P. longistylus*, with the area of interest for the food web based on areas where effort for this fishery is greatest. Fishery landings for the Torres Strait food web were calculated from final catch data of 2002-2003 (Table 18) in the area of 15 655 km^2 stipulated by Skewes et al., (2004) to be the eastern Torres Strait. Given that the area of interest is based on distribution of effort within the fishery, data more specific to that area is likely to be available and should replace these initial estimations.

Table 18: Total catch and catch in t/km^2/year for commercial prawn species in the Torres Strait

PRAWN SPECIES	**2002-2003 FINAL CATCH (t)**	**FISHERY LANDINGS IN EASTERN TORRES STRAIT (t·km^{-2}·year^{-1})**
Metapenaus endeavouri	750	0.048
Penaeus esculentus	665	0.043
Penaeus longistylus	121	0.008
Other prawns*	5	0.0003

Source: Adapted from TSPZJA (2006). Final catches have been divided by 15655 km^2 to obtain fishery landings in the eastern Torres Strait
*ABARE (2003)

Finfish

The finfish, or line, fishery of the Torres Strait is restricted to the eastern and central strait, with no line fishing allowed west of 142º30'19''E. The fishery is split into islander and non-islander commercial fishers, with only the non-islanders required to keep logbook records. Records of reef fish caught by islanders and then purchased by non-islander or community operated freezers are used to estimate islander commercial catches in the absence of logbooks (Mapstone et al., 2003). The total median commercial catch by islanders in the late 1990's (Table 19) was added to the 57.1 tonnes recorded in 2000 (Mapstone et al., 2003) by non-islander commercial fishermen to obtain an estimated overall fleet catch. No specific areas of finfish fishing was stipulated, so the east Torres Strait area stated by Skewes et al., (2004) of 15 655 km^2 was used to calculate the fleet landing of 0.0045 t/km^2/year. Net fishing also occurs in the Strait, but in small numbers and is restricted to islanders fishing in the outside but near region.

Table 19: Catch of finfish in the Torres Strait in the late 1990's

PLACE	**CATCH (TONNES)**	**MEDIAN CATCH (TONNES)**
York Island	1-2	1.5
Darnley Island	4.5-5.5	3.5
Murray Island	8-10	9
Total		14

Source: adapted from Mapstone et al. (2003)

Rock lobster
Commercial fishing in the Torres Strait tropical rock lobster fishery is for a single species of rock lobster, *Panulirus ornatus*. Lobsters are taken via hand or short hand spear by free divers or hookah (surface supplied air) divers working from dinghies. Commercial fishing occurs all year round, with the exclusion of October and November where the fishery is closed. There is also ban on trawlers taking rock lobsters to prevent pressure on the fishery from an outside source (TSPZJA, 2006).

In 2000, the commercial rock lobster catch was 210 t tail weight or 500 t live weight (tail weight is 42% of total live weight: Ye et al., 2004). Based on a map of the rock lobster fishing grounds, an estimated area of 11 265 km² was used in combination with the 500 t live catch of the fishery to calculate landings of 0.044 t/km²/year.

Beche-de-mer
The Torres Strait beche-de-mer fishery is an important source of income for Torres Strait islanders with 178 Traditional Inhabitant vessels presently licensed for the fishery, in addition to the one non-islander license holder. Since 1996 catches have averaged about 80 t per year with an approximate value of $250,000. In relation to the East Torres Strait fishery, recent years have shown quotas not reached, fishermen reporting decreased catch rates, and observed declining catch per unit effort (CPUE) (Skewes et al., 2004). Additionally, surveys have revealed changes in the distribution of sand fish on the northern section of Warrior Reef, the possible cause of which is believed to be alterations of the environment in the area and/or illegal fishing (Skewes et al., 2004). Previously high valued species have also now been made off limits to commercial fishermen as a result of overexploitation. Commercial holothurians are collected via free diving from dinghies or by hand collection along reefs exposed at low tide (TSPZJA, 2006.).

The reported catch of 2001/2002 for all commercial holothurians combined was 122.1 t in the east Torres Strait, an area of 15 655 km² (Skewes et al., 2004). These values were used to calculate the 0.0078 t/km²/year landing of holothurians in the east Torres Strait.

Trochus
The Torres Strait Trochus (*Trochus niloticus*) fishery is a small, single-species commercial and subsistence fishery providing income for islander women and children despite the large decline in overseas market demand in recent years. Participation within the fishery is low, although small catches are taken from the central and eastern Islands of the Strait. Collection of trochus is usually via free diving, though SCUBA and hookah are permitted and reef top collection is possible at low tide.

Records from the prime mother ship transporting trochus out of the Torres Strait show approximately 24 t harvested in 1999, 1.8 t in 1998, 17 t in 1997 and 9.35 t in 1996, all well below the 150 t total allowable catch set for the fishery (TSPZJA, 2006; D´Silva, 2001). These records are for trochus harvesting of the entire Torres Strait Protected Zone, an area of roughly 30 000 km² that gives an estimated 0.0008 t/km²/year when the 24 t catch of 1999 is used in the calculation.

Pearl shell
The main target species of the Torres Strait pearl shell fishery are primarily the gold-lipped pearl shell (*Pinctada maxima*) and secondly the black-lipped pearl shell (*Pinctada margaritifera*) (TSPZJA, 2006). The fishery collapsed in the 1960's with no recorded catches made by the fishery in recent years (ABARE, 2003). Restrictions on the fishery are aimed at promoting the harvesting of pearl shells for farming purposes and protecting the stock currently in the Strait.

Spanish mackerel
Operating primarily in the eastern Torres Strait, the Spanish mackerel fishery targets the narrow-barred Spanish mackerel (*Scomberomorus commerson*) and, as of 1999, target species were expanded to include school mackerel (*Scomberomorus queenslandicus*), grey mackerel (*Scomberomorus semifasciatus*), spotted mackerel (*Scomberomorus munroi*) and shark mackerel (*Grammatorcynus bicarinatus*) (TSPZJA, 2006). Small numbers of commercial operators make the majority of Spanish

mackerel landings in the Strait using trolls, a baited line trailed behind a slowly moving boat. Islander participation within the fishery is unknown but considered low.

Catch of Spanish mackerel made per square kilometer in a year (0.177 $t/km^2/year$) was calculated using the 278 t reported catch of 2000-2001 (ABARE, 2003) and the15655 km^2 area of the east Torres Strait fishery (Skewes et al., 2004).

Dugong
Under the Torres Strait Treaty between Australia and Papua New Guinea, Torres Strait islanders are entitled to hunt dugong as part of their traditional way of life and livelihood (Marsh et al., 2004) and as such, the fishery is restricted to catching of dugong for those purposes only, no dugong meat or dugong products may be sold. In 1994 approximately 860 dugong were taken in the Torres Strait Protected Zone, with the majority of those catches occurring in the western region (TSPZJA, 2006). Number of dugongs caught was converted to total catch weight by multiplying 860 and the 0.370 t average weight of a dugong. These calculations gave a catch weight of 318.2 t, which was then divided by the 30 000 km^2 area of the Torres Strait Protected Zone to convert catch into tonnes per square kilometre per year (0.012 $t{\cdot}km^{-2}{\cdot}year^{-1}$).

Turtle
Turtles are taken in all areas of the Torres Strait Protected Zone, but landings are restricted to islanders for traditional purposes. Sale of turtle meat and products are strictly prohibited and only traditional fishing methods can be used. Islanders primarily catch green turtles with little exploitation of other turtle stocks. Calculations using the average weight of a green turtle (0.1 t), total landings in a year (2600 turtles) and the 30000 km^2 area of the Torres Strait Protected Zone results in 0.0087 $t/km^2/year$ of turtles caught in the Torres Strait for traditional purposes (TSPZJA, 2006).

Considerations

A good deal of the information specific to the Torres Strait, particularly fisheries data, stated the east Torres Strait as the source of information, with no mention of the dimensions of the area. Instances where this occurred assumed the same area value of the east Torres Strait used by Skewes et al., (2003) in their study of holothurians. Additionally, many functional groups, such as sessile animals, are too aggregated for exploring the effects of benthic biomass removal on the Torres Strait ecosystem, with many important processes and interactions likely to be lost. At this point there is not enough data to split the groups, so they have remained aggregated until such information can be found or estimated from other sources. It is important that this is done at some stage to properly examine the secondary effects of trawling on the benthic biomass of Torres Strait.

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The impact of climate change on macroalgae and kelp forests of southern Australia

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Introduction

Marine macroalgae are distributed throughout Australia's southern temperate bioregions, and some can be found further north in intertidal and sub-tidal rocky habitats, especially along west and east coasts. This macroalgal flora exhibits high levels of endemism because of its isolation from other continents and due to limited connectivity among suitable habitats (Phillips, 2001). A rich diversity of macroalgal species evolved in Australia, particularly on the southern continental shelf. This shelf is one of the largest in the world and it encompasses a variety of habitats suitable for macroalgae such as rocky shores and subtidal reefs, sheltered embayment's, offshore islands, and subtidal rocky platforms that are exposed periodically by shifting sand (Phillips, 2001).

Oceanographic conditions shape the distributions of macroalgal assemblages in the context of this substratum availability. For instance, zones of upwelling and cool nutrient-rich waters promote algal productivity. Southward moving currents such as the East Australia Coast current and Zeehan current (a continuation of the western Leeuwin current) bring warm water from the north creating complex ocean structure along east and west coasts and a variety of micro habitats of varying temperatures (Edyvane, 2003). The relative stability of these systems enables macroalgae and associated assemblages to persist. Changes in these oceanographic conditions such as intensification of these currents, increasing sea surface temperatures, changing nutrient levels, changes in zonal wind (and upwelling) patterns, changes in low pressure systems, increases in UV radiation, and rising sea level are likely to modify these assemblages. Indeed, such changes will likely degrade these assemblages (in the context of the ecosystem services they provide presently), as many of them are bounded geographically, e.g., to the south.

Although the ecology and dynamics of the bulk of Australia's macroalgal species is poorly known, certain species such as *Macrocystis pyrifera, Macrocystis angustifolia* and *Ecklonia radiata* have received a high level of attention due to their apparently higher economic and ecological importance. These three species are thought to facilitate the presence of a broad suite of organisms because they provide primary production that is utilized by the faunal assemblage and a large variety of micro habitats, shelter, and stable surface amongst their foliage and branching holdfasts for a wide array of invertebrate and fish species (O'Hara, 2001). Macroalgae also manipulate the surrounding environment by altering light availability, sediment, and hydrological flows (Wernberg *et al.*, 2005) and can be thought of as foundation species within their environment.

Certain macroalgal species are potentially useful indicators of climate change impacts on temperate marine environments because cool sea surface temperatures and adequate nutrients are required for these important species to survive and reproduce. This paper is focused largely on *Macrocystis pyrifera* (giant kelp) because of its potential usefulness as an ecological indicator, its role as a foundation species, and because much research has been conducted on this species due to its high economic and visual appeal. We also occasionally refer to other macroalgal species in cases where available information is pertinent to the question of climate change impacts.

