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Effects of trawling on the benthos and biodiversity: development and delivery of a spatially-explicit management framework for the Northern Prawn Fishery

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Australian Government
Fisheries Research and Development Corporation

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OBJECTIVES:

1.1. DETERMINE THE ACCUMULATED EFFECTS OF TRAWLING ON BENTHIC COMMUNITY STATE AND COMPOSITION.

1.2. QUANTIFY KEY BENTHIC ECOSYSTEM PROCESSES OF IMPORTANCE TO PRAWN PRODUCTION AND BIODIVERSITY ALONG A TRAWL INTENSITY GRADIENT.

1.3. DEVELOP, AND PROVIDE FOR ADOPTION MANAGEMENT STRATEGY EVALUATION TOOLS FOR BENTHIC ECOSYSTEM IMPACTS.

1.4. DESIGN AND DELIVERY OF A SPATIALLY EXPLICIT MANAGEMENT FRAMEWORK FOR THE NPF.

OUTCOMES

1. *AFMA, the industry, advisory groups and science agencies will be in a better position to complete the collaborative development of a spatially explicit management framework.*

The spatial MSE delivered here provides, for the first time, an operational-analytical framework that will allow managers and the industry in particular, to evaluate alternative spatial management options and their consequences when achieving simultaneously the fisheries economic and stock objectives. This outcome will also give to managers and industry the ability to evaluate other conservation and environmental management objectives (e.g. closures or MPAs). This project also provided to managers, industry, and public a clear understanding of the impacts of trawling on the benthic ecosystems of the Gulf of Carpentaria (GoC) and demonstrated how that information is or should be used in spatial management of such impacts. The project also managed to build effective technical communication between various other existing science-based supports tools to the management of the NPF (economics, risk and stock assessments, bycatch management). This resulted in an integration of knowledge and a cost-efficient use of pre-existing tools towards the provision of one single spatial MSE framework for the NPF. The industry and management were kept directly involved and informed on the project's progresses and, in conjunction with AFMA, it is foreseen that managers will communicate and disseminate results to the NPF's RAG and MAC.

2. *The NPF will have a capacity for developing science-based objectives, targets, reference points and performance indicators for management of the impacts of trawling on benthic ecosystems.*

This project has provided a wide range of quantitative and numerical information of great value to management. This will allow the NPF to develop and use a range of indicators, performance measures, and set model-based targets for the spatial monitoring and evaluation of the impacts of trawling. In particular, the evaluation of the extent, nature, and variability of the impacts of trawling on the benthic ecosystem and its processes confirm previous findings that indicate that trawling at the current NPF's fishing effort and GoC spatial scales do not in themselves affect overall biodiversity. We believe that this is because the NPF operates predominantly over soft sediment benthic habitats were the site-specific variations depend on nature and scale of disturbance, and the relative importance of smaller-scale biological factors to broader-scale physical factors, i.e. biological responses, waves, currents, sediment transport, tidal bores, cyclones or floods). There are still however some specific important impacts at the small spatial scales (local) and at particular taxon/species levels that should be addressed in part by the current bycatch action plan and the ecological risk management responses for those taxa at risk. In short, this project has provided to the industry and managers with the tools and ability to respond to increasing environmental and conservation management policies, as well with an operative integration of all existing tools into a formal spatially-explicit ecosystem-based management framework.

3. *Managers of the NPF will be better able to delineate effective management stocks for use in spatial management. The knowledge and data from this project will include maps of prawn habitats and maps of environmental regions in the south-western Gulf of Carpentaria.*

The project made use of the existing prawn stock region boundaries developed and already in use by the bioeconomic stock assessment (Dichmont et al. 2006, 2008, 2010), so there was no need to develop new ones as expected originally in the project's proposal (2004-05). However, this project, in conjunction with a parallel and complementary spatial data project done for the northern region of Australia (Rochester et al. 2007), composed an extensive digital spatial library available at CMAR's http://www.marine.csiro.au/datacentre/ext_docs/mbp_north/characterisation/index.html whose output maps are included in Appendix 12. Therefore, this project did collate and integrate all spatially-explicit data available for the NPF providing managers with a detailed portrait and characterisation of the fisheries ecosystem under management, its state, composition and spatial variability. We believe that this information, in addition to the spatial MSE and the collection of analytical tools, provides this fishery and the region with unique and particularly strong science-based foundations needed for the management of its natural resources, its conservation and overall environmental sustainability.

4. *Risk assessment for benthic habitats, invertebrates and fish in the NPF, needed for sustainability assessment, will be enhanced. Quantitative estimates of the responses of benthic habitats and species will be provided by analysis of the Gulf survey samples.*

This project brought in the data, models and predictions of the current NPF's ecological risk assessment (ERA) as part of the core components to be integrated into the spatial MSE. The ERA models were used and enhanced, providing the managers with: (i) an independent spatial prediction for species at risk (species distribution model, see Appendix 8), (ii) the assessment itself, which is now part of integrated framework that bridges across various management tools, and (iii) the ability to use the ERA to create and define various spatial management scenarios or risk based closures –i.e. the scenarios of Low density High fishing effort (LdHf), the High density Low fishing effort (HdLf), and the Low density Low fishing effort (LdLf). The use of these ERA-based spatial closures has taken the current approach of ERA and how to inform the resulting AFMA ecological risk management plans (ERM), one step forward. Thus, the project has not only enhanced existing risk-based tools applied to biota, but has also integrated them into a holistic framework that provides this industry and the managers with a state-of-the-art operational spatial assessing tool for biota at risk.

The various estimates of the biological responses to the trawling impacts as derived from the performance measures, indicators and metrics (e.g. see section 10.3.9) are all quantitatively based, new and established in this project. This information is now readily available and will be provided to the industry and managers for use in the upcoming overall environmental assessment process.

5. *Sampling of benthic habitats and communities will be more cost-effective in the future, because of estimates of the spatial variation in benthic habitats and communities provided by this project.*

This outcome was identified hoping that, concurrently to the project and in the future, there be more biological and ecosystem field-based research in northern Australia that will impact and likely benefit the NPF environmental management. Towards that, we contributed substantially with the integration of most existing spatially-explicit biophysical datasets for the region and in particular the GoC. These data were used directly to plan and execute the SS 2005/03 and SS 2005/04 surveys (see Appendix 11). In addition, and based on the work developed by Rochester et al. (2007) and our own datasets, this project created a range of maps for habitats and its communities (see Appendices 6 and 9) and the ecosystem biophysical characterisation as well as detailed maps of the spatial distribution of the main functional groups of the fisheries ecosystem of the GoC (see Appendix 12). This range of spatial outcomes will, again, put managers, the industry and researchers in particular, in the best informed position for the planning and design of sampling, surveyor and, more importantly, monitoring and evaluation programs.

An additional outcome is that these spatial maps and data are and will be the core of the information basis for the bioregionalisation of the northern region (DEWHA 2008) and the design and placement of conservation MPAs within the areas of further assessment within the region (DEWHA 2009). Thus, delivery of this project regarding the required spatial data and evaluation tools is contributing substantially towards anticipating these conservation-based processes, if fisheries mangers and industry decide to do so.

This project developed a multidisciplinary approach to quantitatively evaluate the ecological effects of trawling on the ecosystem, and delivered analytical tools to evaluate such effects in spatially-explicit contexts under multiple management objectives. The focus of this project was the bentho-pelagic ecosystem of the tiger-endeavour fishing grounds of the Northern Prawn Fishery (NPF), the most valuable prawn species of this fishery. That fishery ecosystem includes mostly the shallow sedimentary shelves and submerged river beds of the south-western Gulf of Carpentaria (GoC). The ultimate goal was to deliver, to managers and the industry in particular, a science-supported spatial management framework and the derived evidence-based advice with regard to the spatial management of trawling impacts. This will allow the evaluation of alternative spatial management options addressing fishing impacts, while considering their trade-offs when achieving simultaneously fisheries economic and stock objectives, as well as other conservation and environmental management goals.

The main outcomes will be that (i) managers, industry and stakeholders will be ahead of the upcoming spatial fisheries and conservation management policies expected for northern Australia's fisheries and bioregions and (ii) an operational spatially-explicit ecosystem-based fisheries management approach (EBFM) will be available to the NPF.

The first two objectives of this project were aimed at determining and evaluating the likely effects of trawling on the ecosystem's biodiversity and some known ecological processes. To achieve these objectives the project made use of historical datasets and existing tools to conduct a targeted field-based survey focussing on the fisheries ecosystem and collect data following a natural experiment design. Thus, using the known spatial and historical extent of trawling, and the known environmental data, a research cruise of the National Facility R/V *Southern Surveyor* was conducted in 2005. The aim of this survey was to gather information and samples along a trawling intensity gradient and across all regions of the south-western GoC affected by trawling, while controlling for environmental variability. Most of the derived ecological information from all existing field surveys was used to in the construction of the food web, species distribution and risk assessment models.

The major finding for the likely impacts on biodiversity was that the results were consistent with partial effects of both trawling and habitat affecting positively some functional groups, taxa and species and negatively others. However, empirical relationships between catch composition and trawling intensity did not in themselves indicate that trawling affects overall biodiversity The association of fragile animals such as bryozoans and brittle stars with low trawling is consistent with a trawling effect. The relationship of reef-associated fish with low trawling and prawn predators with high trawling is consistent with a habitat effect. In general, the regional and day/night variation explained the variability in abundance and diversity indices better than the trawling intensities did. We could conclude that at the current 2005-10 levels of trawling and fishing effort, the management concerns should focus on spatial management options for those taxa that have been assessed to be negatively affected or being at risk and taxa that are specially protected, such as endangered, threatened and protected species (TE&P). We believe that lack of clear and overall trawling impacts could be due to: (i) the highly dynamic nature of soft-sediment habitats, that regularly encounter high natural variation (storm surges, tide, flooding, cyclones, etc.), in addition to trawling, and (ii) the biodiversity baseline of before-trawling does no longer exist and we could not find suitable and comparable untrawled areas, since all exploited benthos in south-west GoC show the results of 40+ years of continual trawling.

The environmental variables used as proxies for ecosystem processes did not show any clear and unambiguous relation with trawling intensities, nor did they seem to be affected by them. Giving the surveying period and within the limits imposed by sampling, the differences among water column and bottom sediment environmental variables and their associated ecosystem processes were largely explained by region and habitat, as well as small-scale temporal variability. There was, however, a significant effect of trawling on the trophic processes at the level of functional groups, taxa and species. Indeed, when the affected food web was simulated under current and historical fishing levels, fishing lowered the mean trophic levels of the catches (TL). That ecosystem effect occurred during the fishery expansion in the 1970s to mid and late 1980s. When fishing effort was reduced (starting in the mid to late 1980s) the TL increased (returned) steadily to the predicted values in 2010, close to those estimated for the mid 1970s. The simulation of the food web processes indicates unambiguously that the ecosystem is still influenced by trawling. Theses simulations also demonstrate that the reduction of fishing (from 286 vessels in 1981 to 52 vessels in 2009) has resulted in clear reductions of the overall impacts on biomass (bycatch) and trophic levels. These effects will however increase as fishing effort increases.

The two other objectives of this project were designed to make use of existing management data, analytical tools and models and to integrate them into an operational spatial management framework as an explicit form of EBFM. This project successfully integrated the bioeconomic stock and ecological risk assessment models with the food web, effect of trawling and species distribution models, all interacting in an adaptive form of a spatial management strategy evaluation framework (spatial MSE).

Since the spatial MSE combined various tools designed for different objectives (e.g. stock, economics, risk, biomass, etc.), it has the ability to evaluate multiple objectives, at multiple temporal, spatial and ecological scales. The results from the scenarios simulated so far are considered as proof-of-concept for the delivery of an operational spatial management framework. **These have not been yet agreed and evaluated with managers, industry and stakeholders in general, nor have they been applied to manage the fishing or to conserve biodiversity yet.**

The starting point is the achievement of the current fishery's stock and economic targets. This means that the overall fishing effort will remain unchanged across all scenarios and spatial redistribution of effort does occurs to adjacent and away areas around the spatial closures. The resulting fishing effort levels were used to evaluate the following scenarios: a base case BC (status quo), three ecological risk-based closures, one conservation network of marine protected areas (MPAs), and one adaptive closure derived from triggering trawling impacts thresholds. All scenarios use the same maximum economic yields (MEY) strategy as used at present. The delivered spatial MSE is able to evaluate simultaneously the ecosystem consequences of these multiple spatial management closures while achieving the NPF's fisheries management targets (stock and economics). This approach removes upfront the perceived conflicts between conservation and fisheries management objectives, in the case of evaluating MPAs.

The effects of the spatial management scenarios did not affect fisheries targets overall. It was found that at the current fishing effort levels, the trawling-induced changes are small and with little net variation across the range of performance measures and metrics used. This is evidence that the changes induced on the benthic ecosystems as whole are currently small and minor. These changes though are in an increasing trajectory as fishing effort increases over time, as prawns recover to biomass levels at maximum sustainable yields (B_{msy}). All spatial management scenarios tested exhibited high spatial variability and, as expected, most trawling effects were found on the tiger-endeavour fishing-affected habitats. However, these effects varied regionally across habitats and the affected ecological functional groups, taxa and species showed both positive and negative changes. We tested the spatial MSE to evaluate various likely spatial management objectives including; (i) biodiversity (biomass and trophic level, TL), (ii) sustainability of by-products, (iii) protection of habitat forming taxa (seagrasses, sessile epibenthos), (iv) reduction of species at risk, (v) protection of TE&Ps, and (vi) minimisation of effort displacement due to closures. Depending on the management objectives, the various simulated spatial management scenarios can act as best, nil and worst-case scenarios for any particular functional group, performance measure or evaluation metric. This is clearly shown in the case of TE&P species where closures tend to show that could increase and decrease the biomass of such taxa. Comparing across simulated scenarios, the more the fishing closures, the more responses across the whole ecosystem. The scenarios that have the least closures will have no or relatively little responses in relation to the BC (do nothing). This was clear when TE&P taxa were considered, where their high spatial and temporal variability of their responses was also the results of other-than-trawling-factors, such as movements, predation, closure edge effects, etc. We also found that the coarser the spatial scales for any given performance measure or metric, the less changes. These results highlight and confirm the need for spatial information at the same spatial scales of the impacts. No single scenario satisfied all or most management objectives. Evidence that if any fishing closures (or MPA network) is to be established in the northern region affecting the NPF, these need to be thoroughly assessed and tested against multiple criteria, performance measures and metrics derived from the objectives stated from the affected parties.