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Species and Distribution

Most of the visually dominant macroalgae species on temperate Australia's rocky coastlines include brown algae (Phylum: *Phaeophyta*). The distribution of each species is dictated by a combination of temperature, substratum preferences, intra- and inter-specific competition, and disturbance history. The combination of these factors produces an integrated mosaic of algal species. Kelp forests are not single species environments; they are a complex and dynamic system that incorporates a vast array of stationary and mobile organisms. Like terrestrial forests, canopies overshadow a complex structure of understorey levels consisting of a number of other specialised kelp species. These can include erect understories where fronds are held vertically above the rock layer, prostrate canopies where the fronds lie horizontal to the substratum, algal turfs covering rocky substrata, and encrusting coralline algae (Dayton, 1985).

Tasmania's coastline harbours *M. pyrifera* (Figure 1)*, Xiphophora gladiata, Lessonia corrugate*, *Caulocystis cephalornithos,* and *M. angustifolia* (Figure 2) around northern Tasmania and Victoria. *Durvillaea potatorum* (Figure 3) grows in all southern intertidal zones (Clayton, 1990; Sanderson, 1997). *Sargassum* spp, *Caulerpa* spp and *Ecklonia radiata* (Figure 4) survive throughout most regions of the subtidal habitats; and *Cystophora* spp and *Acrocarpia* spp (Figure 5) are found exclusively along the southern coastline (Sanderson, 1997). Womersley's (1990) geographic analysis of macroalgae along the southern coast indicated that 32% of the total number of species are generally found along the southern coast; 38% in eastern South Australia, Victoria, and Tasmania; 15% along the western coast; and only 13% are widespread. The distinct physiographic conditions that led to the formation of such a diverse array of unique macroalgal assemblages underpin one of the highest levels of macroalgal species richness and endemism in the world (Phillips, 2001).

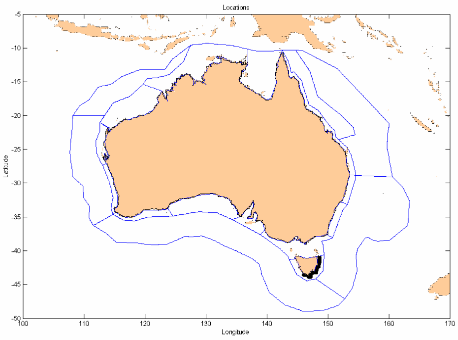

Figure 1: Distribution of *Macrocystis pyrifera*.

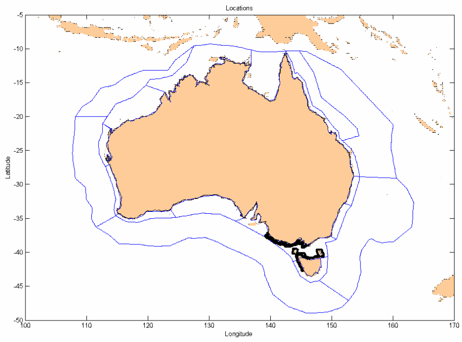

Figure 2: Distribution of *Macrocystis angustifolia* .

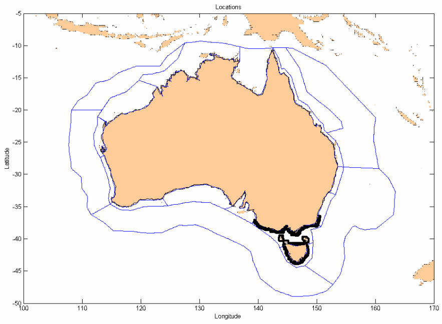

Figure 3: Distribution of *Durvillaea potatorum.*

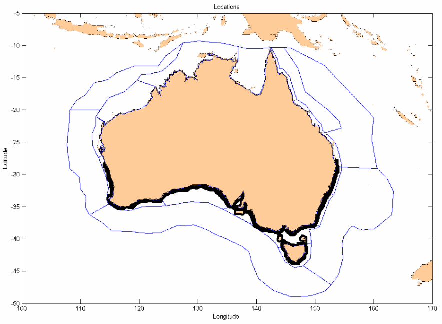

Figure 4: Distribution of *Ecklonia radiata.*

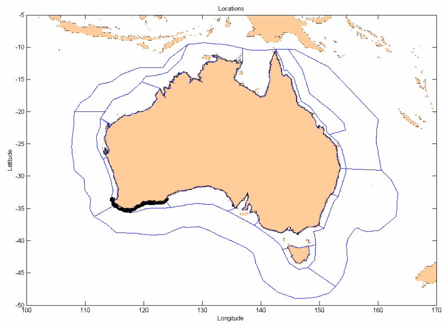

Figure 5: Distribution of *Acrocarpia robusta.*

Figure 6 illustrates the CONCOM regions that were proposed in 1985—determined according to the ocean temperatures and substratum characteristics which largely govern macroalgal assemblages. The Bass Strait and the Tasmanian coast feature cool waters (5-15ºC) and high annual temperature variation (up to 10ºC). The Lower West Coast, Lower East Coast, and Central East Coast zones offer moderate to warm temperatures (12-25ºC), and the remaining regions experience fluctuations between these two temperature zones (Sanderson, 1997).Very little specific information is available on the vulnerability of climate change impacts on the macroalgal species in these regions, and at different canopy levels.

Figure 6: CONCOM zones that were proposed in 1985 and determined largely according to ocean temperatures (Sanderson, 1997).

Processes affecting distribution

Oceanographic and atmospheric processes including currents and low pressure systems have shaped the development of kelp diversity and distribution patterns in Australia. Prominent currents, for example, affect the temporal and nutritional conditions of the Australian coast line. The Leeuwin current flowing south along the west coast shelf of Australia turns at Cape Leeuwin and enters the Great Australian Bight as the Zeehan current. This warm, low nutrient current has inhibited ice growth during the winter months on the cool south coast over the past thousands of years (Herzfeld, 1997). The Zeehan current is also responsible for the dispersal and establishment of macroalgae species with tropical affinities into the cool southern waters of the Great Australian Bight (Phillips, 2001). These warm currents have allowed complex macroalgal communities to develop by sustaining favourable oceanic conditions on the West Coast of Australia and the Great Australian Bight.

The East Australia Current however does not extend into the Great Australian Bight; it normally veers offshore at New South Wales, and only its warm eddies bring warm, saline, nutrient poor waters as far south as Tasmania, especially during summer months when it is strongest. A continual mixing of ocean mediums occur between the Leeuwin Current and the East Australia Current creating a variety of temperature and nutrient gradients that are suitable for diverse macroalgal growth across Australia's southern coastline (Edyvane, 2003). Oceanic upwellings and strong westerly winds drive colder waters up from the south, bringing nutrients essential for macroalgae (Edyvane, 2003). A smaller amount of nutrients is gained from the land via rainfall runoff.

El Niño Southern Oscillation (ENSO) is a naturally occurring climatic event that affects atmospheric and oceanographic processes and in turn changes distributions of macroalgal communities; these include El Niño and La Niña phases. El Niño brings warm, nutrient stressed conditions that cause declines of the larger kelp species *Macrocystis pyrifera* and sometimes an increase of the smaller understorey populations, whilst La Niña brings cold nutrient rich conditions that promote growth of *Macrocystis pyrifera* and the reduction of understorey populations (Edyvane, 2003). The Southern Oscillation Index can also alter the East Coast Low; these are extreme low pressure systems that bring heavy rainfall to the south of Australia resulting in large amounts of runoff and associated nutrients or anthropogenic affects (Edyvane, 2003).

Giant kelp (Macrocystis spp.)

Giant kelp (*Macrocystis pyrifera)* grows up to 30 meters tall and forms extensive surface canopies. It defines and structurally dominates kelp forests thus helping to support high biodiversity and productivity in these habitats. Many organisms rely on such productivity with declines in fish fauna diversity and lobster and abalone populations and economical losses in the fishing industry noted where kelp forests are removed (Bodkin, 1988). The ecology, biology, and impact of human and natural disturbances on *Macrocystis* have been of great interest. In addition to creating a three dimensional habitat architecture and influencing understorey sub-systems, pieces of the plant detach and form drifts, these drifts provide shelter, substrata, and food thus strongly influencing the abundances and distributions of a myriad of other marine organisms (e.g., Edgar 1983a, b, Okey 1997, 2003). Eventually these drifts are consumed or break down adding to the oceans detritus or wash onto the coast where shorebirds and other animals utilize them for nesting and foraging (Mount, 2005).

Impacts of Climate Change

Many studies have reported the decline of the giant kelp *Macrocystis pyrifera*, as a response to changing oceanographic and atmospheric conditions in connection with climate change. Such losses have been as high as 64% in the waters of Tasmania over the last 50 years (Edyvane, 2003). One hypothesis for these changes is that steady increases over the previous 50 years in the frequency and magnitude of El Niño events are associated with the rise of mean sea surface temperatures in Australia's temperate waters above optimal ranges for giant kelp. *Macrocystis pyrifera* can not survive or reproduce in temperatures above 20ºC and as such their geographical northern limit is related to summer sea temperatures (Edyvane, 2003). El Niño events produce sustained warm waters, a reduction in the strength of Pacific Trade Winds that would normally bring cool nutrient rich waters, a reduction in rainfall that limits nutrient runoff and higher sea surface temperatures. California has sustained large losses of kelp forests in connection with El Niño extended events (Dayton & Tegner, 1984; Bodkin, 1988), however such accusations have not been proven and some debate still exists. A change in the East Australian Current also appears to be occurring and this may be induced by stratospheric ozone depletion causing a change in oceanic wind patterns and driving more warm water into the southern cool waters of Australia (Cai, 2006; Cai *et al.*, 2005). Both the El Niño event and the change in the East Australian Current are delivering warm waters to warm intolerant macroalgae species possibly causing large losses.

It is tempting to point to El Niño to explain patterns of kelp forest decline. Nonetheless, macroalgal declines might have resulted from a combination of environmental and biological changes. These include changes in temperature and nutrients, as discussed previously, and changes in predators and anthropogenic or natural disturbances. Environmental and biological effects are most likely interactive, and these are discussed in the following sections.

Temperatures and Seasonality

In addition to general increases in temperate Australia's sea surface temperatures, seasonal variability in sea surface temperatures (the range of temperatures) has generally decreased. Macroalgal species are adapted to particular seasonal variability's. For instance, *Macrocystis pyrifera* in California were found to reproduce most successfully where seasonality was most pronounced (Edyvane, 2003), with most macroalgal growth occurring during autumn when nitrate is more readily dissolved in the water. Indeed, many macroalgal species exhibit seasonally alternative growth patterns, with higher growth occurring when nutrients are most readily available (Dayton, 1985). Furthermore, the cooler autumn and winter period appears to be the most favourable for kelp reproduction, with gametophyte survival and movement at its highest due to the readily available dissolved nutrients (Dayton, 1985). The loss of seasonality has the potential to alter environmental cues for macroalgal reproduction or growth.

Massive kelp recruitment in devastated areas of California kelp forests after a 1982-83 storms suggests that gametophytes are able to survive in a semidormant state (Dayton & Tegner, 1984). This dormancy length differs among species, implying that changes in ocean conditions would lead to changes in species composition and community structure. A high degree of variability in resistance to changes within their environment exists among macroalgae species. During the strong El Niño event in 1983 on the coast of California, both the adults and new recruits of the giant kelp *Macrocystis* were seriously affected by high sea temperatures and low nutrient levels, whilst associated understory kelp species experienced very little impact (Dayton, 1985). The giant kelp *Macrocystis* appears particularly sensitive to rising temperatures and reduced nutrients. Below average rainfall and patterns of warming, particularly on the northeast coast of Tasmania, mirrors patterns of the greatest losses of giant kelp in Tasmania (Edyvane, 2003). If current temperature trends continue in Australia's shallow marine ecosystems, macroalgal assemblages might shift from canopy dominated assemblages to understory dominated assemblages. The substantial reduction in the distribution of such an important component of this environment over the last 50 years portend a grave outlook for regional lobster and abalone fishing industries, tourism, and the broader biological community that is supported by this biogenic habitat (Edyvane, 2003).

Nutrients

Preference for macroalgae to reproduce and grow during cooler winter months has lead to studies designed to determine whether it is temperature or nutrients associated with temperature that determine giant kelp (*Macrocystis pyrifera*) distributions. Survival of kelp plants during the 1983 El Niño event of the coast of California was greater when positioned in the path of nutrient pulses, whilst others outside of these nutrient flows did not survive (Zimmerman & Robertson, 1985). Similarly higher canopy foliage in warm low nutrient waters experienced die back whilst lower foliage of the same plant in the more nutrient rich waters did not (Dayton *et al.*, 1992). In the waters surrounding Southern California's Santa Catalina Island, the giant kelp recruits that replaced plants removed during the 1983 El Niño event were limited by modified weather and temperature patterns coupled with reduced nutrient availability. It appears that nutrients could be pivotal to kelp distribution along the California coast, but this is less clear for Australia.

Australian coastal marine ecosystems are low in essential nutrients because of limited upwelling, the old age of our soils and their depleted nitrate and phosphate content, and the tropical origins of the predominant coastal currents. It is well known that high temperatures are related to low nutrient levels (Sanderson, 1997), with a large proportion of Australia being surrounded by such warm saline waters. Nutrient runoff processes function differently in southern Australia than in other similar temperate environments, as this region receives relatively little rain and evaporation often exceeds freshwater runoff from the land thereby limiting sediment transport and encapsulated nutrients from river systems (Gray, 2002). The marine fauna and flora of the Southern Australian coastline obtain nutrients flows from limited upwelling and the impingement of cold nutrient-rich waters from the sub-Antarctic. But some evidence suggests that nutrient requirements are not the primary cause behind the loss of kelp within Australian waters. Indeed, nitrate levels in some areas of kelp declines might have been increasing in recent years, though nutrient peaks have been more sporadic than normal, as they are often related to rainfall runoff after large storm events or strong winds driving sub-Antarctic waters north (Edyvane, 2003).

Large storm events are thus alternative sources of nutrients, but the nitrogen storage capacity of giant kelp is very limited (Steneck *et al.*, 2002). El Niño events are also altering atmospheric patterns causing a reduction in the frequency of storm events and rainfall. This makes giant kelp vulnerable if the time between each runoff event is longer than the storage length. A much greater understanding of the processes that shape macroalgal assemblages is needed to determine whether temperature change is a main driver of observed declines in Australian kelp forests. More research is turning towards not only the large sporophyte stage but also the separate life stages of the gametophytes and juveniles that require different threshold conditions (Dayton, 1985). The effects of temperature and nutrients on Australian macroalgae are difficult to separate because warm seawater contains low nitrogen levels. A safe default assumption is that both are instrumental.

Debate exists on whether the storm disturbances that have recently been occurring are actually detrimental to macroalgae species or important levels of disturbance with there own ecological role (Dayton, 1985). Increased storm activity as a consequence of higher intensity El Niño events have caused elevated numbers of kelp mortality in the south and east coast of Tasmania with no doubt that entire communities have been severely impacted. Full understanding of the long term impacts and dynamics within the whole community is largely unknown. Just as terrestrial forests benefit from disturbance, the removal of kelp forests allows light to penetrate and understory or encrusting coralline algae to flourish (Reed & Foster, 1984). This disturbance could be a necessary process for the species and ecosystem long term survival. However, large distances between established individuals might be detrimental, as it is highly important to have an optimal maximum recruitment density of 3 m, as a high density of spores is needed to achieve successful fertilisation (Dayton, 1985).

Currents

Evidence suggests that there is increasing penetration of the East Australia Current and Leeuwin Current into the cool waters of temperate Australia, causing temperatures to increase and nutrient availability to decrease. Species of a more tropical affinity that have survived and evolved in the Great Australian Bight originally dispersed south through these currents (Phillips, 2001). These tropical species may become more prevalent as the warm currents infiltrate further altering conditions in favour of tropical species and to the determent of dominant temperate macroalgae and other associated flora and fauna. As it is recognised that flora of the tropical waters has a lower diversity of species then cool temperate waters (Sanderson, 1997), a predicted drop in species numbers could be assumed. Reports indicate the northern limits of sub-Antarctic water may have shifted south during the 1990's (Edyvane, 2003). The implications of this shift for giant kelp and dependent and associated organisms are largely unknown. A worrying aspect is the absence of any nearby landmass or shallow rocky reef south of Tasmania still surrounded by the receding sub-Antarctic waters that these threatened kelps could retreat to. This retreat of sub-Antarctic waters seems to be attributed to the East Australian Current, pushing further into the southern waters of Australia by the change in oceanic wind patterns. The East Australian Current brings warm water that is forcing the cooler waters of the Antarctic to increasingly retreat south (Cai, 2006; Cai *et al.*, 2005). No Australian landmass surrounded by the cold Antarctic waters will soon remain that could harbour warm intolerant macroalgae species.

Grazers

Changes in hydrological and temperature conditions can cause cascading effects such as grazing pressures by sea urchins and abalone striping large areas of macroalgae. In Australia the three principal species that feed on brown algae are Black Sea Urchin (*Centrostephanus rodgersii*), the Purple Sea Urchin (*Heliocidaris erythrogramma*) and the Blacklip Abalone (*Haliotis rubra*) (Edyvane, 2003). These species can feed on all life stages of macroalgae (Dayton, 1985). Cyclic fluxes of deforestation by sea urchins is not commonly seen in Australian waters, however following a natural or human disturbance coupled with temperature and hydrological changes kelp forests are more susceptible to grazing pressures causing large areas of barren substratum. The role that sea urchins play in the susceptibility of kelp to environmental changes was seen in California following the more recent El Niño event. Kelp forest regrowth and recovery was higher then that recorded after the 1957-59 El Niño event and this was attributed to the increased fishery of the kelp predator sea urchins since the previous atmospheric event (Zimmerman & Robertson, 1985). Suggestions have also been made that the numbers in sea urchin populations increase faster then the growth rate of kelp under conditions of increasing sea surface temperatures. If this were to occur sea urchin grazing pressure would be too great for the growth of kelp forests creating barren areas as sea temperatures increased (Leighton *et al.*, 1966).

Conclusion

There is no doubt that changes are occurring within species assemblages and distribution in connection to climate change. Macroalgae as a dominating species has a highly important role as a primary producer in providing food, shelter and nesting grounds for a diverse range of organisms. Very little research or correlation has been made between the loss of large areas of dominate macroalgae forests and the consequences on associated species such as important crustaceans or molluscs. Population numbers of lobsters or urchins have been noted as decreasing or increasing but only in relation to overfishing or the removal of said species predator or prey. The potential impacts are not just environmental but also economical; lobster and abalone fisheries are large industries in the southern rich waters and would be negatively affected with changes in kelp distribution. The tourism industry relies on the beauty of the macroalgae and the rich community of unique species that it supports. Loss of such large areas of kelp and the modification of the environment will have devastating affects on the tourist industry impacting small and large businesses alike.

There is recognition that climate change is having an impact on the macroalgae dominated coast line of temperate Australia. Changes in the marine environment in oceanographic and atmospheric conditions are taking their toll on an important and dynamic system. The El Niño event is bringing warmer waters and the dominant southward flowing currents on the east and west coast of Australia are progressively pushing south bringing not only warm waters but tropical related macroalgae that could potentially out-compete the cooler affinity species. However, the real extent of impact of climate change on these organisms is still as such unknown and until further studies can be performed this will remain an unanswered question.

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Scoping the construction of a Gulf of Carpentaria Ecopath model

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Abstract

Few exploratory studies have encompassed the entire Gulf of Carpentaria partly due to its isolation and large uninhabited areas. Those that have been conducted did not sample for all functional feeding groups that exist within this environment. Using the functional feeding groups included in the preliminary Albatross Bay Ecopath model (Okey, 2006) missing biomass values that would allow the construction of a Gulf of Carpentaria Ecopath model were identified. It was found that a large number of functional groups did lack adequate data to allow the calculation of their total biomass within the Gulf of Carpentaria region. Future sampling studies would be beneficial to gain the necessary data to allow the construction of the best possible Ecopath model.

Introduction

The purpose of the present exercise was to scope the possibility of a rapid construction of a trophodynamic Ecopath model of the entire Gulf of Carpentaria by expanding a preliminary Albatross Bay Ecopath model (Okey, 2006). Ecopath is a mass-continuity trophodynamic computer model designed to characterise the whole food web trophic interactions and integrate non-trophic mediation effects and environmental forces as an ecological synthesis of the broad biological and fisheries information known about an area (Okey, 2006). Albatross Bay is a relatively small area (approx. 5,788 km2) within the much larger Gulf of Carpentaria (approx. 400,000 km^2) (Figure 1).

A Gulf of Carpentaria model can be expanded from the Albatross Bay model using new biomass estimates that take into account the different abundances over the larger area. Where possible, information was gathered from published literature on exploratory surveys conducted incorporating the entire Gulf and all its habitats. The functional groups specified for the model (Table 1) will be the same as those used for the Albatross Bay model (Okey, 2006) as we could expect to encounter similar feeding guilds for the entire system. Production/biomass ratios, consumption/biomass ratios and diet compositions were assumed to remain the same for all functional groups as found in the Albatross Bay model for the purposes of the present exercise (Okey, 2006). Biomass will be the only value recalculated at this time, as this would be expected to change most when the entire Gulf of Carpentaria is considered for the Ecopath model.

Once a viable Gulf of Carpentaria Ecopath model is constructed, simulations could then be run projecting possible ecological effects of changes to specific functional group biomasses due to human or natural disturbances. This report is an initial attempt to identify and highlight functional groups in the model for which estimates for Gulf of Carpentaria biomasses are missing or uncertain to provide future efforts with a head-start in constructing a Gulf of Carpentaria model.