We believe that the spatial management framework delivered by this project can efficiently perform the above mentioned assessment and tests. However, if the closures are too large, the use of fishing history to predict fleet movements becomes questionable. **The improvement of the fleet dynamic models is probably the most important area for improvement of the spatial MSE.** Additionally, it is necessary to incorporate industry, managers and stakeholder's inputs for the desirable performance measures and in developing the likely spatial management scenarios. The managers and industry in particular, are now in a good position to address the challenges imposed by dealing with the off-reserve (outside of MPAs) effects of fishing on the species at risk, and to deal with the bioregionalisation of the northern regions of Australia. This spatial management framework in conjunction with other spatial-explicit systematic conservation tools, can evaluate alternative designs and placements of conservation MPAs within the areas of further assessment within these northern regions. Thus, this project has effectively provided the industry and managers with the tools and ability to respond to increasing management needs for environmental and conservation management policies. This project has also provided an operative integration of all existing management tools into a formal spatially-explicit ecosystem-based management framework. Overall, we believe that the results of this project provide a front-end example of the best-practices and tools needed for the implementation of an ecosystem-based management approach for the natural resources of the NPF, its conservation and overall environmental sustainability of Australia's marine ecosystems.

KEYWORDS:
Effects of trawling, ecological impacts, ecosystem processes, management strategy evaluation, spatial management, ecosystem-based fisheries management.

2 ACKNOWLEDGMENTS

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3 BACKGROUND

A full, spatially explicit, ecosystem approach to management of the NPF will consider all of the following: the target stocks; the economic and social consequences of alternative management strategies; the effects of the fishery on bycatch, byproduct, and benthic habitats, processes and communities; and indirect effects on other interacting components of the ecosystem. To date, research on the NPF has addressed some, but not all, of these components. Since the early 1990s, there has been significant research on the bycatch of the NPF (FRDC 1993/179), mainly aimed at reducing bycatch through turtle excluder devices (TEDs) and bycatch reduction devices (BRDs) (FRDC 1996/257, 1998/202, 2000/173, and 2002/064). Still, relatively little is known about the effects of fishing on the benthic habitats, communities and key ecological processes of the NPF.

The first study designed to address the impacts of trawling on benthic habitats, processes and communities was a desktop evaluation of historical environmental and biological data (Surrogates I, FRDC 2000/160, Hill et al. 2002). The aim was to identify environmental surrogates that could be used to characterise the impacts of prawn trawling in the NPF. This study, which relied mostly on historical data from prawn trawl samples, was only able to identify weak surrogates because of the lack of simultaneous, integrated and dedicated sampling for describing and characterising the different benthic habitats of the fishery. The Surrogates I project developed a basic and local MSE model for trawling impacts that has a good fine-scale trawling pattern model, but uses depletion parameters from the Great Barrier Reef and a theoretical recovery model that requires validation.

CSIRO also looked at the effects of trawling on the benthos of the NPF (FRDC 2002/102) with knock-down experiments that measure the depletion of benthos caused by a single trawling event and the recovery of the benthos following that event (Haywood et al. 2005). That project provided NPF parameters for the temporal component (depletion and recovery) of the MSE model developed in the Surrogates I project. The key information on the wider regional distribution of habitats, unavailable for past projects will, has been in part, provided by this project based on 2005 surveys.

The project objective of this report provides key information on ecosystem processes and benthic habitats at the broader spatial scale, and provides the models required to develop a spatial management framework that considers the environment and the target resources in the NPF. The project introduces: (a) recent changes in effort levels; (b) spatial management scenarios that evaluate the effects on benthic habitats and communities; and (c) informs on some aspects of indirect ecosystem effects into the likely spatial management of the NPF. Specifically, it develops and evaluates a range of management strategy scenarios for the complex spatially heterogeneous effects of prawn trawling on benthic habitats and communities. This project did build upon the outputs of other NPF-related projects by integrating its own results with those of the effects of trawling project (FRDC 2002/102), the bycatch risk assessment project (FRDC 2002/035), the Weipa ecosystem project (FRDC 2004/024), and bioeconomic stock assessments and the economic analysis projects (FRDC 2004/022).

The project also made full use of both new and historical data sets for the NPF, current scientific findings, expert inputs from the management and industry (NORMAC-REC and the NPF-RAG) as well as direct involvement of AFMA as a project member, and from members of the industry. The project had a start-up Steering Committee to oversee its commencement. Unfortunately, the committee only met once for the design and discussion of the workplan and did not continue their involvement, largely due to the changes in member's jobs and availability. The project PIs, however met and reported regularly to industry and managers, seeking directions and direct feedback from the NPF-RAG and AFMA managers.

4 NEED

Recent assessments of the NPF have identified a need for the fishery to be managed at a finer spatial scale than that of the NPF managed area (Hill et al. 2002, AFMA 2003, DEH 2003, Haywood et al. 2005). The stocks of some prawn species appear to comprise regional subpopulations that, although not genetically isolated, mix little enough to be manageable as separate stocks. This view is consistent with the experience that depleted stocks in some regions (e.g. north of Mornington Island and Weipa) have not recovered when stocks elsewhere in the NPF were healthy.

The assessments so far also identify a need to broaden the scope of management of the NPF beyond prawn stocks, iconic species and bycatch, to include benthic habitats and species. Prawn fishing has a number of impacts on the ecosystem. These include: removal of target species; removal of bycatch and byproduct; removal of benthic plants and animals; removal of habitat-forming species; disruption of sediment structure; suspension of sediment; and feeding of dolphins, sharks, seabirds, fish and benthic invertebrates with discards (Poiner et al. 1998, Haywood et al. 2005, Burridge et al. 2006, Pitcher et al. 2008). Some impacts, such as the removal of seagrass in nursery habitats, are known to negatively affect prawn stocks. Other impacts are likely to affect stocks in unknown ways, positively or negatively, and in some habitats may affect their sustainability.

Broadening management of the NPF to include impacts on benthic ecosystems is therefore prudent from both the environmental and industry viewpoints. It is also consistent with the recommendations of the NPF strategic assessment (DEH 2003), and will prepare the industry for the increasingly sophisticated environmental awareness of export markets.

Management of the NPF is currently based on sound bieconomic stock assessment and population monitoring procedures and uses maximum sustainable yield as the management limit reference point. Spatial stock assessment has been investigated with mixed success, but until 2008 was not used operationally. Current environmental management focuses on fragile habitats (mainly seagrass), prawn spawning areas, iconic species (e.g. turtles) and bycatch. Recently, ABARE suggested a move towards economic efficiency targets, such as maximum economic yield. (Rose and Kompas 2004). To do this FRDC project 2004/022 did integrate the existing stock and economic assessments into an MSE process (Dichmont et al 2008, Dichmont et al. 2010).

To enable stock, economic and environmental objectives to be effectively pursued in a spatial context with minimal conflict, the stock, economic, bycatch and ecosystem components of NPF management must be integrated into a single, spatially explicit management framework. This project will contribute with major missing elements for that integration and will develop this needed spatial management framework. Equally, the timing of this project is opportune given the bieconomic stock and ecological risk assessments. Monitoring is already mature, interactions with iconic species and bycatch are becoming well understood, and the integration of stock assessment with economics is currently underway and implementing towards output quota management. To achieve highly effective technical communication and integration, the PIs from past and present projects were involved as well as a Steering Committee with members from CSIRO, AFMA, and the NORMAC-REC and NPF-RAG committees.

5 OBJECTIVES

5.1 DETERMINE THE ACCUMULATED EFFECTS OF TRAWLING ON BENTHIC COMMUNITY STATE AND COMPOSITION.

5.2 QUANTIFY KEY BENTHIC ECOSYSTEM PROCESSES OF IMPORTANCE TO PRAWN PRODUCTION AND BIODIVERSITY ALONG A TRAWL INTENSITY GRADIENT.

5.3 DEVELOP, AND PROVIDE FOR ADOPTION MANAGEMENT STRATEGY EVALUATION TOOLS FOR BENTHIC ECOSYSTEM IMPACTS.

5.4 DESIGN AND DELIVERY OF A SPATIALLY EXPLICIT MANAGEMENT FRAMEWORK FOR THE NPF.

6 METHODS

This section has been organized according to the different methodologies and approaches used to achieve the project's objectives. The project relied largely on the NPF's historical information and the data generated in the research surveys conducted in the south-western Gulf of Carpentaria (GOC) in March–April 2005 (RV Southern Surveyor voyages SS 03/2005 & 04/2005). These surveys collected biological and environmental samples within an experimental design used principally to assess the impact of trawling on the benthic communities and its likely effect on key ecological benthic processes (Objs. 1 and 2 respectively). The resulting data were then used to develop modelling and analytical tools to provide the NPF with a spatially explicit management strategy evaluation (MSE) framework (Objs. 3 and 4). A succinct description of the general methods used for the acquiring, processing, analysis and integration of this research towards the achievement the project's objectives is presented below.

6.1 OBJECTIVE 1

6.1.1 BIOPHYSICAL SURVEY

To achieve this objective, we conducted a research survey using the RV Southern Surveyor between 23 February and 21 March 2001 (SS 03/2005), following the track depicted in Figure 1a. The survey design was based on a natural experiment in which the environment, habitats, species and processes were sampled or observed at sites in three main prawn fishing grounds: north of Mornington Island, north-east of the Vanderlins and north of Groote Eylandt (Figure 1 b, c, and d, respectively). These regions are the main representative areas of a trawl fishing gradient, covering some of the main tiger–endeavour prawn fishing grounds in the south-western GOC. They are also the regions whose benthos experiences the most intensive trawling in the GOC. In addition, the numerous contrasting environments identified by Hill et al. (2002) indicate that these areas provide good contrast between habitats (reef platforms, soft sediments, deep siliclastic deposits) and water masses (strong productivity, salinity, oxygen and water movement gradients).

From the NPF's historical logbook and recent vessel monitoring system (VMS) data, it was not possible to confidently identify suitable prawn fishing grounds (i.e. soft sediments at depths between ~15 m and 45 m) that had not been trawled at least once, so there were no real control sites for a fully before vs. after and control vs. impact (BACI) experimental design (Smith 2002). Consequently, fishing grounds were identified with fishing intensity maps created with the 1999–2003 VMS data alone, using the method developed by Surrogates 1 (FRDC project 2000/16). Low trawl sites had zero or very low trawling in the five-year period covered by the fishing intensity maps, while medium and high trawling sites were fished reasonably consistently throughout the five-year period and in an increasing fashion. The trawl intensity stratification derived from the VMS data is described in Figure 1 which shows a clear gradient of fishing effort, increasing 3-fold, from low to high.

Table 1. Descriptive statistics for the three trawling intensities, measured as hours trawled per year at each nautical mile squared (NM^2) of tiger-endeavour fishing ground.

Trawl Intensity	Mean	s.d.	Min.	Q05	Q25	Median	Q75	Q95	Max.
Low	0.194	0.232	1.000	1.000	1.000	0.103	0.309	0.733	0.959
Medium	16.161	5.405	6.156	6.790	12.271	17.335	20.815	23.339	23.994
High	63.020	21.710	24.429	28.767	46.154	62.119	79.834	97.825	123.386

Constrained by the survey duration (~25 days) and the ability to process samples on board, each of the three sampling regions was allocated around 42 sampling sites evenly distributed among the three fishing strata and day/night periods. Sampling sites were distributed randomly within the fishing strata, and once on site, manually edited and/or changed to avoid hard or unsuitable ground. The in-situ changes also included historical sampling sites and the elimination of extreme random clumping. To maximise the power of the survey to detect trawling effects, each sampling region was chosen to be relatively uniform in attributes other than trawling intensity (e.g. habitat type and environmental variability). However, to maximise the generality of our conclusions, the sampling regions were also chosen to differ environmentally from one another (regional effects identified by Hill et al. 2002). The sampling regions were thus chosen following consultation with field ecologists and multivariate analysis of environmental data from FRDC project 2000/160 and the databases compiled for the North Marine Region for the Department of the Environment, Water, Heritage and the Arts (DEWHA) marine bioregional planning process (http://www.environment.gov.au/coasts/mbp/north/index.html).