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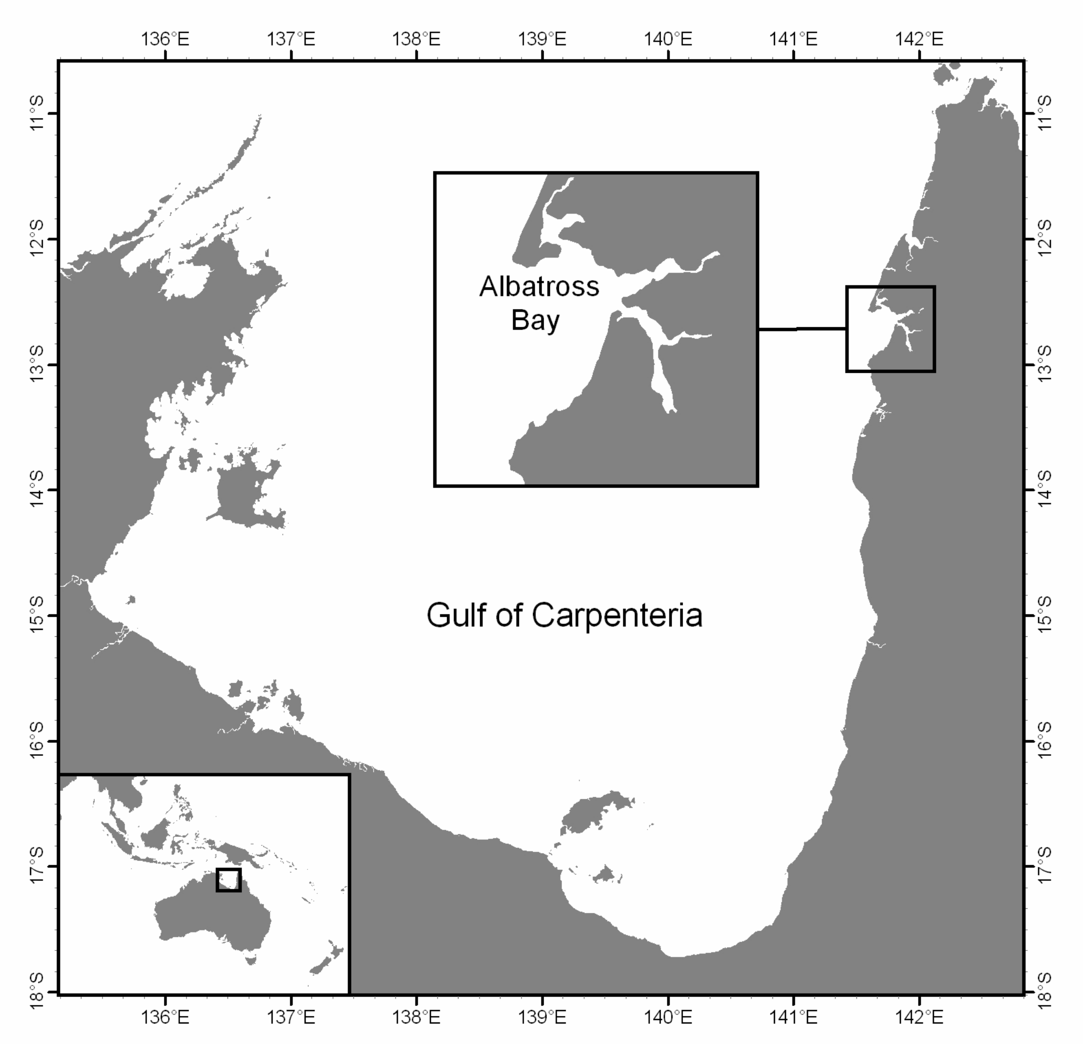

Figure 5: Map of the Gulf of Carpentaria marine ecosystem including adjacent estuaries (produced by Ian McLeod, CSIRO Marine & Atmospheric Research, Cleveland).

Methods

The Gulf of Carpentaria is a shallow body of water reaching a maximum depth of 70 m with numerous estuaries entering from the coastline of Queensland and the Northern Territory. The total area of the Gulf of Carpentaria was specified as 400 000 km² (Living Planet Analysis, 1993) for the purposes of the Gulf of Carpentaria model. Biomass values were calculated from annual collections reliant on the exploratory survey that was conducted, this altered between the different feeding groups. Sampling methods used differed dependant on which biota was been targeted. The functional groups specified (Table 1)--production/biomass ratios, consumption/biomass ratios and the ecotrophic efficiency values—were kept the same for all functional groups as in the preliminary Albatross Bay model (Okey, 2006).

Past surveys (Appendix A, Table 2) from which biomasses were obtained have noted an array of environment types that support their own functional groups and intra-and inter-specific interactions, these include estuarine zones, reef flats, muddy sediments, sandy sediments and pelagic zones. However there have been very few exploratory studies that encompassed the entire Gulf of Carpentaria partly due to its isolation and large uninhabited areas. Those that have been conducted did not sample all functional groups used in the Ecopath model. For this reason values for a number of the functional groups are missing or have been extrapolated from field surveys for the Albatross Bay (Okey, 2006). Values that have been entered are only estimates from available data found and therefore can be modified in the future to allow for incorporation of studies that were not included into this conceptual model.

Results and Discussion

The remainder of this report is a listing of the functional groups identified during this scoping exercise for which biomasses are uncertain for overall Gulf of Carpentaria. Suggested additional work has been identified and listed in each section. I did not attempt to derive fisheries information for the entire Gulf of Carpentaria. That should be accomplished in future efforts. The basic biological parameters used in this model of the Gulf of Carpentaria model listed in Table 3 in the Appendix.

Estuarine and Marine Zones

The preliminary Albatross Bay model was designed to focus primarily on the commercial prawn species within their broader ecosystems of the Albatross Bay region. Therefore, the preliminary Albatross Bay model explicitly integrated estuarine and marine 'sub-webs' in order to gain insights into the ecological (and fishery) importance of this interface (Okey, 2006). I assumed that the estuarine zone and affiliated species should be included in the overall Albatross Bay model due to the importance of this system as nursery grounds for many fish and invertebrate species and its role in detritus flows. Species were identified as inhabiting either the marine or estuarine bioregions and as such treated separately with separate functional groups. However the total area (km²) of Gulf of Carpentaria estuaries needs to be estimated to enable derivations of biomasses for estuarine functional groups such as estuarine fish, gastropods, bivalves, seagrass, mangroves, crocodiles, worms, meiofauna, forams, zooplankton and primary producers, particularly in relation to the Ecosim and Ecospace components of the modelling approach. Estimates of the total area of estuaries in the Gulf of Carpentaria might be available from the Queensland Department of Primary Industries (QDPI).

Marine Mammals and Reptiles

Dugongs, Turtles and Dolphins

Very few surveys have been conducted on marine mammals and reptiles across the entire Gulf of Carpentaria; these functional groups include dolphins (bottlenose, Indo-Pacific humpback and Irrawaddy River dolphins), dugongs and sea turtles (green and leatherback turtles). Values for the biomass estimates of these megafauna were estimated from figures published by Marsh *et al.,* (2004) from quantitative aerial surveys conducted during 1992. Uncertainty in these estimates reflects the remoteness of the area, the diving characteristic of these animals, difficulties with identification, and problems with extrapolation. Biomass values for dugongs, turtles, and dolphins used in the preliminary Albatross Bay model were gathered as rough placeholders via personal communication with C. Limpus (Okey, 2006). More refined estimates are possible using existing information, but care should be taken in making these estimates and in soliciting the advice of experts in this field.

Sea Snakes

More data is needed on sea snake numbers. Current abundance values for the Gulf of Carpentaria were sourced from Wassenberg *et al.* (1994) which gathered data over a number of years using two different methods: fishing and research trawlers. Both of these survey types relied on incidental captures and since they were not specifically targeting sea snakes the results may be biased. Large differences in abundances between years may be the result of changes in trawler catch effort.

Crocodiles

The calculated biomass value for crocodiles in the preliminary Albatross Bay model (Okey, 2006) was found to be conflicting with the value that the Ecopath model calculated when mortality due to inshore gillnet fishing was accounted for. An estimated biomass for Albatross Bay was calculated using the Ecopath routine with a specified ecotrophic efficiency of 0.80. This estimates that 80% of the proportion of crocodile production is consumed by predators. There is no indication of the possible population numbers for the entire Gulf of Carpentaria. Being an estuarine animal they are not commonly found in the open ocean and remain close to the coastline, although larger specimens have been sighted in deeper water. Because of their allusive nature and there choice of habitat it will be difficult to gather accurate population numbers.

Birds

Large nesting colonies comprised of species such as the brown booby, lesser frigate bird, crested tern and common tern have been estimated in their thousands. These colonies are situated in the south-eastern region of the Gulf of Carpentaria mainly around North Bountiful Island, Rocky Island and Manowar Island and it can be assumed that the birds travel up 70 km from their nests to forage (Blaber & Milton, 1994). A more recent survey needs to be carried out to determine the current size of nesting colonies in order to make accurate estimates of bird biomasses. The most recent surveys reported increases in total nest numbers within a colony when compared to previous annual counts, and this increase was linked to the development of trawl fishing in the region (Blaber & Milton, 1994).

Fish

Demersal and Estuarine Fish

Demersal fish fauna biomasses were found to be twice as high in the Albatross Bay area then in the entire Gulf of Carpentaria (Blaber *et al.*, 1994). This difference in abundance explains the lower biomass values calculated for the Gulf of Carpentaria Ecopath model compared to those entered into the preliminary Albatross Bay Ecopath model. The Gulf of Carpentaria biomasses were calculated by using the data for the most abundant fish on the night and day trawls of surveys conducted by Blaber *et al.*, (1990) and Blaber *et al.* (1994) and categorized into functional groups. Catch rates for each functional group were then used in an equation taken from Blaber & Harris (1994) to calculate the overall biomass. Estuarine fish biomass values were sourced from Blaber *et al.* (1989). A small amount of research and literature exists on sawfish in the Gulf of Carpentaria however it seems that there is not enough data to estimate biomasses for this group accurately.

Pelagic Fish

Very little data at all exists for the functional groups of small/medium/large pelagic fish as these fish fauna are not a major target in the northern fisheries. However, illegal, unregulated, and unreported fishing has been increasing and this might be impacting the whole biological community through the removal of pelagic predators, for example. More research would have to be conducted focusing on sampling that targets pelagic fish fauna. The current biomass values in the Gulf of Carpentaria model were taken from those estimated for the preliminary Albatross Bay model (Okey, 2006). Considerably more desktop and field research is needed to derive accurate estimates of pelagic fish biomasses.

Mega invertebrates

Cephalopods
As for the Albatross Bay model (Okey, 2006), biomass values for octopus, squid and cuttlefish obtained from Dunning *et al.* (1994) are believed to be inaccurate and underestimates. The burrowing behaviour of octopus is thought to have influenced the accuracy of the sample caught; such an underestimation of abundance is also thought to have occurred for squid and cuttlefish. Due to the free swimming nature of the cuttlefish and squid, sampling methods designed to estimate benthos would not indicate the true cephalopod population size accurately. The values added into the Ecopath model for cuttlefish and squid are therefore a placeholder and only an indication of the biomass that can be found in the Gulf.