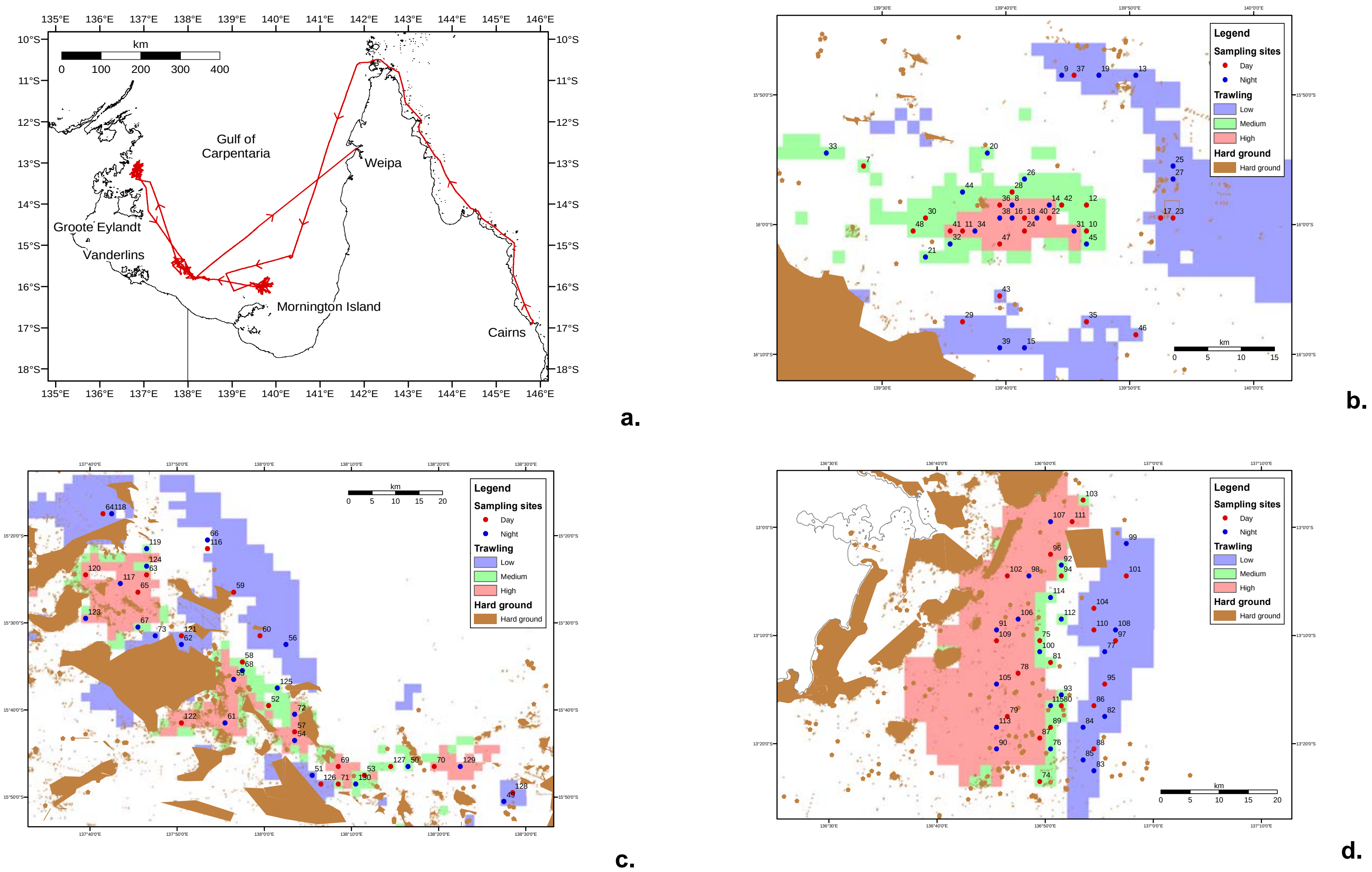

Figure 1. (**a**.) Cruise track (~7,000 km) for SS 03/2005 departing Cairns on 23 February 2005 and arriving at Weipa 21 March 2005. Maps show the sampling locations in Mornington (**b**.), Vanderlins (**c**.), and Groote Eyland (**d**.) regions, illustrating the random allocation of day and night stations with respect to high, medium, and low trawling intensities.

Table 2. SS 03/2005 sampling station number (site), location (lat-long), dates and regions.

Site	Date	Latitude	Longitude	Region	Site	Date	Latitude	Longitude	Region
1	2005-02-26	−15.26	140.36	EAST GOC	66	2005-03-09	−15.34	137.89	Vanderlins
2	2005-02-26	−15.26	140.29	EAST GOC	67	2005-03-09	−15.51	137.76	Vanderlins
3	2005-02-26	−15.28	140.30	EAST GOC	68	2005-03-10	−15.59	137.96	Vanderlins
4	2005-02-26	−15.31	140.30	EAST GOC	69	2005-03-10	−15.77	138.14	Vanderlins
5	2005-02-27	−15.70	138.87	EAST GOC	70	2005-03-10	−15.77	138.33	Vanderlins
6	2005-02-27	−16.10	139.04	EAST GOC	71	2005-03-10	−15.81	138.14	Vanderlins
7	2005-02-27	−15.92	139.48	Mornington	72	2005-03-10	−15.67	138.06	Vanderlins
8	2005-02-27	−15.97	139.68	Mornington	73	2005-03-10	−15.52	137.79	Vanderlins
9	2005-02-28	−15.81	139.74	Mornington	74	2005-03-11	−13.39	136.83	Groote
10	2005-02-28	−16.01	139.78	Mornington	75	2005-03-11	−13.17	136.83	Groote
11	2005-02-28	−16.01	139.61	Mornington	76	2005-03-11	−13.34	136.84	Groote
12	2005-02-28	−15.97	139.78	Mornington	77	2005-03-12	−13.19	136.93	Groote
13	2005-02-28	−15.81	139.84	Mornington	78	2005-03-12	−13.22	136.79	Groote
14	2005-02-28	−15.97	139.73	Mornington	79	2005-03-12	−13.29	136.78	Groote
15	2005-03-01	−16.16	139.69	Mornington	80	2005-03-12	−13.27	136.86	Groote
16	2005-03-01	−15.99	139.68	Mornington	81	2005-03-12	−13.21	136.84	Groote
17	2005-03-01	−15.99	139.88	Mornington	82	2005-03-12	−13.29	136.93	Groote
18	2005-03-01	−15.99	139.69	Mornington	83	2005-03-13	−13.37	136.91	Groote
19	2005-03-01	−15.81	139.79	Mornington	84	2005-03-13	−13.31	136.89	Groote
20	2005-03-02	−15.91	139.64	Mornington	85	2005-03-13	−13.36	136.89	Groote
21	2005-03-02	−16.04	139.56	Mornington	86	2005-03-13	−13.27	136.91	Groote
22	2005-03-02	−15.99	139.73	Mornington	87	2005-03-13	−13.32	136.83	Groote
23	2005-03-02	−15.99	139.89	Mornington	88	2005-03-13	−13.34	136.91	Groote
24	2005-03-02	−16.01	139.69	Mornington	89	2005-03-13	−13.31	136.84	Groote
25	2005-03-02	−15.92	139.89	Mornington	90	2005-03-13	−13.34	136.76	Groote
26	2005-03-02	−15.94	139.69	Mornington	91	2005-03-13	−13.16	136.76	Groote
27	2005-03-03	−15.94	139.89	Mornington	92	2005-03-14	−13.06	136.86	Groote
28	2005-03-03	−15.96	139.68	Mornington	93	2005-03-14	−13.26	136.86	Groote
29	2005-03-03	−16.12	139.61	Mornington	94	2005-03-14	−13.07	136.86	Groote
30	2005-03-03	−15.99	139.56	Mornington	95	2005-03-14	−13.24	136.93	Groote
31	2005-03-03	−16.01	139.76	Mornington	96	2005-03-14	−13.04	136.84	Groote
32	2005-03-03	−16.02	139.59	Mornington	97	2005-03-14	−13.17	136.94	Groote
33	2005-03-04	−15.91	139.43	Mornington	98	2005-03-14	−13.07	136.81	Groote
34	2005-03-04	−16.01	139.63	Mornington	99	2005-03-15	−13.02	136.96	Groote
35	2005-03-04	−16.12	139.78	Mornington	100	2005-03-15	−13.19	136.83	Groote
36	2005-03-04	−15.97	139.66	Mornington	101	2005-03-15	−13.07	136.96	Groote
37	2005-03-04	−15.81	139.76	Mornington	102	2005-03-15	−13.07	136.78	Groote
38	2005-03-04	−15.99	139.66	Mornington	103	2005-03-15	−12.96	136.89	Groote
39	2005-03-05	−16.16	139.66	Mornington	104	2005-03-15	−13.12	136.91	Groote
40	2005-03-05	−15.99	139.71	Mornington	105	2005-03-15	−13.24	136.76	Groote
41	2005-03-05	−16.01	139.59	Mornington	106	2005-03-15	−13.14	136.79	Groote
42	2005-03-05	−15.97	139.74	Mornington	107	2005-03-16	−12.99	136.84	Groote
43	2005-03-05	−16.09	139.66	Mornington	108	2005-03-16	−13.16	136.94	Groote
44	2005-03-05	−15.96	139.61	Mornington	109	2005-03-16	−13.17	136.76	Groote
45	2005-03-05	−16.02	139.78	Mornington	110	2005-03-16	−13.16	136.91	Groote
46	2005-03-06	−16.14	139.84	Mornington	111	2005-03-16	−12.99	136.88	Groote
47	2005-03-06	−16.02	139.66	Mornington	112	2005-03-16	−13.14	136.86	Groote
48	2005-03-06	−16.01	139.54	Mornington	113	2005-03-16	−13.31	136.76	Groote
49	2005-03-06	−15.84	138.46	Vanderlins	114	2005-03-17	−13.11	136.84	Groote
50	2005-03-07	−15.77	138.28	Vanderlins	115	2005-03-17	−13.27	136.84	Groote
51	2005-03-07	−15.79	138.09	Vanderlins	116	2005-03-17	−15.36	137.89	Vanderlins
52	2005-03-07	−15.66	138.01	Vanderlins	117	2005-03-17	−15.42	137.73	Vanderlins
53	2005-03-07	−15.79	138.19	Vanderlins	118	2005-03-18	−15.29	137.71	Vanderlins
54	2005-03-07	−15.72	138.06	Vanderlins	119	2005-03-18	−15.36	137.78	Vanderlins
55	2005-03-07	−15.61	137.94	Vanderlins	120	2005-03-18	−15.41	137.66	Vanderlins
56	2005-03-08	−15.54	138.04	Vanderlins	121	2005-03-18	−15.52	137.84	Vanderlins
57	2005-03-08	−15.71	138.06	Vanderlins	122	2005-03-18	−15.69	137.84	Vanderlins
58	2005-03-08	−15.57	137.96	Vanderlins	123	2005-03-18	−15.49	137.66	Vanderlins
59	2005-03-08	−15.44	137.94	Vanderlins	124	2005-03-18	−15.39	137.78	Vanderlins
60	2005-03-08	−15.52	137.99	Vanderlins	125	2005-03-19	−15.62	138.03	Vanderlins
61	2005-03-08	−15.69	137.93	Vanderlins	126	2005-03-19	−15.81	138.11	Vanderlins
62	2005-03-09	−15.54	137.84	Vanderlins	127	2005-03-19	−15.77	138.24	Vanderlins
63	2005-03-09	−15.41	137.78	Vanderlins	128	2005-03-19	−15.82	138.48	Vanderlins

A total of 130 sampling stations were surveyed (Table 2), where a range of biological and physical data and samples were collected. Some data were measured directly on board, while other data required processing of samples on land after the voyage. All post-voyage processing was conducted in the CSIRO Cleveland and Hobart labs, Griffith University, Geoscience Australia in Canberra, and the Queensland Museum. This generated a unique, integrated dataset that includes acoustics, geomorphology, sediments, benthic biota and other physical and ecological data. At each sampling station we (a) sampled benthic sessile and mobile biota (invertebrates mostly) with an epibenthic sled, (b) sampled demersal fish with a prawn trawl, (c) collected surface sediment box core samples, (d) collected water samples, (e) measured water properties with the Southern Surveyor's CTD/turbidity meter, and (f) measured water and sediment flux with a current meter instrument frame on-board ADCP.

Of the total 130 stations, 124 were from the natural experiment component of the survey (Table 3). The remaining six were visited for other voyage objectives.

Table 3. Number of sampling stations combining surveyed sites and regions as part of the natural experiment component to evaluate the impact of trawling

	Trawling Intensity						
	Low		**Medium**		**High**		
Region	**Day**	**Night**	**Day**	**Night**	**Day**	**Night**	**Total**
Groote Eyland	7	7	7	7	7	7	42
Vanderlins	7	7	5	7	7	7	40
Mornington Island	7	7	7	7	7	7	42
Total	21	21	19	21	21	21	124

All resulting samples from the prawn trawl and the epibenthic sled were frozen and/or preserved and transported to the CMAR research laboratories in Cleveland for detailed sorting, counting, sizing, weighing, and identification of all known species and alpha classification of unknowns using the specimen and photograph collections maintained by CSIRO. The fish taxonomy group at CSIRO in Hobart provided taxonomic oversight and curation of reference specimens. Seabed benthic fauna was identified by CSIRO and the Queensland Museum and specimens were deposited in the Northern Territory and Queensland museums. Samples of benthic macro-infauna were obtained with a sediment box-corer sampling 0.05 m^2 of sea floor. Sediment containing benthic macro-infauna was taken at 14 sites for each of the three trawl intensity strata in each of the three regions of the GOC. Benthic macro-infauna were then separated from the sediment using a 500-µm (0.5-mm) screen, sorted by taxon, counted, wet weighed, stained (with Rose of Bengal) and preserved in 1-3% seawater-buffered formalin.

6.1.2 DATA ANALYSES

To evaluate the trawling impacts, statistical techniques were used to identify key benthic indicators that are most sensitive to trawling and can reflect the state and extent of the trawling impact on the benthic ecosystem. Variation in benthic biodiversity among the survey trawling strata was described and tested with univariate analysis of ecological indicators and multivariate analysis of catch composition. The ecological indicators were abundance and diversity indices. The univariate methods included box-and-whisker plots and ANOVA, while the multivariate methods were transformation based principal components analysis (PCA), redundancy analysis (RDA), variation partitioning and nonparametric multivariate analysis of variance (NPMANOVA). The transformation based variations of PCA and RDA eliminate the ecological disadvantages of ordinary PCA and RDA relative to correspondence analysis while retaining the advantage of their use of Euclidean distance (Legendre and Gallagher 2001). All analyses were performed with the `rda` function of the `vegan` library of the R computer program (R Development Core Team 2007).

6.2 OBJECTIVE 2

The methods employed here included the assessment of biotic and abiotic indicators at all and selected sampling stations where key benthic processes related to ecosystem functioning (diet, food web and benthic processes) were assessed, specifically in those likely to contribute to prawn production and maintenance of biodiversity. Analyses included; fish diets, isotope signals, benthic primary productivity and respiration.

6.2.1 WATER-COLUM VARIABLES

Table 4 lists the numbers of samples taken in all three regions and different trawling intensities for a range of water column variables above each of the benthic sampling stations (Table 4a) and various benthic and sedimentary variables (Table 4b).

During the RV Southern Surveyor voyage SS 03/2005 a large dataset of water column oceanographic information was collected to provide the hydrological context of the water column processes likely related to the benthos. Conductivity-Temperature-Depth (CTDs) casts were performed at each sampling site to provide vertical profiles of temperature, salinity, oxygen and fluorescence. In addition, water samples were collected at the surface and every 10 m through the water column for dissolved nutrient (nitrate/nitrite, ammonium, phosphate, silicate) concentrations. Periodically, photosynthetic activity (yield) of surface and bottom waters was also measured. The Acoustic Doppler Current Profiler (ADCP) was run continuously through the voyage to measure current speed and direction at multiple levels in the water column.

6.2.2 BENTHIC VARIABLES

During the Southern Surveyor 03/2005 voyage, mini sediment cores were collected from each of the box cores taken at every sampling site. The cores were used for a number of analyses: denitrification rates; sediment oxygen demand rates; stable isotope signatures; and carbon and nitrogen content. Sediment oxygen demand (SOD) incubations were carried out (in the dark) on-board, as was data analysis. Due to the need to undertake 24 h incubations, only three to four sites for each trawling intensity, for each region, were completed. Cores for denitrification measurements were also incubated on-board but analysis of samples by mass spectrometry for stable isotope signatures, as well as carbon and nitrogen contents, were conducted in the CSIRO Hobart lab.

Triplicate sediment cores were collected at every sampling site. The top 2 cm of each undisturbed core were sliced off and frozen until returned to the laboratory. In the laboratory, sediments were screened through a wire mesh screen to remove rocks and rubble, dried at 60°C and ground. Half the sample had 1M HCl added, the sediment stirred, and then allowed to effervesce in the fume hood to dissolve the carbonaceous material. Once effervescence stopped, the acidified sample was dried at 60°C for 24 h. Samples were then analysed in a mass spectrometer to determine δ13C-carbon ratios and %carbon concentrations. The other half of the sample, which was not acidified, was analysed in a mass spectrometer to determine δ15N-carbon ratios.

Sediment oxygen demand (SOD) incubations were carried out in the dark on-board in sealed Perspex chambers fitted with oxygen electrodes and pumps circulating water above the sediment. Changes in oxygen concentration were recorded every 10 min for 24 h in duplicate chambers. The number of sites analysed were constrained by equipment and staff availability. As a result, only three to four sites were completed for each trawling intensity and region.

Sediment cores were collected at 27 sites for denitrification measurements (i.e. nitrogen gas production from microbial activity). Within the different trawling strata of high, medium and low, three sites were sampled and incubations conducted.

Infauna was also sorted from triplicate sediment box cores at six sites within each region for stable isotope analysis. Infauna was grouped into crustaceans, molluscs, worms and fish. Numbers of animals were typically low. Samples were dried at 60°C for 24 h. Samples were ground and half the sample was acidified as described above for the determination of δ13C-carbon. The other half was analysed without acidification for δ15N-nitrogen ratios. All samples were analysed in a mass spectrometer.

Statistical analyses, using SAS (Version 9.1) software, were performed to examine differences in the organic carbon and δ13C-carbon ratios in the sediment samples between regions and trawling strata. Both organic carbon and δ13C-carbon ratios were square-root transformed before being analysed using PROC GLM.

Table 4. Number of samples taken at each surveyed region and trawling intensity to characterise (a) water column and (b) sedimentary benthic habitats. * = sample number too low, not included in most analyses.

	High			**Medium**			**Low**		
	Groote	**Mornington**	**Vanderlins**	**Groote**	**Mornington**	**Vanderlin**	**Groote**	**Mornington**	**Vanderlins**
(a) Water column habitat									
Salinity (PSU)	14	13	14	14	13	14	13	13	12
Oxygen (µM/L)	12	13	14	14	14	14	14	13	12
Phosphate (µM/L)	14	13	14	14	14	14	14	13	12
Nitrate (µM/L)	14	13	14	14	14	14	14	13	12
Silicate (µM/L)	14	13	14	14	14	14	14	13	12
Ammonia (µM/L)	14	13	14	14	14	14	14	14	12
(b) Sedimentary benthic habitat									
Oxygen_demand (µmol/m^2/h)*	4	4	4	4	4	2	3	3	6
Denitrification (µmol/m^2/h)*	3	3	3	3	3	2	3	5	4
Carbon (%)	14	14	14	14	14	12	14	14	13
δC13 (ppm)	14	14	14	14	14	12	14	14	13
Carbonate (%)	14	14	14	14	13	12	14	13	14
Gravel (%)	14	14	14	14	13	12	14	13	14
Sand (%)	14	14	14	14	13	12	14	13	14
Mud (%)	14	14	14	14	13	12	14	13	14

6.2.3 DIET ANALYSES

To explore the broad-scale direct and indirect effects of historical prawn trawling on the benthos, we analysed the diet of a series of key benthic predatory fish species. We designed this under the assumption that if there were any impact on the trophic structure and function of the trawled benthos, its effects would be more obvious on the benthic predatory species that prey on demersal prey. The trawl samples were taken with a standard industry ‘Florida Flyer’ style NPF prawn net – i.e. headrope length of 21.2 m (12 fathoms), with a diamond net pattern of 55 mm (~2 inch) stretch-mesh size and Bison #9 trawl boards. The net was towed over the seabed at a median speed of 1.6 m/s for a median distance of 1,064 m.

The predator fish species chosen for dietary analysis were those that were benthic feeders and representative of the demersal fish assemblage most likely to be affected by trawling (Harris & Poiner 1991, Pender et al. 1992, Blaber et al. 1994, Stobutzki et al. 2001). The fish samples were frozen at sea (–20˚C) and later freighted to CSIRO laboratories where they were thawed and their stomachs removed. Prey items were extracted from small teleost stomachs and identified under a binocular microscope to the lowest taxonomic level possible. The items termed ‘Natantia’ were Decapods that were prawn or shrimp like as per Grey et al. (1983). The prey were then weighed (wet weight, 0.000 g), counted and measured where possible (mm). The stomach contents of each predator species were examined in detail to provide a complete list of identified prey taxa, prey biomass and prey frequency of occurrence data.

A stomach fullness index was calculated for each predator in order to explore whether there was variation in the biomass of prey consumed in different regions and fishing intensities. This was calculated using the equation:

$$\text{Stomach fullness index } = \frac{\text{total wet weight of prey}}{\left(\text{predator wet body weight}\right) \ - \ (\text{total wet weight of prey}}$$

Statistical comparisons of the diets of the selected predator fish species were made between regions and fishing intensities using the multivariate analysis procedures in PRIMER (Version 6.1.10, Clarke & Warwick 2001). In preparation for the multivariate analyses, the stomach samples were randomly allocated to one of four replicates for each trawl regions and fishing intensities per predator species. This was done to incorporate the natural variability in diet composition that often occurs between individual stomach samples. Prey items were represented in terms of biomass as percentage wet weight.

The data were square-root transformed to reduce the influence of highly weighted taxa and a similarity matrix was constructed using Bray-Curtis similarity coefficients for the prey biomass (%) for each replicate. The resulting predator-prey matrices were then analysed using non-metric multidimensional scaling (nMDS) and analysis of similarities (ANOSIM) to investigate differences in diet between species, between regions for each species, between fishing intensities at each region, and between fishing intensities over all regions. Similarity percentages (SIMPER) were used to identify the prey responsible for significant differences between a priori groups as determined by ANOSIM. This procedure identified the prey taxa that contributed to the dissimilarity between groups (high and low fishing intensity), and the similarity within each group (at each fishing intensity).

Apart from investigating differences in diet composition between regions and fishing intensity for each species, it was of interest to investigate whether the total amount (wet weight) of prey consumed by fish in high and low fishing intensity areas differed. As some species were not caught in sufficient numbers in some regions to facilitate a balanced full-factorial design, separate two-factor ANOVAs were performed for each species. This allowed testing for differences in the mean stomach fullness index between regions (Groote, Mornington and Vanderlin) and fishing intensities (high and low). Both factors were considered fixed. Cochran's and Shapiro-Wilk's tests were used to analyse homogeneity of variances and normality of the data, respectively (Zar 1984). Data for each species were log10 (x + 1) transformed before analysis in an attempt to stabilise heteroscedastic variances, which was not successful in all cases. Following the recommendations of Underwood (1981), analyses were undertaken on the transformed data, with alpha set to 0.01 to minimize the chances of incurring Type I errors. Tukey's Honestly Significant Difference (HSD) tests were used for a posteriori comparison of means.

6.3 OBJECTIVE 3

To achieve this objective we put together various existing models and tools using the resulting data and information basis generated in Objectives 1 and 2 and complemented by existing historical and NPF management-related projects (e.g. from FRDC 2000/160, FRDC 2002/102, FRDC 2004/022, FRDC 2004/024, FRDC 2001/002, FRDC 2004/022). For this, we developed, modified and adapted various existing analytical and predictive models and tools developed for the management of the NPF. Specifically, this project made use of: (i) regional and fine scale fleet dynamics (Venables et al, 2009), (ii) a benthic effects of trawling (EoT) model (Ellis and Pantus 2001), (iii) a multivariate species distribution model (Browne et al. unpub. ms.), (iv) a spatially explicit trophic mass-balance model (Christensen & Pauly 1992, Pauly et al. 2001 and Christensen & Walters 2004), (v) the ecological risk assessment (ERA) model for the NPF (Zhou et al. 2008), and (vi) the bioeconomic stock assessment model for the NPF and its related MSE (Dichmont et al. 2008).

6.3.1 EFFECTS OF TRAWLING

This model was developed originally for use in the Great Barrier Reef Marine Park (Ellis and Pantus, 2001) and has been modified for use in three FRDC projects (FRDC 2000/160, FRDC 2002/102 and FRDC 2004/022). In the latter project the model was used in the operating model to provide a performance measure of the status of the benthic biota. In this project the EoT model is used as an *assessment* model i.e. a model used in the management strategy: the performance measures have been adapted to provide indicators for the spatial management of prawn trawling in the NPF. The material in this section has been adapted from Venables et al (2009) in which the EoT model was used in detail and project FRDC 2004/022 describes the methods.

6.3.1.1 MODEL EQUATION AND PARAMETERS

The EoT model estimates the primary effects of repeated trawling on the biomass of benthic organisms (ignoring any long-term consequences of the removal on the ecosystem, including any effect on prawn productivity). In each 6-min grid cell, *g*, a Schaefer-like biomass-dynamic differential equation operates for each benthic species. Prior to fishing, the species is assumed to be at carrying capacity. The dynamics of each species, *s*, is completely determined by (i) the initial biomass, $B_0^{s,g}$, (ii) the recovery rate, r^s, (iii) the depletion rate per unit effort rate, d^s and (iv) the effort rate in the grid, E^g. The differential equation is as follows:

$$\frac{dB^{s,g}}{dt} = r^s B^{s,g}\left(1 - \frac{B^{s,g}}{B_0^{s,g}}\right) - d^s E^g B^{s,g} \tag{1}$$

where $B^{s,g}(t)$ is the biomass at at time *t*, and $E^g(t)$ is the effort in grid cell *g* at time *t* in units of proportion of grid cell area swept per unit time.

This biomass-dynamic logistic equation, which operates on the large scale (6-min squares), arises from the scaling up of impacts operating at the small scale (20m squares within the path of the trawl net). Ellis and Pantus (2001) have shown how recovery and depletion rates measured at the 20m scale can be converted into recovery and depletion rates at the 6-min grid scale of Eq. (1), if the distribution of fishing at the 20m scale is known. In this application we have assumed random trawling within each 6-min grid; the consequence is that the coefficient d^s in Eq. (1) is the same as the depletion rate per tow, which has been measured experimentally. The assumption of random fishing is conservative because, for this fishery, trawling is somewhat aggregated at the sub-grid cell level (Deng et al., 2005), which means that grid-cell scale impacts will be slightly over-estimated by the EoT model.

Because the EoT model is used here as an assessment model, it must not have any knowledge of the operating model, in particular, of the spatial distribution of the benthic biomass in the operating model. The benthic spatial distribution arises from best current knowledge as described in section 6.3.2 below. This information is used to calibrate the operating model (Appendix 10.3.2) and so cannot be used in the assessment model. Therefore in the EoT model we assume no prior knowledge about the spatial distribution of benthic biota; instead we assume that the initial distribution prior to fishing is spatially uniform (i.e. $B_0^{s,g} = 1$).

The values for the parameters of Eq. (1) for each group of benthic species (Table 5) were based on a series of depletion–recovery experiments on the Great Barrier Reef (GBR, FRDC Project 2000/160 and FRDC 2002/102) (Burridge et al., 2003; Poiner et al., 1998; Pitcher et al., 2007a,b), and the NPF (Hill et al., 2002; Haywood et al., 2005), with the NPF values taking precedence over the GBR values. The types of fauna chosen for modelling were a variety of taxonomic classes of benthic fauna that were found on the SS 03/2005 research voyage at all three regions, north of Groote Island, the Vanderlins, and north of Mornington Island. . Some groups (marked with a # in Table 5) were aggregates of multiple species; in these cases the recovery and depletion rates were derived from a combination of sources and a representative rate proposed by an expert panel.

Although several studies have examined the rate at which benthos is removed by trawling, there is little information on the rate at which benthos recovers from impacts. There is some good quantitative information for benthos in the GBR which was monitored for 5 years following a trawl depletion experiment (Pitcher et al, 2004). Studies conducted by FRDC 2000/160 and FRDC 2002/102 carried out an evaluation of the vulnerability and recovery of the NPF benthos to trawling as part of the estimation of the sustainability of the benthos. In that evaluation, each benthic group was scored on five sustainability criteria and the average score was related linearly to the recovery time, τ, which was the time taken for biomass to recover from 50% carrying capacity to 95% carrying capacity. The recovery rate is thus $r^s = 3/\tau$.

Table 5. Recovery (r^s) and depletion (d^s) rates for species (functional groups) in the EoT model. Species marked (*) were used to provide indicators to the spatial management. For species marked (#) the recovery and depletion rates were derived from a combination of sources.

Species, *s*	r^s	d^s	Species, *s*	r^s	d^s
Annelids detrit/carn#	1.50	0.01	Mud crab	0.52	0.13
Asteroids	0.97	0.11	Ophiuroids	0.63	0.03
Bivalves	0.52	0.10	Other commercial prawns	0.52	0.13
Echinoids#*	0.40	0.14	Other non-commercial prawns	0.52	0.13
Holothurians*	0.56	0.16	Sand crab and other large crabs	0.52	0.13
Large gastropod carnivore*	0.41	0.20	Sessile epibenthos#*	1.04	0.27
Lobsters	0.52	0.13	Spatangoids#	0.40	0.07

6.3.1.2 EXTENT OF THE MODEL

For this project, our focus was the tiger-endeavour fishery. This fishery targets up to four species of prawn, where fishing involves long duration (generally 3 hour) trawls that are in contact with the seabed the whole time. We therefore considered that, of all the fisheries in the GoC, this fishery is likely to have the greatest impact on the seabed fauna. The benthic model was restricted to the area that was trawled between 2000 and 2004 (years in which the fleet was smaller than 100 vessels, and the length of the fishing season was similar to the present).