Crustaceans
Long *et al.*, 1995 has some information on some decapod and megafauna species in the bioregion. However, more targeted studies of these functional groups are required in the future. A small amount of stomatopod abundance data does exist (Long *et al.*, 1995) yet was considered a less then true estimation. There are good data to suggest that stomatopods are prevalent in the region as they encompassed a large proportion of fish gut contents and numbers have reported to be extremely high in some commercial tows in the Gulf.

Macro invertebrates

Gastropods
I was unable to find any data detailing gastropod abundances and no values were available from the preliminary Albatross Bay model (Okey, 2006). The majority of the macro-invertebrate data was sourced from in-faunal benthic community structure and function research by Long & Poiner (1994), no abundances on estuarine or marine gastropods existed within this paper. It may be possible to use data from other field research that has been completed in alternative areas of northern Australia.

Prawns

A fair amount of research has been carried out on the numbers and distribution of commercial prawns such as the tiger, banana, endeavour and king prawns due to the large fishing industry that exists in the Gulf region. Data used for estimating biomasses for the entire Gulf region were identified in Somers (1994). Surveys were conducted over sequential years from 1983-1991 for the species of tiger, endeavour, banana and king prawns and the mean was taken from the catch per species over all the years. A percentage of 118% was added to the value to take in consideration natural mortality and a further 4% was added for other mortality, these percentages were those used in the preliminary Albatross Bay model (Okey, 2006). The Ecopath model was used to calculate the biomasses for each age group, this included juveniles, sub adults and adults, and this calculation was already set from the preliminary Albatross Bay model (Okey, 2006).

Zooplankton, Meiofauna & Microbial Heterotrophs

No data were identified pertaining to the biomasses of these functional groups. Values have been included in the preliminary Albatross Bay model from calculations for zooplankton; these remain as place holder values in the Gulf of Carpentaria model.

Primary Producers

Very little data are available on the biomass, productivity or identification of primary producers in the Gulf of Carpentaria. It appears that the only research that has been conducted on this important part of the food web is that of Rothlisberg *et al.* (1994) studying phytoplankton community structure and productivity. Phytoplankton was measured through chlorophyll-*a* quantities in the water at different depths. Using these figures it may be possible to calculate values for phytoplankton biomass. The biomass of mangroves within the entire Gulf of Carpentaria region may be calculated using the litter fall and tree height information from the preliminary Albatross Bay model (Okey, 2006), however an idea on the total area that the mangroves occupy throughout the entire Gulf of Carpentaria would need to be quantified. Mangrove estuarine areas are not present along the entire coastline and are found only where inlets exist, 136 mangrove inlets can be found spread around the coast line of the Gulf (Living Planet Analysis, 1993). It has been suggested that this information may be available through the QDPI.

In my research I was unable to locate abundances of seagrass throughout the entire Gulf, however remote sensing and site location studies have been performed by CSIRO, it may be possible to source and calculate values through these avenues (Living Planet Analysis, 1993). Seagrass species tend to grow in depths of between 0-5m however they are not distributed throughout all substrata at these depths. More knowledge must be obtained of their distribution patterns before an accurate biomass value can be obtained.

Measurements for the remaining primary producers in the system were unable to be found and it appears that surveys measuring for the other primary producers in the Gulf of Carpentaria system have not yet occurred.

Detritus

Values or information on detritus in the Gulf of Carpentaria could not be identified from any of the literature so far mentioned. Whether or not these values could be obtained from Ecopath models previously constructed on similar environments would have to be investigated.

Conclusion

The Gulf of Carpentaria is a largely unexplored body of water that is home to a vastly diverse and dynamic array of organisms. With its shallow waters and mangrove lined inlets the Gulf of Carpentaria is facing the threat of human disturbances particularly from the increasing fishing pressures of prawn trawlers and long lines. To try and foresee potential impacts and changes within the food web interactions on an increasingly vulnerable habitat, it is essential that we endeavour to gain as much knowledge and understanding as possible of the whole community and its processes. A Gulf of Carpentaria Ecopath model would allow us to move one step closer to achieving such a goal.

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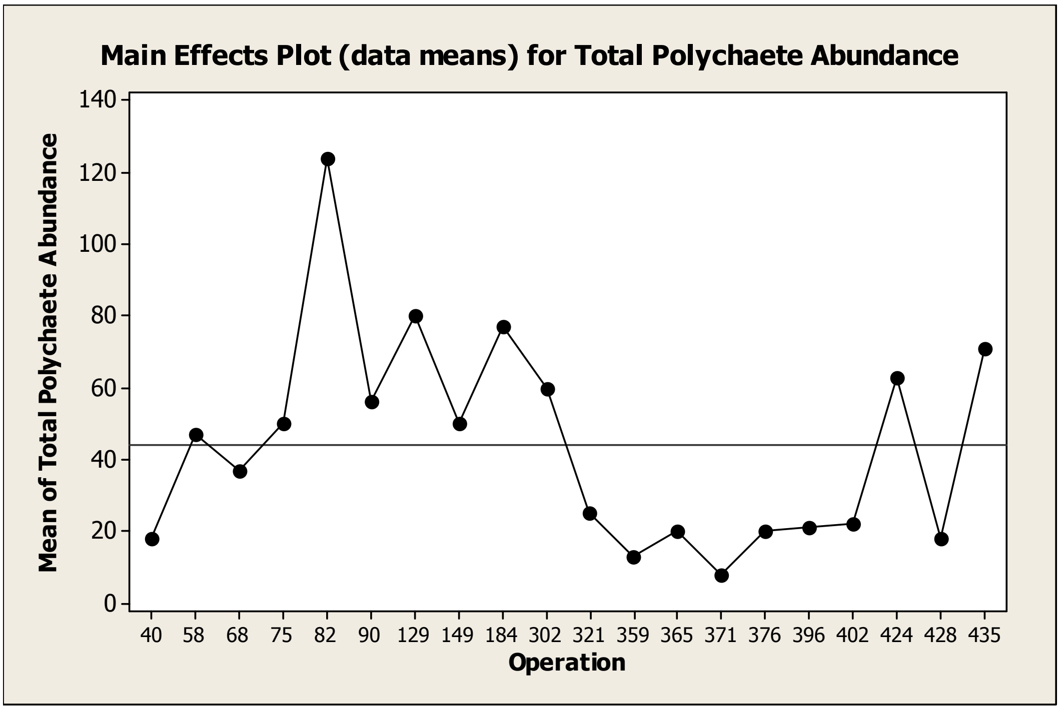
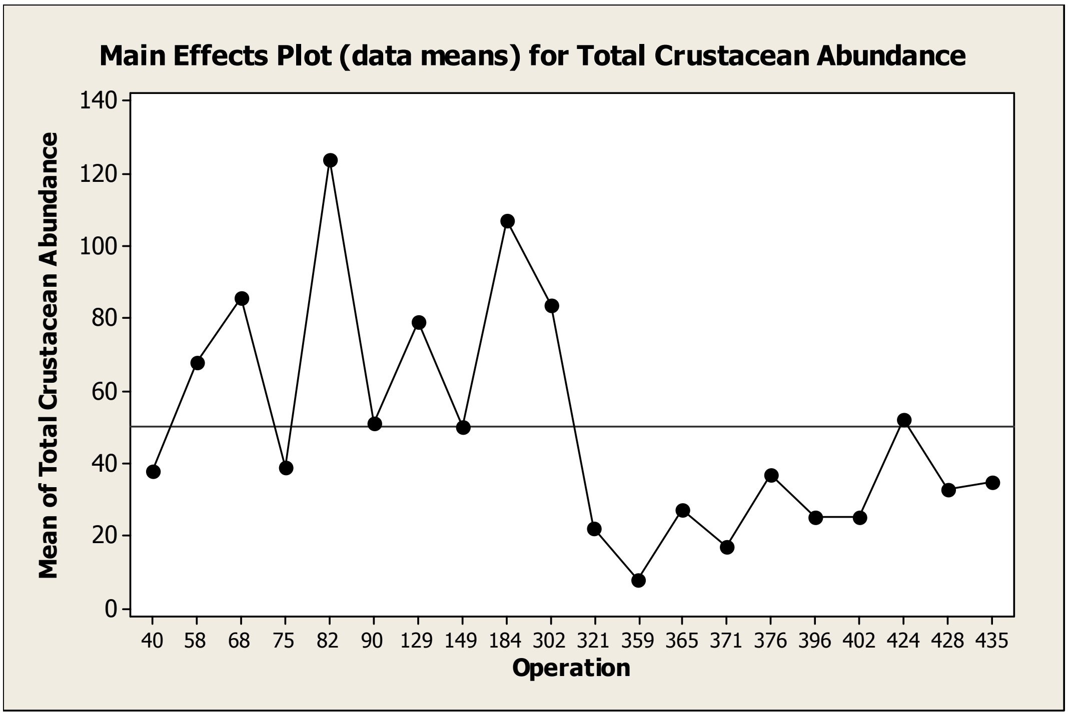
Appendix A. Plots of infaunal abundances from the Gulf of Carpentaria

The following plots A-H are illustrations of mean abundance against individual operations. The first 10 operation sites (40-302) are Mornington sites and the last 10 (321-435) are Vanderlins sites. These plots help to see the variation among operation sites.

A.



B.