6.3.1.3 INDICATORS FOR SPATIAL MANAGEMENT

Although the model was originally designed to produce time-series of the biomass of benthic taxa as a performance measure within an operating model, in this project the model was used as an assessment model instead. We adapted the model so that it would provide indicators to the spatial management model (section 6.4.1).

The indicators are estimates of the long-term biomass of the most vulnerable species within each habitat. The most vulnerable species, as measured by the ratio of depletion rate to recovery rate, were large gastropod carnivores, echinoids, holothurians and sessile epibenthos. The habitats corresponded to the habitats of the EwE model (section 6.3.3). However, since the assessment model should not know the details of the operating model, we used a modified definition of the habitats instead, in which the grid cells near the boundary of a habitat were randomly assigned to the neighbouring habitat. This simulates the fact that in reality we would not know with perfect accuracy the true habitats on the sea bed.

The calculation of the indicators is relatively simple and does not in fact rely on having the full time series of $B^{s,g}(t)$. The EoT model keeps track of $\overline{E}(t)$, the average annual total effort, and $\overline{C}^{hg}(t)$, the spatial effort pattern, over the most recent 5 years. The quantity $\overline{C}^{hg}(t)$ is the average trawl coverage in grid cell *g* within habitat *h* over the 5 years prior to time *t*, where coverage is the average number of tows of the trawl gear over any patch within the cell. A trawl width of 28m, a trawl speed of 3 knots and a time spent trawling per boat day of 10 hours were assumed in order to convert boat days to coverage. For a species *s*, the long-term relative biomass in a grid cell *g* under Eq. (1) with constant coverage $E^g(t) = C^g$ is

$$B_{\infty}^{s,g} = \begin{cases} 1 - d^s C^g / r^s & C^g < r^s / d^s \\ 0 & C^g \geq r^s / d^s \end{cases} . \tag{2}$$

(This is seen by setting the left-hand side of Eq. (1) to zero and solving for $B^{s,g}$.) For each habitat *h* and species *s*, the indicator I_{hs} is the average long-term relative biomass under a constant effort regime at E_{MEY} (section 7.2.4). Under this regime, the distribution of effort is assumed to follow the distribution for recent years, so the projected coverage in future years, from the viewpoint of year *t* is

$$C_{\infty}^{h,g}(t) = \overline{C}^{h,g}(t) E_{MEY}(t) / \overline{E}(t). \tag{3}$$

This projected coverage is substituted for C^g in (2) to provide an estimate of $B_{\infty}^{s,g}(t)$. Then the indicator is simply

$$I_{hs}(t) = \frac{1}{N_h} \sum_{g \in h} B_{sg}^{\infty}(t), \tag{4}$$

where the summation is over all grid cells in habitat *h*, and N_h is the number of grid cells in habitat *h*.

6.3.2 SPECIES DISTRIBUTION MODEL

We adapted and used a regression tree[1] method for the biophysical and spatial prediction of GoC benthic biota. The technique, commonly known as *random forests*, first proposed by Breiman (2001), is a powerful non-parametric estimation method. In the field of ecology, random forests have been considered by Prasad et al. (2006) and were applied by Garzon et al. (2006), Garzon et al. (2007) for estimating species distribution and habitat suitability, as well as by Schwartz et al. (2006) for predicting future species distributions.

Random forests may be described as bagging regression trees. That is, many bootstrap samples are selected from the data and a binary decision tree is fit to each. When splits are made at each node in each tree, a subset of predictors is chosen at random for evaluation. The predictor and the split point

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[1] Trees are a standard tool in statistical pattern recognition and will not be described in detail here. Readers may refer to any text in pattern recognition or machine learning, e.g. Duda et al. 2001 for an introduction.

are chosen so as to minimize the sum of squared residuals. Trees constituting the random forest are not pruned using cross-validation, but are instead fully grown (i.e. over-fit). These measures act to increase the random variability of the constituent trees within the forest so as to reduce correlation in the errors of the ensemble of estimators. Given sufficient independence between estimators, some ability of the trees to approximate the underlying function and a reasonably large number of trees, then the final estimate produced by averaging estimates over the ensemble can be expected to have good bias/variance properties (Boinee et al., 2005).

As an estimator, random forests are based on sound bootstrapping principles (Davison and Hinkley, 2003), and a growing body of empirical results suggests random forests are an effective applied estimation tool. By averaging estimates produced by a large set of trees, they avoid the instability, model uncertainty, and much of the bias associated with any single regression tree. A significant attraction of random forests is the manner in which they satisfy the competing goals of high model flexibility and strong generalization ability.

As mentioned above, the tree constituting the random forest estimator is based on optimizing sums of squares or sums of absolute deviations. From standard regression theory, it is known that departures from normality decrease the efficiency of such estimators (Box and Cox 1964). In realistic ecological surveys, outliers often exist, and the variation of the response may be related to the mean estimates. Non-constant variation gives greater weight to data with higher variation, and therefore, as with other forms of regression, it is often desirable to transform the response variable when applying trees (De'Ath 2002).

We therefore developed a procedure where the response variable *y* (the species density) was transformed according to a power law:

$$y^{(\varphi)} = y^{\varphi}; \quad \varphi \in [0,1]$$

The exponent ϕ was estimated using a procedure analogous to the likelihood maximisation method of Box and Cox (1964). Here, we replaced the likelihood with Kendall's rank-correlation coefficient between the estimated transformed response and the observed transformed response. The value of ϕ that maximised the rank-correlation coefficient was used. The theory behind the likelihood approach of Box and Cox (1964) does not strictly extend to random forests because they are not a linear model. However, given that the random forest estimator is optimized so as to minimize the residual (square) deviations, we have a strong expectation that similar properties should apply.

6.3.2.1 BIOLOGICAL AND PHYSICAL DATA

Survey and historical data from northern Australia that covered the NPF were used for random forests with Box-Cox power transformation model –i.e. species distribution model. Most of these data are explained in detail by Hill et al. (2002), Rochester et al. (2007) and in the Appendix 8. Physical environmental data used as correlates to the biological data were obtained from each sampling station from a compilation of seabed characteristics and ocean climatology from Rochester et al. (2007), resulting in a total of 25 variables. Physical data were interpolated, resampled, averaged and mapped to a 1 minute cells or 0.0167° resolution (i.e. ~11 km) across the GoC region between 10° - 18° S and 135° - 142° E, resulting in 230,400 grid cells (480 x 480 cells).

6.3.3 TROPHIC MASS-BALANCE DYNAMICS

The Ecopath with Ecosim and Ecospace (EwE v6) modelling tools were used herein (Christensen & Pauly 1992, Pauly et al. 2001 and Christensen & Walters 2004). We focused this model on the south-west portion of the Gulf of Carpentaria. Data used included those collected and collated in objectives 1 and 2; historical and literature-derived data, and data derived from other existing EwEv6 models developed for the north-eastern Gulf of Carpentaria (FRDC 2004/024) and for assessing Australia's northern Illegal Unregulated and Unreported (IUU) fisheries (Rothlisberg et al. 2007). These two latter

EwE models were expanded and enhanced with additional field data to better characterise the benthic food web and the overall biological community affected by the operations of the NPF. In this work we focus on a model specification aimed at describing the benthic effects of the second season of the each year, commonly known as the "tiger and endeavour" season (April to June).

Because this model was used to primarily investigate fishing effects on demersal and benthic groups, we disaggregated lower trophic level functional groups and aggregated higher trophic levels (e.g. sharks, pelagic fishes, etc) in order to focus on the ecological effects of prawn trawling in the GoC using dynamic simulations addressing the stated objectives. The model was constructed to distil data on the biomass of different functional groups, their production rates, their diets and consumption rates, fisheries and the environmental characteristics, and other information to describe the trophic flows and non-trophic attributes of the ecosystem.

6.3.3.1 THE ECOPATH WITH ECOSIM APPROACH

Ecopath trophic models describe the state of energy flows in a food web. They are designed to include all biotic components of an ecosystem, and biomass wet-weight (used here) is usually the 'currency'. The first Ecopath master equation expresses the law of conservation of mass or energy and it indicates the basic input parameters. This equation balances a group's net production (terms to the left of the equal sign) with all sources of mortality, migration, or change for that group (terms to the right). More specifically, it says that the net production of a functional group equals the sum of (1) the total mass (or energy) removed by predators and fisheries, (2) the net biomass accumulation of the group, (3) the net migration of the group's biomass, and (4) the mass flowing to detritus.

$$\mathrm{Bi} \cdot (\mathrm{P/B})\mathrm{i} \cdot \mathrm{EEi} = \mathrm{Yi} + \Sigma\ [\mathrm{Bj} \cdot (\mathrm{Q/B})\mathrm{j} \cdot \mathrm{DCji}] + \mathrm{BAi} + \mathrm{NMi}$$

Bi and Bj are biomasses of prey (i) and predators (j) respectively; P/Bi is the production/biomass ratio, equivalent to total mortality (Z) in most circumstances (Allen, 1971); EEi is the ecotrophic efficiency; the fraction of the total production of a group utilized in the system; Yi is the fisheries catch per unit area and time (i.e., Y = F*B); Q/B_j is the food consumption per unit biomass of j; DC_{ji} is the contribution of i to the diet of j; BA_i is the biomass accumulation of i (positive or negative); and NM_i is the net migration of i (emigration less immigration).

The Ecopath model is a system with as many such linear equations as there are functional groups. The energy balance (conservation of matter) within each functional group is ensured with the second master equation as follows:

$$Consumption = production + respiration + unassimilated\ food$$

The implied thermodynamic constraints of this equation underscore the power of *Ecopath* models as a focal point for refinement of ecosystem information. The need to reconcile energy production and demand among components of the food web narrows the possible ranges of parameter estimates for particular groups. Inclusion of a biomass accumulation factor and migration factor in the general *Ecopath* equation distinguishes *Ecopath* modelling as an 'energy continuity' approach rather than a strictly 'steady state' approach. Conservation of energy (*continuity*) is assumed for every identified component of the ecosystem, and the whole system. This basic constraint enables representation of changes in populations (i.e., functional groups) when expressed in dynamic form.

Ecopath was refined considerably with the dynamic simulation routines *Ecosim* and *Ecospace* (Walters et al., 1997; Walters et al., 1999; Pauly et al., 2000; Walters et al., 2000). In *Ecosim*, information in the static *Ecopath* file is re-expressed in a dynamic formulation:

$$\frac{dB_i}{dt} = g_i \sum_j Q_{ji} - \sum_j Q_{ji} + I_i - \left(M0_i + F_i + e_i\right) \cdot B_i$$

where dB_i/dt is the change in the biomass of group i (B_i) over time, g_i is the efficiency of the conversion of food into growth, Q_{ji} is the rate of consumption by predators j of prey group i, I_i is the immigration rate, $M0_i$ is the natural mortality rate, F_i is the fishing mortality rate, and e_i is the emigration rate.

The dynamics and sensitivity of Ecosim models is largely controlled by the consumption rates (Q_{ji}) which are limited by the proportion of a given predator group's prey that exist in a vulnerable state. Prey vulnerability is controlled within the expression of consumption rate by a user-specified (or calculated) transfer rate of prey movement between vulnerable and invulnerable pools (v_{ij} and v'_{ij}), thus representing the universal community stabilizer of prey refugia. The consumption rate (Qji) includes the prey vulnerability parameters:

$$Q_{ji} = \frac{v_{ij} \cdot a_{ij} \cdot B_i \cdot B_j \cdot T_i \cdot T_j \cdot S_{ij} \cdot M_{ij} / D_j}{v_{ij} + v'_{ij} \cdot T_i \cdot M_{ij} + a_{ij} \cdot M_{ij} \cdot B_j \cdot S_{ij} \cdot T_j / D_j}$$

a_{ij} is the rate of effective search for prey i by predator j, T_i and T_j are the relative feeding times of prey i and predator j, S_{ij} is the user-defined seasonal and long-term environmental forcing effects, M_{ij} is the non-trophic mediation forcing effects, and D_j represents the effects of prey handling time by predators, which further limits consumption rates. See Walters et al. (1997), Christensen and Walters (2004) and the Ecopath with Ecosim user's guide for more information. The free software can be downloaded from www.ecopath.org.

Parameters v_{ij} and v'_{ij} represent prey vulnerabilities, or the rate of exchange of biomass between two prey behavioural states: a state in which all predators have full access to prey and a state in which prey have full refuge from predators. Prey use refugia in real ecosystems. Thus, not all prey biomass is vulnerable to predation at any given time, and predator-prey relationships are limited by behavioural and physical mechanisms. Ecosim is designed so that the user can specify the type of trophic control (Lotka-Volterra type vs. donor control) that mediates any interaction in the food web. Maximum consumption rates are hypothesized, and thus the rate of exchange of biomass (v_{ij}) that a predator normally exerts. For high prey vulnerability (v_{ij}) the functional relationship approximates a mass-action flow, or Lotka-Volterra type interaction implying a strong 'top-down' effect. For low prey vulnerabilities the functional relationship approaches a donor-controlled (bottom-up) flow rate so v_{ij} is the maximum possible instantaneous mortality rate that j can cause on i (Walters et al., 1997).
Prey vulnerabilities can be specified by adjusting the proportion of prey in vulnerable and invulnerable states (pools) via adjustment of the v values, which are scaled such that pure Lotka-Volterra (top down) type control and pure donor control. In the real world, this mixture of trophic control is mediated by temporal or spatial refugia, or by the relative primacy of physical and biotic forces in regulating communities, i.e., predator-prey interactions. Derivation of v values for each functional group is discussed in detail under the section fitting Ecosim to time series data.

6.3.3.2 MODEL CONSTRUCTION PROCEDURE

The following steps were undertaken when constructing the Ecopath with Ecosim and Eco model:

1. **Define the ecosystem in space and time** – The spatial extent of the system and the represented time period must be clearly defined. Parameter estimates are expressed in annual units.
2. **Define functional groups** – Myriad species comprise interaction webs, but these species must be aggregated into related groupings that make sense in terms of ecological function, and the questions of interest.
3. **Estimate basic parameters** for each functional group.

4. **Estimate fisheries information** – Landings, discards and discard fates is derived and entered for each fishery gear type. Effort and catch time series should also be specified such that the catches and discards in the initial modelled period is expressed properly over time.
5. **Estimate additional *Ecopath* parameters** – Detritus fates, assimilation rates, multi-year trends, temporal distributions, and habitat associations.
6. **Enter parameters** into the windows-based input interfaces (see www.ecopath.org).
7. **Balance the model** according to thermodynamic constraints.
8. **Calibrate Ecosim model** – the Ecosim model is fitted to time series data in order to confidently make future predictions
9. **Run Ecosim scenarios** – "what if" scenarios are run by varying fisheries, biological or environmental parameters.

6.3.3.3 SPATIAL AND TEMPORAL EXTENT OF THE MODEL

The Gulf of Carpentaria (hereafter referred to as the GoC) is a large (370,200 km^2), shallow (<70 m) tropical marine embayment located along Australia's northern coastline between the Cape York Peninsula to the east and the Wessel Islands and Arnhem Land to the west. The Arafura Sea and New Guinea lie to the east. The colored area of Figure 2 is the area included in the present Ecopath model of the GoC.

The year 1990 was chosen to characterise a static description of the trophic flows in the GoC and the model forecasts were run for a 20-year period. The changes in fishing behaviour and fishery management policies imposed in 2008 were simulated and the model allowed forecasting the consequences of these perturbations over a 20 year period to 2028.

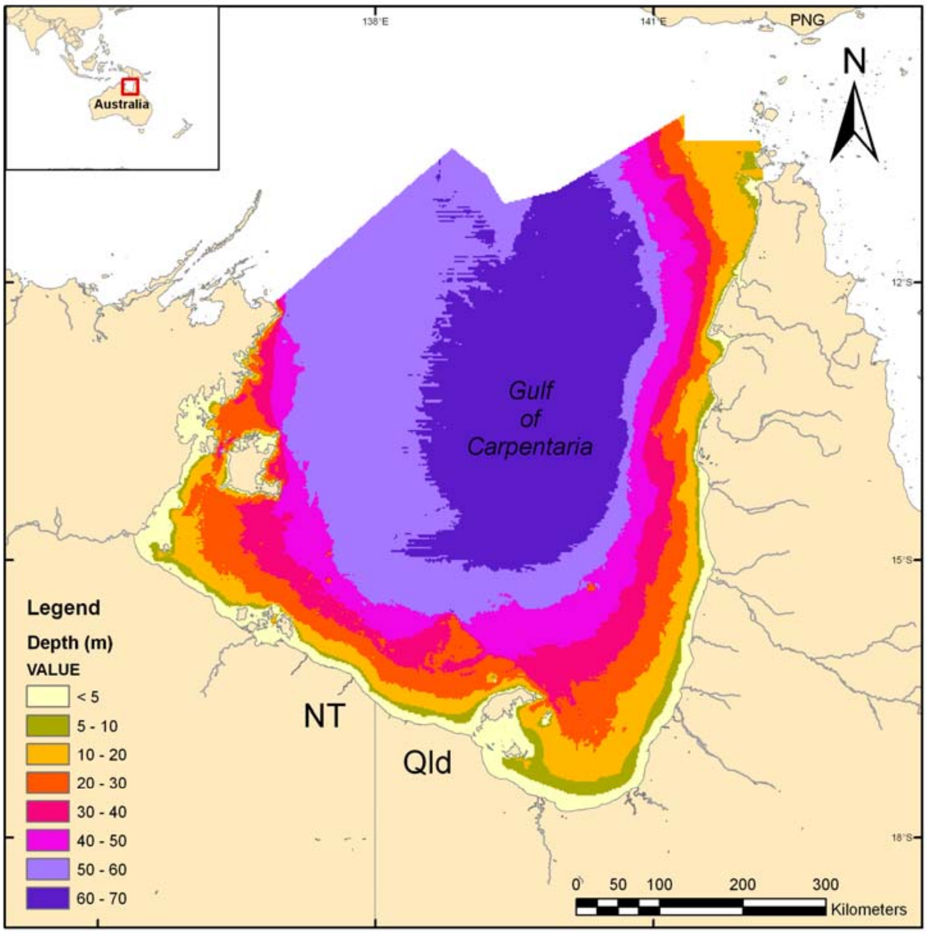


Figure 2. The area included in the present Ecopath model of the Gulf of Carpentaria, Australia. Colours represent the depth zones of the gulf.

6.3.3.4 MODEL STRUCTURE

The biota of the Gulf were aggregated into 53 functional biological groups based on the best information available (Table 6). These groups were chosen based on ecological ‘guild’ similarity criteria such as preferred habitat, feeding type and diet, sizes, and rates of production and consumption, in addition to the questions of interest in this study. Functional groups act as a single biomass pools, or species, even though they include numerous species. Thus, the aggregation of species into these functional groups may affect model dynamics in some cases. The approach taken here was to emphasise disaggregation of benthic and demersal groups in order to articulate the mechanisms and cascades that may be important in the real ecosystem, while aggregating other groups, such as pelagic fish, sharks and seabirds, that we considered not to have strong bearing on the outcomes of the simulations of interest in this project. Single-species stock models that characterise the interaction of different life stages (juvenile and adult) of functional groups are represented by separate adult and juvenile functional groups (i.e. tiger and banana prawns) and were embedded in the overall model to increase the robustness of the simulated dynamics. Aggregation was done in the case of discarded bycatch that includes all non-target taxa captured by the fisheries that is not retained and discarded overboard. This is an important non-dynamic functional group that is treated as a detrital pool that does contribute back to the food web.

Table 6. List of the 53 functional groups of the GoC benthic model and its basic input parameters.

Group name	Trophic level	Habitat area (fraction)	Biomass in habitat area (t/km2)	Z (/year)	Production / biomass (/year)	Consumption / biomass (/year)	Ecotrophic efficiency	Production / consumption
1. Dolphins	4.4	1	0.001		0.1	41.07	0.155	0.002
2. Dugongs	2.4	1	0.001		0.08	36.5	0.324	0.002
3. Turtles	3.4	1	0.008		0.192	3.5	0.138	0.055
4. Sea snakes	4.7	1	0.001		0.46	6.3	0.322	0.073
5. Sea birds	3.7	1	0.003		0.131	45.8	0.103	0.003
6. Large sharks	4.8	1	0.008		0.33	3.468	0.313	0.095
7. Small sharks	4.5	1	0.011		0.568	7.158	0.876	0.079
8. Sawfishes	4.3	1	0.003		0.29	2.66	0.506	0.109
9. Rays	3.9	1	0.163		0.565	4.561	0.195	0.124
10. Pelagic carnivores Fish	4.2	1	0.164		0.79	8.76	0.227	0.09
11. Pelagic invert feeders Fish	3	1	3.71		0.845	13.87	0.793	0.061
12. Benthopelagic carnivores Fish	4	1	0.35		1.792	10.788	0.338	0.166
13. Benthopelagic invert feeders Fish	3.4	1	0.995		2.62	10.927	0.815	0.24
14. Benthic carnivores Fish	4.1	1	0.225		1.6	7.567	0.529	0.211
15. Benthic invert feeders Fish	3.5	1	0.8		2.008	9.732	0.909	0.206
16. Red snappers Fish	4.4	1	0.132		1.505	5.667	0.522	0.266
17. Reef assoc. carnivores Fish	4.3	0.2	0.125		1.2	10.063	0.829	0.119
18. Reef assoc. invert feeders Fish	3.8	0.2	0.144		2.105	14.489	0.9	0.145
19. Reef assoc. herbivores Fish	3	0.2	0.179		1.3	31.413	0.957	0.041
20. Detritivores Fish	2	0.11	0.124		2.18	17.44	0.811	0.125
21. Cephalopods	4.1	1	0.17		3.4	17.338	0.59	0.196
22. Stomatopods	3.5	1	0.1		4.25	14.5	0.686	0.293
23 Banana prawn juv	3.2	0.24	0.004	3.42		55.636	0.182	0.061
24. Banana prawn adult	3.3	1	0.009	3.2		19.2	0.679	0.167
25. Tiger prawn juv	3.3	0.18	0.011	3.45		55.686	0.177	0.062
26. Tiger prawn adult	3.3	1	0.02	3.2		19.2	0.87	0.167
27. Other commercial prawns	3.3	1	0.18		3.2	19.2	0.605	0.167
28. Thallasinid prawns	3.1	1	0.963		4.65	19.2	0.633	0.242
29. Other non-commercial prawns	3.1	1	0.58		3.44	19.2	0.638	0.179
30. Lobsters	3	1	0.012		0.9	7.4	0.313	0.122
31. Mud crab	3	1	0.003		2.8	10.95	0.055	0.256
32. Sand crab and other large crabs	3.2	1	0.086		2.65	14.56	0.836	0.182
33. Large gastropod carnivore	2.9	1	0.08		0.72	3.755	0.74	0.192
34. Holothurians	2	1	0.055		0.6	2.077	0.3	0.289
35. Spatangoids	2	1	0.013		1.5	5	0.209	0.3
36. Echinoids	2.7	1	0.175		1.65	10.95	0.811	0.151
37. Ophiuroids	2.1	1			1.6	13.992	0.95	0.114
38. Asteriods	2.8	1	0.051		0.46	3.24	0.186	0.142
39. Sessile epibenthos	2.5	1	4.985		0.189	30.213	0.767	0.006
40. Bivalves	2	1	4.5		2.4	9.5	0.895	0.253
41. Small crustaceans	2.4	1	4.72		5.4	47	0.8	0.115
42. Annelids detrit / carn	2.5	1	4.81		4.6	15.3	0.925	0.301
43. Small gastropod omni / Small gastropod c	2.6	1	2.3		3.78	15.515	0.924	0.244
44. Infaunal detrit / carn	2.4	1	5.5		3.8	27.4	0.852	0.139
45. Zooplankton	2.1	1	12.6		17.3	173	0.807	0.1
46. Microbial heterotrophs	2	1			100	215	0.95	0.465
47. Foraminifera	2.1	1			12.5	25	0.95	0.5
48. Phytoplankton	1	1	17		118		0.848	
49. Microphytobenthos	1	1	3.241		706.496		0.069	
50. Seagrass	1	0.31	0.626		8.317		0.968	
51. Macroalgae	1	1	18		8.317		0.661	
52. Discards	1	0.75	0.123				0.465	
53. Detritus	1	1	285				0.349	

6.3.3.5 SOURCES OF BASIC BIOLOGICAL PARAMETERS AND FISHERY DATA

The year 1990, that characterised the static description of the trophic flows in the GoC, was chosen to provide a ‘convenient starting point’ for the dynamic simulations using the Ecosim model since good quality diet and fishery catch data were available for that time. However, in 1990 the shark, mackerel and some other fish stocks of northern Australia may have been recovering from over-exploitation by the Taiwan-Australia joint venture activities, Taiwanese and then Thai trawling would have had some impact at that time as well, but most of those activities were outside the GoC so it is difficult to ascertain what effects might have manifested in the GoC itself. Furthermore, other international fishing undoubtedly occurred during the 1970s and before because international waters occurred beyond 12 nautical miles until the late 1970s. It is best to choose a starting point during which major change is not occurring. Notwithstanding such complications, simulations should still indicate the system tendencies of the effects of examined changes.

The key biological parameters (biomass, productivity, diet composition, etc) for each species or taxon (Functional Groups) in the model were estimated from primary research data, fishery data, or literature (Table 6). The model features a fish diet composition matrix based on stomach content analyses from a number of previous CSIRO projects (Salini *et al.*, 1990; Brewer *et al.*, 1991; Brewer *et al.*, 1995; Haywood *et al.*, 1998; Salini *et al.*, 1998; Griffiths *et al.*, 2007; Griffiths *et al.*, In Press), but also from the extensive dietary work undertaken in the present project (Dell et al., Appendix 6; Tonks et al., Appendix 7), which has been reported in previous milestone reports.

However, owing to the high diversity of the GoC fish assemblage and their generally low commercial value in Australia, region-specific information on the basic biology of many of these fish species is scant. Maximum recorded length was often the only biological parameter available for most species and so we resorted to using the simplest empirical equations. Where the maximum size of a fish from a particular species was recorded in scientific surveys to be within 10% of the species' maximum recorded length we assumed this to be a reasonable proxy of L_∞ in the study region. We then used the empirical equation of Pauly (1980) to estimate natural mortality (M). Since the vast majority of species in the region are not fished commercially or recreationally, we assumed that total mortality (Z) was equal to M, though it probably underestimates total mortality and thus production/biomass (P/B) estimates for some species due to bycatch mortality. If these P/B estimates were used to represent an entire functional group, this would probably tend to make the model more conservative, or less responsive, to strong top-down control from apex predators or fishing impacts.

We resisted the use of biological parameters from other systems outside Australia, as much as we could, since the GoC is a unique system that likely functions very differently from most other systems in various ways, being a large shallow tropical soft sediment gulf of the southern hemisphere with unique monsoonal patterns and a somewhat unique fauna.

For fish species having detailed biological information we used M or Z estimates provided in published studies from the region. Where only growth parameters were available, we estimated M as:

$$M = 1.60K \qquad \text{(as of Jensen, 1996),}$$

where K is the von Bertalanffy growth parameter. For some elasmobranch species where maximum age was known, we calculated M by:

$$M = -\ln(0.01)/\omega \qquad \text{(as of Hoenig, 1983),}$$

where ω is longevity in years. For fish species we knew, or suspected, were fished either as a target or caught incidentally in reasonable numbers as bycatch but fishing mortality (F) was not available, it was assumed – for lack of a better assumption – that the population would be fished at MSY. A precautionary proxy of fishing mortality at MSY can be assumed to be $F = 0.8M$ (Gabriel and Mace, 1999) or $Z = 1.8M$.

For each functional group, we found great variance in the estimated P/B values. To obtain the single value for a particular group, we excluded outlying values and any values we did not have confidence in based on the quality of biological parameters (e.g. VBGF parameters or the mortality estimation method). For example, several Lutjanid species are represented in the GoC ecosystem and can vary markedly in size. Although we had reliable VBGF parameters for some species (e.g. Newman et al., 1996; Newman, 2002; Marriott et al., 2007), VBGF parameters or maximum age estimates were only available from regions in the Indian Ocean, where fish attain significantly smaller or larger maximum lengths than in northern Australian waters. Using these estimates would bias M estimates, and thus P/B. Therefore, we took the average of the most reliable values, and each weighted by the contribution of a particular species to the functional group in terms of biomass. The species contributing the most to the biomass were generally common species, and generally had some reliable biological information available from local or international studies. Therefore, the overall P/B was generally more representative of these common species, and so the potentially underestimated P/B values from data-deficient species that contributed little to the overall biomass of a functional group make an equally small contribution to the group's overall P/B value.