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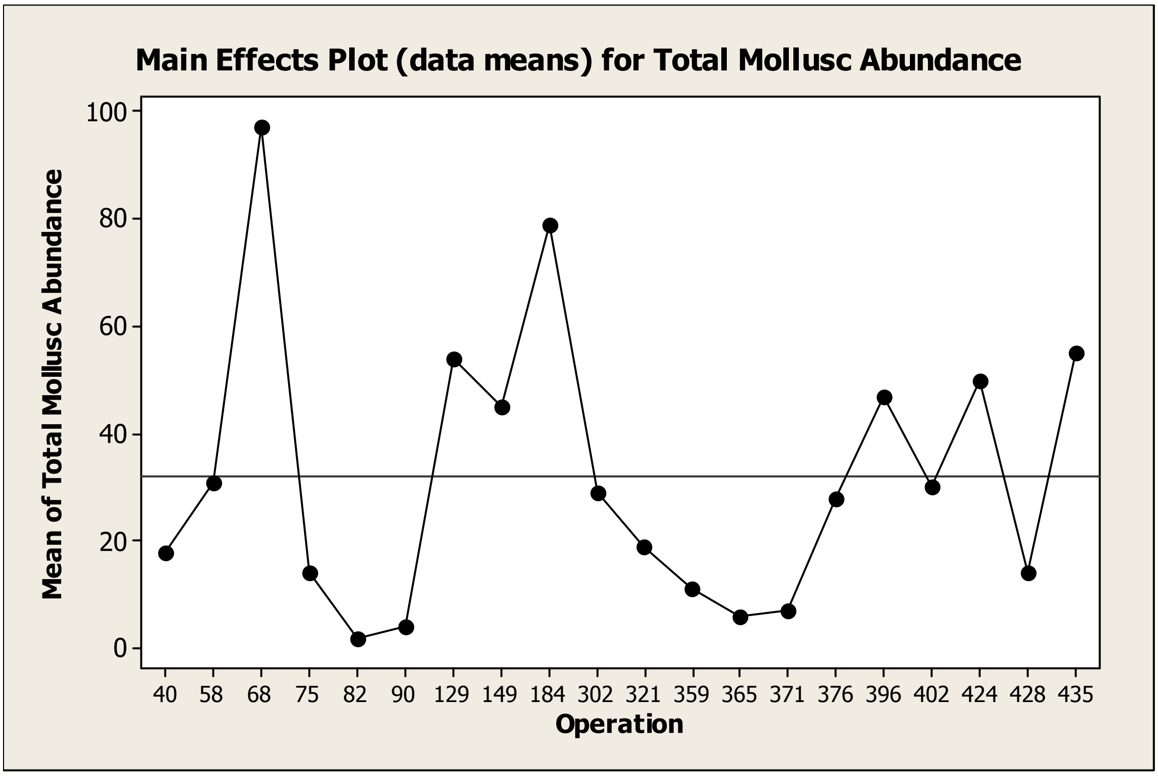
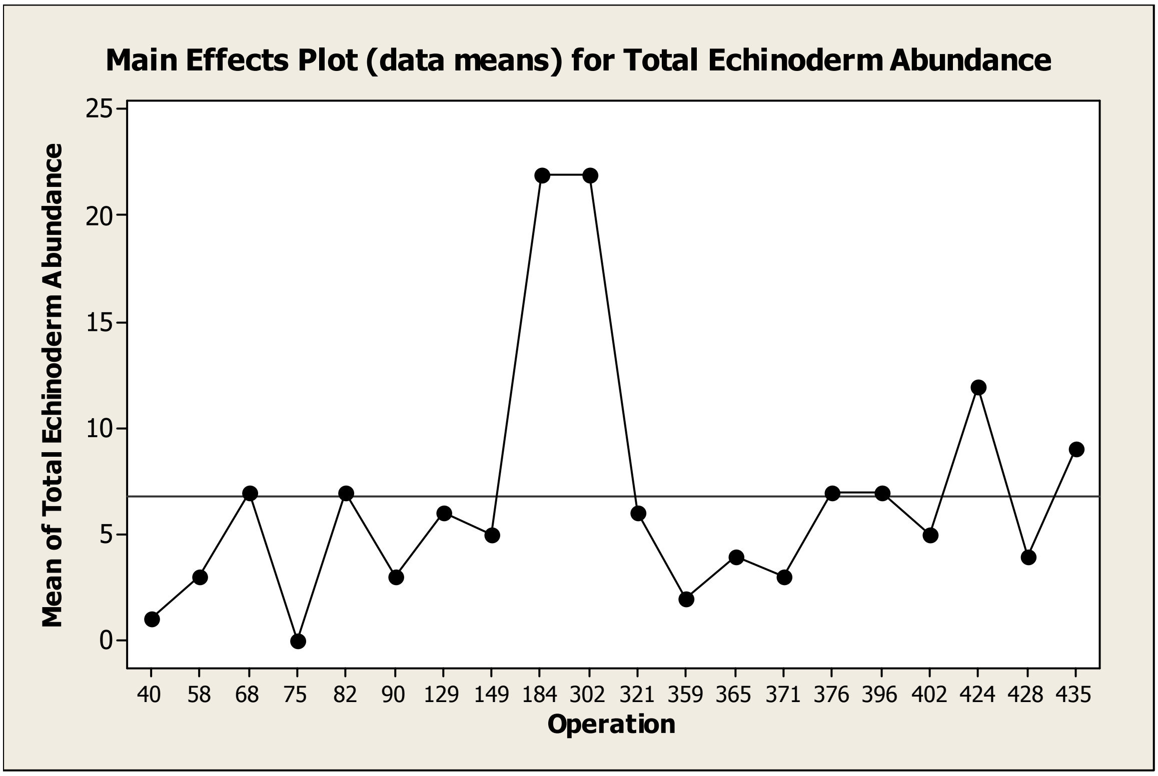
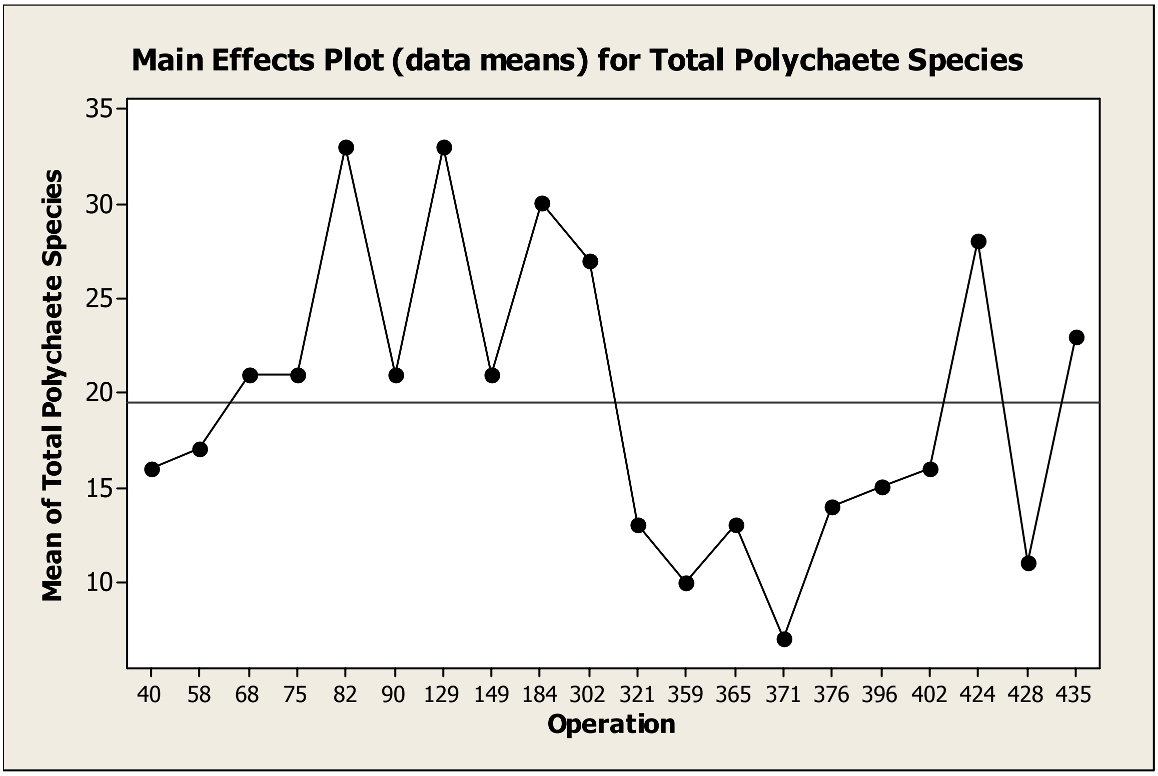

Main Effects Plot (data means) for Total Mollusc Abundance
Mean of Total Mollusc Abundance
100
80
60
40
20
0
40
58
68
75
82
90
129
149
184
302
321
359
365
371
376
396
402
424
428
435
Operation

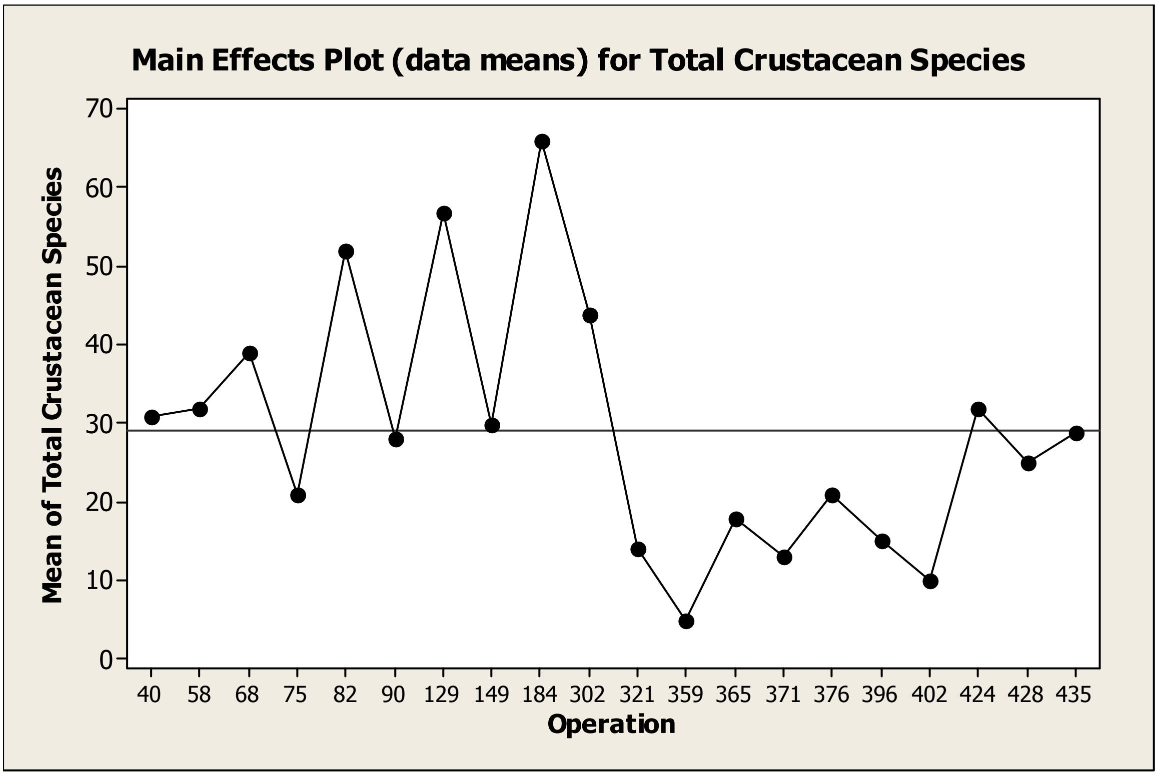
D.