An estimate of Q/B was made for each fish species by using the empirical equation of Palomares and Pauly (1998):

$$\log Q/B = 7.964 - 0.204 \log W_\infty - 1.965T' + 0.083A + 0.532h + 0.398d$$

where W_∞ is weight (in grams) at age infinity, annual mean water temperature $T' = 1000/(°C+273)$, A is the aspect ratio of the caudal fin = h^2/s, h=1 and d=0 for herbivores, h=0 and d=1 for detritivores, and h=0 and d=0 for carnivores.

For fish species having little biological data, we used maximum length recorded in surveys (assumed L_∞) to estimate W_∞ using a length-weight relationship either from the literature if available, or from about 30 years of CSIRO unpublished survey data. To provide an overall Q/B value for each functional group we averaged values using the method used for P/B values. Due to a lack of information of the movement dynamics of species in northern Australia, we assumed that there was no net movement of any functional group in or out of the model, that is, that immigration equalled emigration. Therefore, all mortality and prey consumption occurred within the system.

6.3.3.6 SOURCES OF FISHERIES INFORMATION

The fisheries included in the model were: the Illegal Foreign Fishery (IFF), Northern Prawn Fishery (banana and tiger prawn fisheries separated); the Queensland (Qld) and Northern Territory (NT) line (primarily trolling for Spanish mackerel, but some demersal longline and dropline), gillnet (primarily sharks and grey mackerel) and pot (mud crab) fisheries; developmental fish trawl fishery (primarily red snappers); the indigenous fishery; and the recreational fishery. Annual catch and effort data for each of the primary state and Commonwealth fisheries that operate in the GoC were obtained from logbooks, effort for the IFF was derived from aerial surveillance surveys (Salini et al., 2007), and from scientific surveys in the case of the indigenous fishery (Coleman et al., 2003). The total biomass of each species caught in each fishery in 1990 was summed at the functional group level, and expressed as a biomass (t km^{-2}) within the modelled area.

Assigning fishery catches and bycatch to functional groups was difficult for some fisheries. Fishers are not required to record their catches at the species level in logbooks and are often recorded as broad taxonomic groups. This was a particular problem for the main shark fisheries of interest, namely the gillnet and line fisheries, since all shark catches in logbooks were recorded as "shark – unspecified", which encompasses several functional groups. In order to break down the unspecified catch to its constituent species for these two fisheries, we assumed that the relative contribution of each species to the catch recorded in logbooks was the same as the Qld and NT gillnet and line catch documented by Salini et al. (2006).

Complete catch data for the fish trawl fishery in the GoC (which comprises the state fisheries of NT and Qld) was not available, due to confidentiality of catch data in fisheries/areas where less than five boats operate. We were able to obtain permission to use catch data from the NT fish trawl fishery, but not for the Qld fishery. Since the same operators work in both fisheries and there is no spatial difference in the trawl bycatch near reef areas in the GoC (Stobutzki et al., 2003), we scaled up the catch rate of each species (in terms of kg per boat day) of the NT catch to the total effort in the Qld fishery. The catch estimates from the NT and QLD fisheries were then combined to represent the entire fish trawl fishery in the GoC.

The catch composition and biomass of each species caught in the illegal foreign fishery in northern Australia during 1990 was unknown. Because the illegal foreign fishery uses similar gear as the domestic gillnet and longline fisheries that target sharks, it was assumed that the species composition of the domestic catch would probably be representative of the foreign catch. The catch composition was therefore calculated using data collected by scientific observers from commercial gillnet and longline catches in the GoC and reported by Salini et al. (2006). Species-specific catch biomass from the illegal foreign component was estimated by using the catch in the Australian domestic gillnet in the GoC in 1990. The illegal foreign catch was assumed to be 10% of the Australian domestic catch in 1990, a period when the illegal foreign effort was believed to be significantly less than present day.

Estimates of discards from the banana prawn and tiger prawn trawl fisheries was available and compiled by Dell et al. (2009) for the construction of this model. The species composition and biomass of the discarded catch was not available for the gillnet, line, pot, fish trawl, and illegal foreign fisheries. It is well accepted that there is negligible discarding from the pot and line fisheries, so in the model we assumed there was no discarding. Scientific observers routinely collect quantitative catch data for the gillnet and fish trawl fisheries, however, this was not available due to confidentiality policies of state fisheries agencies. Since the bycatch for both fisheries can be significant, it was important to include even crude bycatch biomass estimates. Despite the fish trawl fishery utilising nets with a slightly larger mesh size than nets in the NPF, confidential data (QDPI&F observer data) show that the bycatch composition is remarkably similar to the NPF, for which bycatch data are voluminous. Therefore, we scaled the NPF bycatch biomass, in terms of total effort in the fishery, headrope length and average trawl time, to reflect the fish trawl fishery. Species composition and biomass of the discarded catch in the gillnet fishery was based on data collected by CSIRO scientists during two fishery dependent surveys in February-March 2005. These data were then scaled up to the total number of days fished in the fishery in 1990. These data were also considered to be representative of the discards from the illegal foreign fishery due to the similarity in fishing methods. We scaled the discarded catch biomass to be 10% of the domestic gillnet fishery, as was undertaken for estimating the IFF retained catch.

6.3.3.7 FITTING THE ECOSIM MODEL TO CPUE DATA

The fitting procedure in Ecosim version 5.1 was used to calibrate this 1990 Gulf of Carpentaria Ecopath model to time series of biomass (t), fishing mortality (F), catch (t) data for adult tiger prawns derived from a recent stock assessment for 1970-2006 (Dichmont et al., In Press) and nominal effort data (number of boat days per year) for ten fisheries for the period 1990-2006 in order to increase the reliability of predictions from the Ecosim scenarios. Annual biomass estimates for adult tiger prawns were scaled to the value in the first year of the time series, effectively reducing the dataset to standardised biomass. Annual adult tiger prawn catches were divided by the model region and expressed as t km^{-2} to be consistent with other Ecopath model biomass inputs. Although a stock assessment exists for banana prawns in northern Australia (Zhou et al., 2008), there were concerns that predicted biomass from the models were highly uncertain, primarily due to difficulty in correcting for changes in fishing power and recent changes in targeting practices. Therefore, only nominal effort and catch were used for this group in the Ecosim model calibration process as well as effort data for state-regulated fisheries (line, gillnet, pot, recreational, charter and fish trawl), aboriginal fisheries and IUU fisheries.

Data were imported into Ecosim and the 15 most sensitive interactions between predators and prey were identified using a non-linear search procedure. The prey vulnerability rate (v) for the most sensitive predator-prey interactions were then iteratively adjusted (from the default of 2) by 1% until the sums of squares error (SS) was minimised to produce the best model fit to the time series data. In searching for the best combination of vulnerability values, the time series data were linked to an estimated primary productivity trend, in the form of a forcing function, forced upon the “phytoplankton” group, which best improved the Ecosim model fit to the time series. A variance value of 50 was used for the model fitting, implying that the model should capture large 'spikes' in the biomass and catch data. However, such high variance values can result in model 'over-fitting', meaning that the model may simply be fitting to noise in the data rather than true trends. Hence, the model was restricted to fit only seven parameters (one for each functional group) using 10 spline points in the primary productivity forcing function, which reduced risk of model over-fitting.

Once the optimal vulnerability search was complete, vulnerability values were inspected to ensure they were ecologically realistic and adjusted slightly manually if required. Although some adjustments to vulnerabilities and feeding time parameters often resulted in better visual fits of the model to observed data, they often resulted in a poorer statistical fit (lower SS) or required unrealistic parameter values, hence they were disregarded. Other sources provide further detailed descriptions of fitting Ecosim models to time-series data (Christensen and Walters, 2004; Christensen et al., 2005).

6.3.3.8 DEVELOPING ECOSPACE

Ecospace is the spatial and temporal module of the Ecopath with Ecosim software package (www.ecopath.org; Christensen et al., 2005). Ecospace is a dynamic, spatial model that incorporates all key elements of Ecosim (including different vulnerabilities and split pools presented above). The Ecospace model is structured on biomass pools, linked by trophic relationships (i.e. predator-prey), which migrate among the grids of cells of the marine park map. Movements of functional groups are driven by parameters such as foraging behaviour, avoidance of predation, and dispersal rates that are linked to a range of defined habitats preferred by each functional group. Robust default estimation for these parameters based on life histories is built into Ecospace (Walters et al., 1999; Christensen and Walters, 2004).

In Ecospace (Walters *et al.* 1999) models the feeding interactions of functional groups in a spatially explicit way. A simple grid represents the study area, and it is divided into a number of habitat types. Each functional group is allocated to its appropriate habitat(s), where it must find enough food to eat, grow and reproduce - while providing energy to its predators and to fisheries. Each cell hosts its own Ecosim simulation and cells are linked through symmetrical biomass flux in four directions; the rate of transfer is affected by habitat quality. Optimal and sub-optimal habitat can be distinguished using various parameters such as the availability of food, vulnerability to predation and immigration/emigration rate. By delimiting an area as a protected zone, and by defining which gear types are allowed to fish there and when, we can explore the effects of marine protected areas (MPAs) and test hypotheses regarding ecological function and the effect of fisheries. Previous authors have used Ecospace in this capacity (e.g., Walters *et al.*, 1999; Pitcher et al., 2005).

The biomass of functional groups is initially distributed over the modeled region and biomasses and fluxes among cells are governed by dispersal rates related to food availability and predation rates in the cells (Christensen *et al*, 2005). Then, the ecosystem is divided into a 2-dimensional grid of cells and each cell is defined according to their habitat type and the preferences of each functional group. In the case of fishing pressure and landings, Ecospace incorporates the time series of effort per fleet per year that were defined in the Ecopath model. It should be noted that these fleets can be varied independently. The spatial distribution of this fishing effort is then controlled by a 'gravity' model, which allocates effort to each cell proportional to the relative profitability of fishing in each cell.

The Gulf of Carpentaria model was represented by a grid of 6,400 cells (80 x 80 cells). The Ecospace habitat base map (Figure 3) was built based on the broad definition of "habitats" –i.e. for this model, habitats were spatially defined areas that contain a major biological, fishery, or geomorphic benthic feature that is directly related to the NPF. The major fishing grounds identified in the gulf were: (1) tiger/endeavour Groote North; (2) tiger/endeavour Groote South; (3) tiger/endeavour Vanderlins East; (4) tiger/endeavour: Mornington north; (5) tiger/endeavour: Mornington east; (6)) tiger/endeavour: Weipa; (7) Banana; while the biological habitats were the (8) untrawlable grounds (hard/reefs); (9) seagrass; (10) inshore and shallow waters; (11) offshore and deep waters. All these habitats and their percentage of cover in the Ecospace map are presented in Table 7.

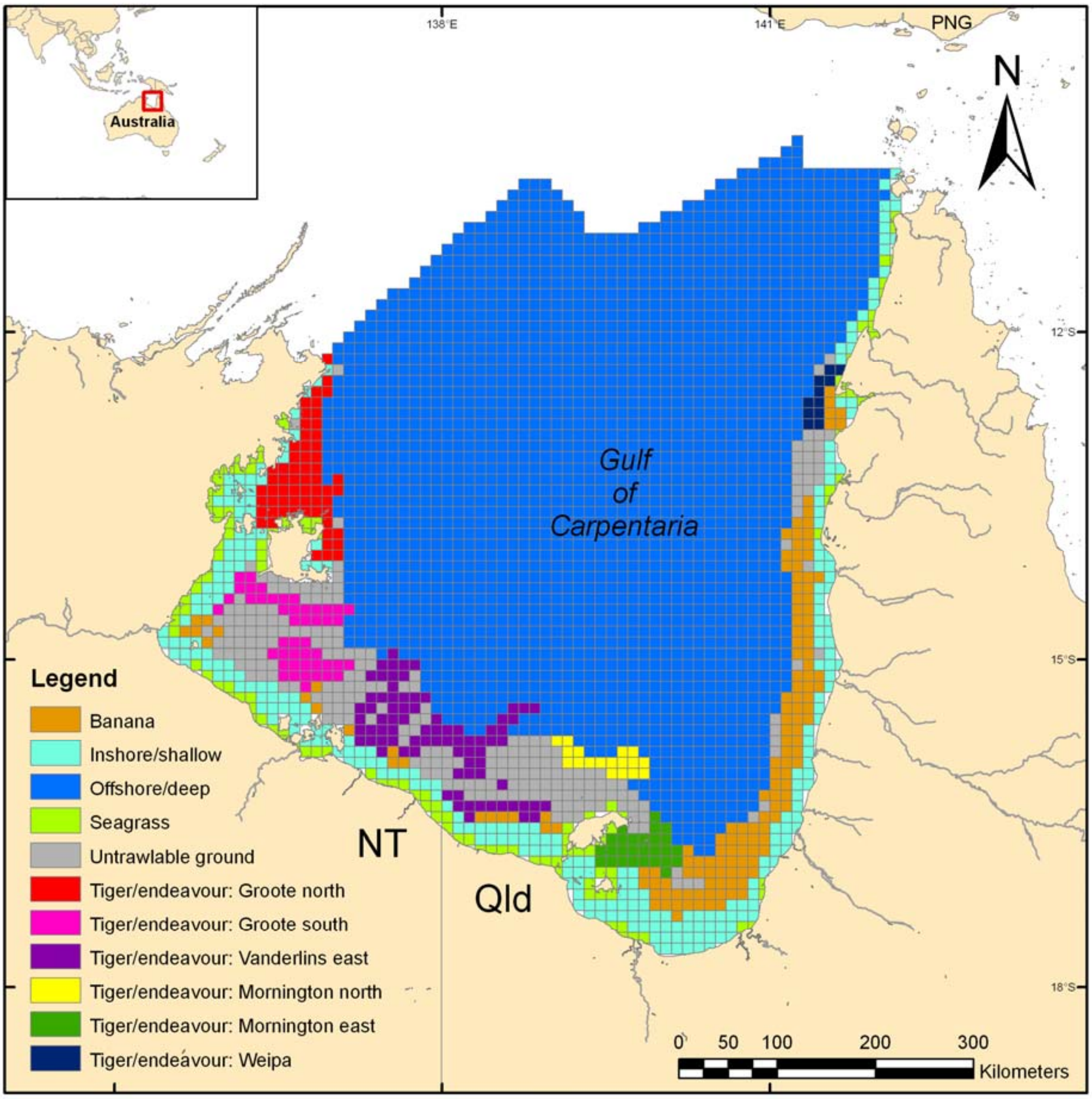


Figure 3. Ecospace basemap of the Gulf of Carpentaria showing the major habitats considered in the model (for descriptions see Table 7).