Main Effects Plot (data means) for Total Echinoderm Abundance
Mean of Total Echinoderm Abundance
25
20
15
10
5
0
40
58
68
75
82
90
129
149
184
302
321
359
365
371
376
396
402
424
428
435
Operation

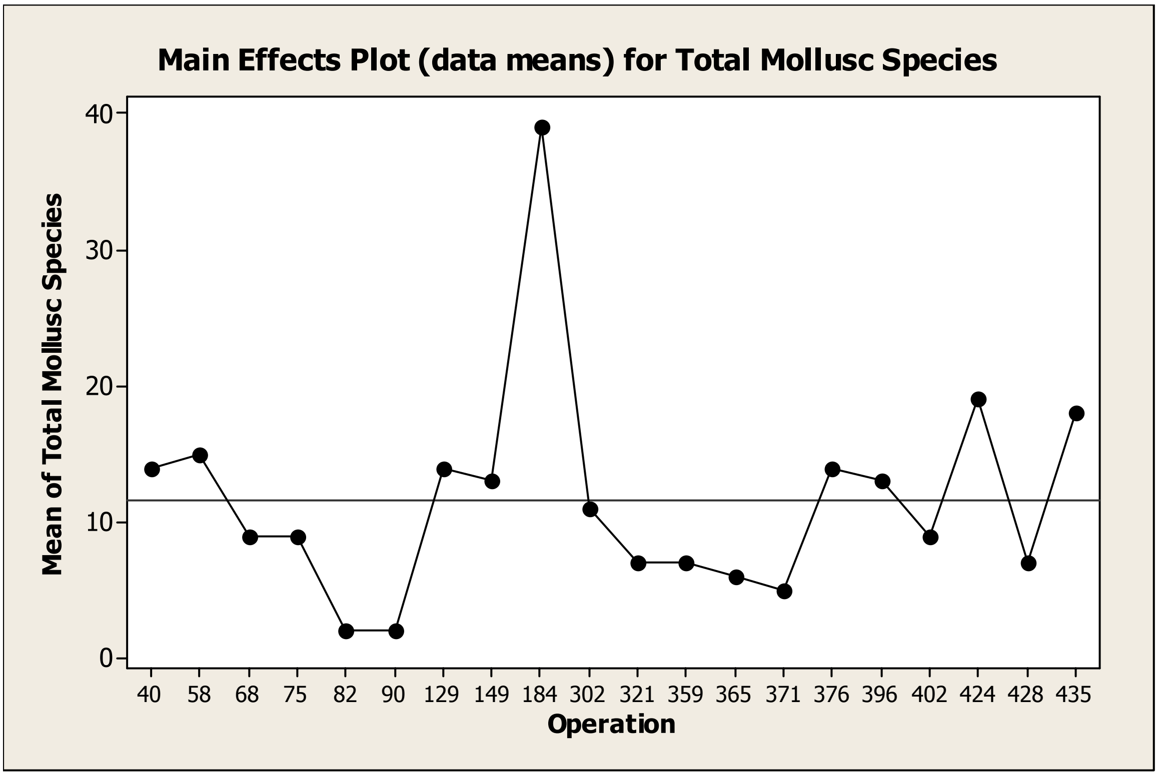
E.


Main Effects Plot (data means) for Total Polychaete Species
Mean of Total Polychaete Species
35
30
25
20
15
10
40 58 68 75 82 90 129 149 184 302 321 359 365 371 376 396 402 424 428 435
Operation

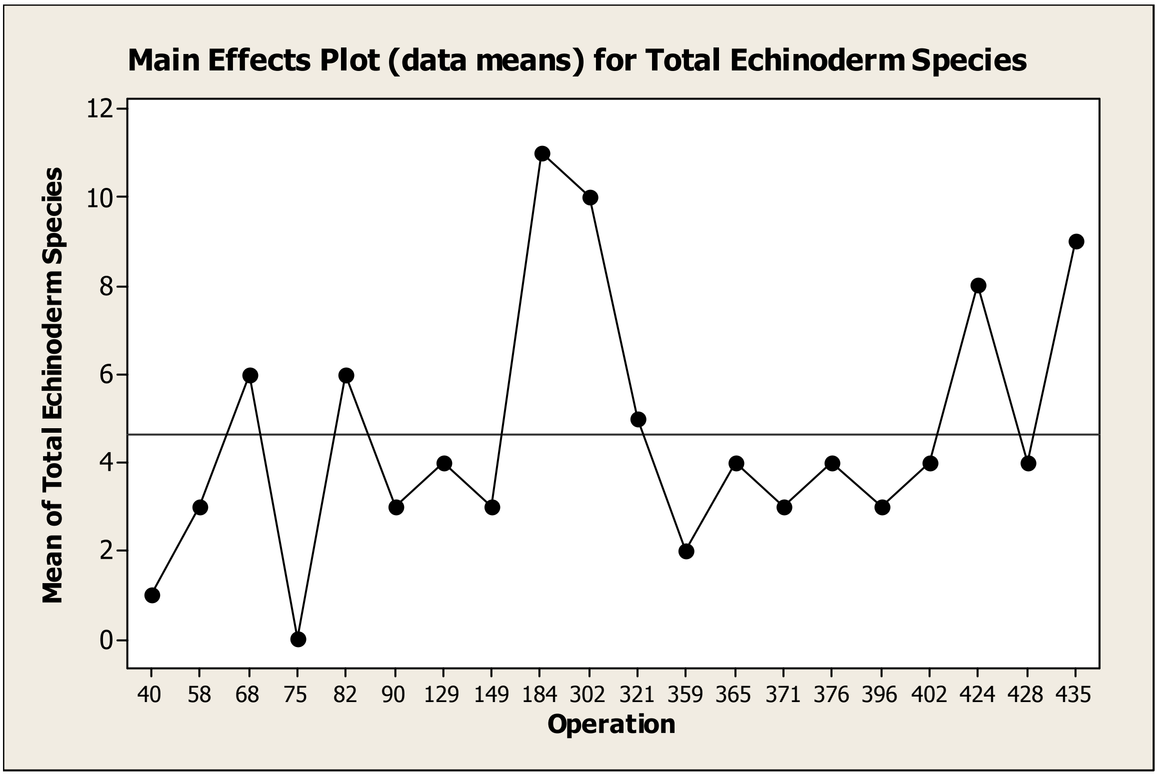
F.


Main Effects Plot (data means) for Total Crustacean Species
Mean of Total Crustacean Species
70
60
50
40
30
20
10
0
40 58 68 75 82 90 129 149 184 302 321 359 365 371 376 396 402 424 428 435
Operation

G.


Main Effects Plot (data means) for Total Mollusc Species
Mean of Total Mollusc Species
0
10
20
30
40
40
58
68
75
82
90
129
149
184
302
321
359
365
371
376
396
402
424
428
435
Operation

H.


Main Effects Plot (data means) for Total Echinoderm Species
Mean of Total Echinoderm Species
0
2
4
6
8
10
12
40
58
68
75
82
90
129
149
184
302
321
359
365
371
376
396
402
424
428
435
Operation

Appendix B. Functional groups, sources, and parameters for the preliminary model of the Gulf of Carpentaria

Table 2: Functional groups of the Ecopath model of Gulf of Carpentaria and adjacent slope and estuarine areas developed by the broad group of experts listed in Okey (2006).

Mammals	
1	Dolphins
2	Dugongs
Reptiles	
3	Crocodiles
4	Turtles
5	Sea snakes
Seabirds	
6	Lesser frigates
7	Brown boobies
8	Crested terns
9	Common terns
Offshore fishes	
10	Large pelagic piscivores
11	Medium pelagic piscivores
12	Small pelagic piscivores
13	Sawfishes
14	Large teleost benthic piscivores
15	Small benthic piscivores
16	Large elasmo benthopelagic piscivores
17	Large teleost benthopelagic piscivores
18	Small benthopelagic piscivores
19	Large benthopelagic invert feeders
20	Small benthopelagic invert feeders
21	Large elasmo benthic invert feeders
22	Large teleost benthic invert feeders
23	Small benthic invert feeders
24	Polychaete feeders
25	Large pelagic planktivores
26	Small pelagic planktivores
27	Benthic herbivore
28	Scavengers
Estuarine fishes	
29	Estuary large elasmo benthopelagic pisc/prawn feeders
30	Estuary large teleost benthopelagic pisc/prawn feeders
31	Estuary large benthic pisc/prawn feeders
32	Estuary large benthopelagic invert feeders
33	Estuary large benthic invert feeders (Rays)
34	Estuary Polychaete feeders
35	Estuary small benthic invert feeders
36	Estuary planktivores
37	Estuary detritivores
38	Estuary benthic herbivores
39	Estuary insectivores
40	Estuary pelagic herbivores
82	Estuarine ichthyoplankton
83	Insects

Mega-invertebrates	
41	Octopus
42	Squid and cuttlefishes
43	Stomatopods
44	Banana prawn juvenile
45	Banana prawn subadult
46	Banana prawn adult
47	Tiger prawn juvenile
48	Tiger prawn subadult
49	Tiger prawn adult
50	All other commercial prawns
51	Thallasinid prawns (Callianassa)
52	All other non-commercial prawns
53	Crayfish
54	The mud crab
55	Red mud crab
56	Sand crab
57	Other large crabs
58	Large gastropods
59	Holothurians
60	Spatangoids
61	Echinoids
62	Ophioroids
63	Asteroids
64	Sessile epibenthos
Macro-invertebrates	
65	Marine bivalves
66	Estuarine bivalves
67	Marine small gastropods
68	Estuarine small crustaceans
69	Marine worms
70	Estuarine worms
71	Marine small gastropods
72	Estuarine small gastropods
Meiofauna	
73	Marine meiofauna
74	Estuarine meiofauna
75	Marine forams
76	Estuarine forams
Plankton	
77	Large jellies
78	Small jellies
79	Marine zooplankton
80	Estuarine zooplankton
81	Marine ichthyoplankton
91	Marine macroalgae
92	Mangroves

Microbes

84 Marine microbial heterotrophs
85 Estuarine microbial heterotrophs

Primary production

86 Marine phytoplankton
87 Estuarine phytoplankton
88 Microphytobenthos
89 Seagrass
90 Estuarine macroalgae

Detritus

93 Discards
94 Detached marine macrophytes
95 Detached estuarine macrophytes
96 Estuarine water-column detritus
97 Estuarine sediment detritus
98 Marine water-column detritus
99 Marine sediment detritus

Table 3: A list of published articles from which biomass values were sourced

1 Blaber S.J.M., Brewer D.T. & Salini J.P., 1989. Species composition and biomasses of fishes in different habitats of a tropical northern Australian estuary: their occurrence in the adjoining sea and estuarine dependence. *Estuarine, Coastal and Shelf Science*. Vol 29. pp 509-531.

2 Blaber S.J.M., Brewer D.T. & Harris A.N., 1994. Distribution, biomass and community structure of demersal fishes of the Gulf of Carpentaria. *Australian Journal of Marine and Freshwater Research.* Vol 45 pp 375-396.

3 Blaber S.J.M., Brewer D.T., Salini J.P. & Kerr J., 1990. Biomasses, catch rates and abundances of demersal fishes, particularly predators of prawns, in a tropical bay in the Gulf of Carpentaria, Australia. *Marine Biology*. Vol 107 pp 397-408.

4 Dunning M., McKinnon S., Lu C.C. & Yeatman J., 1994. Demersal cephalopods of the Gulf of Carpentaria, Australia. *Australian Journal of Marine and Freshwater Research*. Vol 45 pp 351-374.

5 Long B.G. & Poiner I.R., 1994. Infaunal benthic community structure and function in the Gulf of Carpentaria, Northern Australia. *Australian Journal of Marine and Freshwater Research*. Vol 45 pp 293-316.

6 Long B.G., Poiner I.R. & Wassenberg T.J., 1995. Distribution, biomass and community structure of megabenthos of the Gulf of Carpentaria, Australia. *Marine Ecology Progress Series*. Vol 129. pp 127-139.

7 Marsh H., Corkeron P., Preen T. & Pantus F., 2004. Marine species protection program completed projects: Aerial surveys of marine wildlife in the Gulf of Carpentaria. School of Tropical Environmental Studies and geography, James Cook University, sited on 5/01/06 on http://www.nht.gov.au/nht1/programs/mspp/gulf.html.

8 Okey T.A., Editor. 2006. A preliminary Ecopath model of Albatross Bay, Gulf of Carpentaria, Australia, for the period 1986-1992, CSIRO Marine Research, Cleveland, Australia.

9 Wassenberg, T.J., Salini J.P., Heatwole H. & Kerr J.D., 1994. Incidental capture of sea-snakes (*Hydrophiidae*) by prawn trawlers in the Gulf of Carpentaria, Australia. *Australian Journal of Marine and Freshwater Research*. Vol 45 pp 429-443.

Table 4: Basic biological parameters used in the preliminary Gulf of Carpentaria model. Values in bold are calculated (outputs) by Ecopath

Group name	Trophic level	Biomass (t/km^2)	Prod./ biom. (/year)	Cons./ biom. (/year)	Ecotrophic efficiency	Production / consumption
Dolphins	**4.76**	0.00303	0.100	41.070	**0.001**	**0.002**
Dugongs	**2**	0.05	0.080	36.500	**0.747**	**0.002**
Crocodiles	**4.08**	**1.14E-06**	0.318	2.080	0.800	**0.153**
Turtles	**3.24**	0.035	0.192	3.500	**0.812**	**0.055**
Sea snakes	**4.98**	0.0027	0.700	6.100	**0.952**	**0.115**
Lesser frigates	**4.96**	0.00005	0.080	36.700	**0.718**	**0.002**
Brown boobies	**4.81**	0.00202	0.080	33.800	**0.000**	**0.002**
Crested terns	**4.73**	0.00034	0.204	47.500	**0.000**	**0.004**
Common terns	**4.23**	0.00036	0.160	65.100	**0.000**	**0.002**
Large pelagic piscivores	**4.45**	0.0075	0.500	7.767	**0.001**	**0.064**
Medium pelagic piscivores	**4.25**	0.12	0.577	12.307	**0.518**	**0.047**
Small pelagic piscivores	**4.61**	**0.248**	0.831	14.400	0.950	**0.058**
Sawfishes	**4.72**	0.04	0.123	2.575	**29.431**	**0.048**
Large teleost benthic piscivores	**4.11**	**0.0557**	0.566	6.460	0.850	**0.088**
Small benthic piscivores	**4.45**	**0.183**	1.042	5.168	0.950	**0.202**
Large elasmo benthopelagic piscivores	**4.53**	0.0564	0.500	7.856	**5.607**	**0.064**
Large teleost benthopelagic piscivores	**4.43**	**0.236**	0.451	3.421	0.850	**0.132**
Small benthopelagic piscivores	**4.18**	**0.103**	0.868	8.172	0.950	**0.106**
Large benthopelagic invert feeders	**4.47**	0.0019	0.547	7.792	**0.589**	**0.070**
Small benthopelagic invert feeders	**3.91**	**2.359**	2.000	4.800	0.950	**0.417**
Large elasmo benthic invert feeders	**4.12**	0.0752	0.320	9.932	**2.206**	**0.032**
Large teleost benth invert feeders	**4.21**	0.0742	0.577	4.714	**0.931**	**0.122**
Small benthic invert feeders	**3.67**	0.493	1.500	5.026	**0.965**	**0.298**
Polychaete feeders	**3.44**	**0.589**	1.450	7.554	0.950	**0.192**
Large pelagic planktivores	**4.26**	**0.0189**	2.188	16.150	0.960	**0.135**
Small pelagic planktivores	**3.2**	**5.057**	2.189	16.830	0.980	**0.130**
Benthic herbivores	**2**	0.0236	1.510	35.167	**0.703**	**0.043**
Scavengers	**3.73**	0.0009	0.450	6.100	**0.864**	**0.074**
Estuary lg elasmo benthopelagic pisc/prawn feeders	**3.94**	0.317	0.354	4.456	**0.113**	**0.080**
Estuary lg teleost benthopelagic posc/prawn feeder	**3.85**	0.317	0.439	8.392	**1.174**	**0.052**
Estuary large benthic pisc/prawn feeders	**3.7**	0.496	0.370	4.067	**0.513**	**0.091**
Estuary large benthopelagic invert feeders	**3.64**	0.0735	0.506	5.375	**0.080**	**0.094**
Estuary large benthic invert feeders (Rays)	**3.8**	2.444	0.273	6.871	**0.000**	**0.040**
Estuary polychaete feeders	**3.29**	0.286	1.043	9.433	**0.109**	**0.111**
Estuary small benthic invert feeders	**3.59**	**0.283**	1.280	11.100	0.980	**0.115**
Estuary planktivores	**4.2**	**1.818**	2.326	16.420	0.980	**0.142**
Estuary detritivores	**2**	**2.009**	1.175	19.300	0.800	**0.061**
Estuary benthic herbivores	**2**	**0.0068**	1.880	45.750	0.980	**0.041**
Estuary insectivores	**3.42**	**0.0432**	0.690	9.500	0.980	**0.073**
Estuary pelagic herbivores	**2.1**	**0.3**	1.083	36.833	0.900	**0.029**
Octopus	**3.82**	**0.0985**	2.370	7.900	0.900	**0.300**
Squid and cuttlefishes	**3.62**	**0.847**	2.370	7.900	0.950	**0.300**

Group name	**Trophic level**	**Biomass (t/km²)**	**Prod./ biom. (/year)**	**Cons./ biom. (/year)**	**Ecotrophic efficiency**	**Production / consumption**
Stomatopods	**3.37**	**0.365**	3.500	7.432	0.950	**0.471**
Banana prawn juvenile	**3.03**	0.0113	3.720	43.888	**0.000**	**0.085**
Banana prawn subadults	**3.44**	0.0199	3.120	27.181	**1.227**	**0.115**
Banana prawn adult	**3.34**	0.079	3.200	19.200	**0.927**	**0.167**
Tiger prawn juvenile	**3.43**	0.0115	3.400	45.234	**0.069**	**0.075**
Tiger prawn subadults	**3.32**	0.0211	3.200	28.160	**3.219**	**0.114**
Tiger prawn adult	**3.32**	0.121	2.340	19.200	**0.177**	**0.122**
All other commercial prawns	**3.3**	**0.468**	3.000	25.000	0.900	**0.120**
Thallasinid prawns (Callianassa)	**3.17**	**1.004**	3.000	25.000	0.950	**0.120**
All other non-commercial prawns	**3.17**	**23.179**	3.000	25.000	0.950	**0.120**
Crayfish	**2.87**	**0.00987**	3.000	25.000	0.950	**0.120**
The mud crab	**3.07**	**0.0666**	2.800	8.500	0.900	**0.329**
Red mud crab	**3.07**	**0.0558**	2.800	8.500	0.900	**0.329**
Sand crab	**3.12**	**0.0689**	2.800	8.500	0.900	**0.329**
Other large crabs	**2.98**	**5.526**	2.800	8.500	0.900	**0.329**
Large gastropods	**3.28**	0.0228	2.800	14.000	**0.459**	**0.200**
Holothurians	**2.16**	0.065	0.610	3.360	**0.925**	**0.182**
Spatangoids	**2.93**	2.142	1.400	2.810	**0.167**	**0.498**
Echinoids	**2.01**	0.0849	1.650	2.810	**0.790**	**0.587**
Ophioroids	**2.13**	**24.802**	1.400	2.810	0.950	**0.498**
Asteriods	**2.77**	0.0508	0.490	3.240	**0.148**	**0.151**
Sessile epibenthos	**2.4**	4.985	0.800	9.000	**0.715**	**0.089**
Marine bivalves	**2.11**	**94.326**	1.209	23.000	0.900	**0.053**
Estuarine bivalves	**2.11**	**9.106**	1.209	23.000	0.900	**0.053**
Marine small crustaceans	**2.35**	**20.482**	7.010	27.140	0.980	**0.258**
Estuarine small crustaceans	**2.4**	**0.307**	7.010	27.140	0.980	**0.258**
Marine worms	**2.31**	**21.375**	6.850	27.400	0.980	**0.250**
Estuarine worms	**2.31**	**0.931**	4.600	15.900	0.980	**0.289**
Marine small gastropods	**2.55**	**64.033**	2.500	14.000	0.980	**0.179**
Estuarine small gastropods	**2.55**	**0.209**	2.500	14.000	0.980	**0.179**
Marine meiofauna	**2.36**	**18.819**	12.500	25.000	0.950	**0.500**
Estuarine meiofauna	**2.36**	**0.488**	12.500	25.000	0.950	**0.500**
Marine forams	**3.09**	**9.148**	12.500	25.000	0.950	**0.500**
Estuarine forams	**3.09**	**0.0286**	12.500	25.000	0.950	**0.500**
Large jellies	**2.73**	**0.0151**	40.000	80.000	0.500	**0.500**
Small jellies	**2.44**	**0.0268**	40.000	80.000	0.500	**0.500**
Marine zooplankton	**2.12**	5.5	53.000	265.000	**0.757**	**0.200**
Estuarine zooplankton	**2.12**	0.2	52.000	260.000	**0.391**	**0.200**
Marine ichthyoplankton	**2.62**	**0.0021**	50.448	132.130	0.990	**0.382**
Estuarine ichthyoplankton	**2.41**	**4.94E-05**	50.448	132.130	0.990	**0.382**
Insects	**2**	**0.018**	12.600	51.930	0.980	**0.243**
Marine microbial heterotrophs	**2**	**10.32**	100.000	215.000	0.950	**0.465**
Estuarine microbial heterotrophs	**2**	**0.442**	100.000	215.000	0.950	**0.465**
Marine phytoplankton	**1**	3.905	933.083	-	**0.455**	-
Estuarine phytoplankton	**1**	0.389	933.083	-	**0.305**	-
Microphytobenthos	**1**	**1.305**	706.496	-	0.950	-
Seagrass	**1**	**2.25**	2.145	-	0.600	-
Estuarine macroalgae	**1**	**0.818**	12.000	-	0.500	-
Marine macroalgae	**1**	**43.674**	12.000	-	0.500	-
Mangroves	**1**	**0.325**	3.300	-	0.400	-
Discards	**1**	0.7	-	-	**3.947**	-

Group name	**Trophic level**	**Biomass (t/km²)**	**Prod./ biom. (/year)**	**Cons./ biom. (/year)**	**Ecotrophic efficiency**	**Production / consumption**
DetachedMarine macrophytes	**1**	5	-	-	**0.599**	-
DetachedEstuarine macrophytes (estuarine)	**1**	5	-	-	**0.066**	-
EstuarineWater-column detritus	**1**	0.322	-	-	**0.208**	-
EstuarineSediment detritus	**1**	250	-	-	**0.163**	-
MarineWater-column detritus	**1**	33.351	-	-	**0.481**	-
MarineSediment detritus	**1**	250	-	-	**0.435**	-