Table 7. Descriptions of each habitat type considered and defined in the GoC Ecospace model.

#	Type	Habitat Name	Abbreviation	Description
1	Fisheries ground	Tiger/endeavour Groote north	grtn	Northern Groote green-mud hot spot
2	Fisheries ground	Tiger/endeavour Groote south	grts	Southern Groote hot spot
3	Fisheries ground	Tiger/endeavour: Vanderlins east	vane	South-central harder-grounds hotspots
4	Fisheries ground	Tiger/endeavour: Mornington north	mtnn	North-Mornington hotspot (sponges & heart urchins grounds)
5	Fisheries ground	Tiger/endeavour: Monington east	mtne	South-east riverine-tidal sedimentary basin
6	Fisheries ground	Tiger/endeavour: Weipa	weipa	Offshore NE Weipa
7	Fisheries ground	Banana	banana	East-south (including inshore Weipa) banana hotspots
8	Biological	Untrawlable ground	utg	Rough grounds Submerged coral reefs
9	Biological	Seagrass	sg	Shore and coastal/tidal seagrass beds
10	Geomophic	Inshore/shallow	inshore	Depth < 15m, shallow and backreef/utg
11	Geomorphic	Off-shore/Deep	offshore	Depths ~> 45m. oceanic-like but muddy largely off-trawling grounds
12	Geomorphic	Offshore-2 (Boundary of the Model)	Offshore2	North-west Arafura out of model area

6.3.3.9 HABITAT ASSIGNMENT

Once habitats were defined, the functional groups defined in the Ecopath model were assigned to their ‘preferred’ habitat. ‘Preferred’ here means that the group in question has higher feeding rates in the habitat and its survival rate is also higher there (because the predation rate is higher in non-preferred habitat). The habitat assignment was made by experts within CSIRO and based on data from surveys carried out in the waters of the GoC. The habitat assignment and base dispersal rates of the functional groups considered in the model are presented in detail in Table 8.

6.3.3.10 SPATIAL REPRESENTATION OF FISHERIES

In the case of the spatial representation of fishing in Ecospace, the model uses the multiple fishing fleets and their fishing mortality rates (F) included in the Ecopath model. In Ecospace the F’s are distributed using a simple ‘gravity model’, where the proportion of the total fishing effort allocated to each cell is assumed to be proportional to the sum over groups of the product of the biomass. When included within the wider MSE framework, the gravity model for the tiger prawn fleet is replaced by the effort from the fine-scale fleet dynamics model (section 6.3.5.2). For the Spatial MSE (section.6.4.1 below) we used the fishing effort series derived form the bioeconomic stock assessment model (see section 6.3.5). The catchability and profitability of fishing the target groups are also considered and included (Christensen et al., 2005). The Ecospace model considered the 14 fishing fleets included in the Ecopath model and their preliminary spatial distribution within the spatial model (Table 8).

Table 8. Spatial allocation by each habitat of the various functional groups included in the Ecospace model of the Gulf of Carpentaria (grtn= Groote north, grts= Groote south, vane= Vanderlins east, mtnn= Mornington north, mtne= Mornington east, weipa= Weipa, banana= Banana east, utg= Untrawlable grounds, sg= Seagrass, inshore= Inshore/shallow, offshore= Offshore/deep). It also include the values of the annual dispersal rates used for each functional group.

Group \ Habitat #	grtn	grts	vane	mtnn	mtne	weipa	banana	utg	sg	inshore	offshore	Dispersal rate
Percentage area	**1.9%**	**1.5%**	**2.5%**	**0.6%**	**0.9%**	**0.3%**	**4.7%**	**7.2%**	**4.4%**	**8.7%**	**67.3%**	**(km/year)**
1. Dolphins	5.0%	5.0%	5.0%	5.0%	5.0%	5.0%	10.0%	10.0%	0.0%	10.0%	40.0%	300
2. Dugongs	5.0%	5.0%	5.0%	5.0%	5.0%	5.0%	5.0%	5.0%	50.0%	15.0%	5.0%	50
3. Turtles	2.6%	1.2%	1.4%	0.0%	1.0%	0.4%	11.1%	9.0%	15.2%	32.5%	25.5%	300
4. Sea snakes	2.2%	3.9%	5.0%	2.2%	4.7%	2.0%	5.0%	33.0%	10.0%	22.0%	10.0%	100
5. Sea birds	3.0%	3.0%	3.0%	3.0%	3.0%	5.0%	10.0%	5.0%	0.0%	5.0%	60.0%	300
6. Large sharks	1.7%	1.2%	1.9%	0.5%	0.7%	0.2%	6.2%	7.7%	4.2%	11.2%	64.6%	300
7. Small sharks	3.2%	1.7%	2.3%	0.4%	0.4%	0.5%	4.0%	6.5%	5.0%	11.1%	64.9%	300
8. Sawfishes	3.9%	3.1%	15.0%	0.4%	0.3%	0.2%	6.8%	19.3%	12.6%	24.3%	14.0%	100
9. Rays	2.7%	2.2%	6.5%	0.6%	0.2%	0.4%	3.1%	7.8%	1.6%	5.1%	69.7%	100
10. Pelagic carnivores Fish	2.1%	1.6%	2.9%	0.5%	2.2%	0.3%	6.7%	9.9%	3.2%	10.4%	60.3%	100
11. Pelagic invert feeders Fish	3.3%	1.2%	1.6%	0.2%	1.4%	0.4%	6.4%	8.8%	6.2%	14.2%	56.4%	50
12. Benthopelagic carnivores Fish	3.4%	2.2%	2.9%	0.5%	0.5%	0.6%	6.5%	7.9%	3.1%	10.6%	61.9%	100
13. Benthopelagic invert feeders Fish	2.9%	1.4%	2.8%	0.2%	0.3%	1.1%	7.3%	9.3%	4.9%	15.3%	54.5%	50
14. Benthic carnivores Fish	1.6%	1.0%	1.8%	0.6%	0.3%	0.3%	2.2%	4.5%	1.6%	4.6%	81.5%	150
15. Benthic invert feeders Fish	1.3%	0.9%	1.3%	0.3%	0.6%	0.3%	10.7%	6.8%	6.7%	21.1%	50.1%	50
16. Red snappers Fish	0.9%	0.4%	2.0%	0.3%	0.4%	0.1%	3.5%	9.5%	2.1%	6.1%	74.7%	100
17. Reef assoc. carnivores Fish	4.1%	1.5%	2.0%	0.5%	0.3%	0.6%	4.9%	8.6%	3.7%	10.2%	63.6%	100
18. Reef assoc. invert feeders Fish	2.6%	1.1%	1.8%	0.4%	0.1%	0.4%	2.0%	4.5%	1.1%	3.7%	82.2%	50
19. Reef assoc. herbivores Fish	1.2%	0.7%	2.2%	0.3%	0.4%	0.3%	12.3%	11.3%	4.0%	16.5%	51.1%	50
20. Detritivores Fish	6.4%	0.5%	1.0%	0.0%	0.5%	7.2%	15.6%	9.1%	10.9%	31.0%	17.7%	20
21. Cephalopods	1.3%	0.7%	2.0%	0.4%	0.8%	0.2%	4.1%	7.6%	3.8%	11.8%	67.3%	30
22. Stomatopods	10.0%	5.0%	5.0%	5.0%	5.0%	2.0%	10.0%	8.6%	5.6%	17.0%	26.8%	20
23 Banana prawn juv	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	1.0%	1.0%	50.0%	48.0%	0.0%	1
24. Banana prawn adult	3.8%	0.9%	0.4%	0.0%	0.4%	0.7%	65.0%	3.0%	11.0%	10.0%	4.7%	1
25. Tiger prawn juv	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	5.0%	65.0%	30.0%	0.0%	10
26. Tiger prawn adult	12.0%	10.0%	11.4%	12.0%	12.0%	1.0%	3.0%	15.5%	6.1%	5.0%	12.0%	20
27. Other commercial prawns	12.0%	10.0%	11.0%	12.0%	12.0%	1.0%	5.0%	15.0%	5.0%	7.0%	10.0%	20
28. Thallasinid prawns	0.9%	0.4%	2.3%	0.4%	1.4%	0.4%	9.7%	5.3%	4.8%	15.8%	58.7%	20
29. Other non-commercial prawns	4.6%	2.7%	6.7%	9.4%	0.7%	0.7%	4.7%	11.7%	2.7%	8.5%	47.7%	20
30. Lobsters	1.4%	1.0%	10.0%	1.6%	1.4%	0.4%	6.3%	35.0%	3.0%	10.0%	30.0%	10
31. Mud crab	5.0%	4.0%	5.0%	10.0%	6.0%	1.0%	15.0%	10.0%	7.0%	15.0%	22.0%	10
32. Sand crab and other large crabs	4.1%	1.7%	5.6%	2.4%	1.1%	0.9%	8.5%	11.0%	3.7%	12.7%	48.2%	1
33. Large gastropod carnivore	1.3%	1.0%	2.0%	0.6%	1.4%	0.4%	7.2%	8.7%	3.1%	10.5%	63.8%	10
34. Holothurians	3.8%	3.1%	3.6%	1.1%	2.8%	0.1%	4.6%	12.8%	8.0%	19.0%	41.1%	1
35. Spatangoids	1.5%	6.0%	8.2%	0.4%	0.3%	0.0%	2.4%	12.9%	5.5%	12.1%	50.6%	2
36. Echinoids	2.4%	5.0%	5.4%	5.8%	0.4%	0.1%	2.8%	12.2%	3.6%	9.4%	52.9%	2
37. Ophiuroids	1.4%	0.4%	0.7%	0.2%	3.0%	0.3%	9.7%	6.3%	4.5%	14.0%	59.6%	2
38. Asteriods	5.0%	5.0%	5.0%	5.0%	5.0%	2.0%	10.0%	7.0%	3.0%	5.0%	48.0%	2
39. Sessile epibenthos	0.5%	0.8%	0.5%	8.0%	0.7%	0.5%	3.0%	65.0%	3.0%	8.0%	10.0%	0
40. Bivalves	4.6%	3.0%	2.8%	1.0%	0.7%	0.3%	6.4%	14.1%	11.5%	24.7%	31.1%	2
41. Small crustaceans	5.0%	5.0%	5.0%	5.0%	5.0%	3.0%	12.0%	20.0%	5.0%	15.0%	20.0%	1
42. Annelids detrit / carn	2.4%	2.3%	4.1%	1.1%	0.9%	0.2%	3.8%	9.3%	2.1%	6.8%	66.8%	2
43. Small gastropod omni / Small gastropod carn	5.0%	5.0%	8.0%	2.0%	2.0%	1.0%	9.0%	11.2%	3.8%	12.0%	41.0%	2
44. Infaunal detrit / carn	8.8%	5.0%	5.0%	2.0%	1.9%	1.0%	10.4%	6.5%	2.6%	11.0%	45.8%	2
45. Zooplankton	2.1%	1.8%	2.6%	2.8%	0.7%	1.6%	17.1%	11.5%	6.2%	27.1%	26.5%	10
46. Microbial heterotrophs	1.9%	1.5%	2.5%	0.6%	0.9%	0.3%	4.7%	7.2%	4.4%	8.7%	67.3%	50
47. Foraminifera	1.9%	1.5%	2.5%	0.6%	0.9%	0.3%	4.7%	7.2%	4.4%	8.7%	67.3%	50
48. Phytoplankton	1.9%	1.5%	2.5%	0.6%	0.9%	0.3%	4.7%	7.2%	4.4%	8.7%	67.3%	10
49. Microphytobenthos	1.9%	1.5%	2.5%	0.6%	0.9%	0.3%	4.7%	7.2%	4.4%	8.7%	67.3%	5
50. Seagrass	1.0%	1.0%	2.0%	1.0%	1.0%	1.0%	2.0%	2.0%	75.0%	14.0%	0.0%	0
51. Macroalgae	0.8%	0.2%	0.4%	0.1%	1.1%	0.3%	6.8%	22.1%	9.4%	30.7%	28.0%	0

6.3.3.11 DISPERSAL RATES

Each of the groups and species considered in the Ecopath model have an aggregated biomass (B_i) and they are not assumed to move within the modeled area of the GoC (Figure 3). In Ecospace, a fraction of the biomass (B') of each cell is always on the move, according to

$$B' = m \cdot Bi$$

With *m* having the dimension of length/ time (i.e. km/year) i.e., a velocity or ‘speed’. However, *m* is not a rate of directional migration, as occurs seasonally in numerous fish populations. Rather, *m* should be regarded as dispersal and seen as the rate (km year^{-1}) at which the organism would disperse from a given ecosystem as a result of random movements (Christensen et al., 2005). As for the absolute value of *m* to be used in the simulation we used a default value of 300 km year^{-1} (recommended by Christensen *et al.,* 2005) for all groups with high/medium motion activity (fish groups) and a default value of 3 km year^{-1} (Christensen *et al.,* 2005) for those groups with very low motion (sessile groups and non-living groups). The dispersal rates for the groups considered in the model are presented in Table 8. The movement rates in the model were revised by experts from CSIRO. No migration or advection of functional group was considered in this model.

6.3.4 ECOLOGICAL RISK ASSESSMENT (ERA)

In this section we adapt and modify the current NPF model used for the ERA process (Griffiths et al. 2007, AFMA 2008, 2009), using primarily the stand-alone model developed by Zhou et al. (2007, 2009) in combination with the species distribution model (Browne et al. unpubl ms) described above in section 6.3.2. We used the ERA model to define spatial closure scenarios that could protect taxa deemed at risk. The species distribution model was then used to determine statistic estimates of the species biomass density and their spatial distributions based on the biophysical relation with various environmental variables affecting most of the species and taxa caught by the fishery –i.e. data sources include commercial data, scientific by-catch data from the various surveys and pre-fishing season surveys. The ERA methods have already been applied to stand-alone assessments of the species, which implies that the fishing effort implemented into the target tiger prawn assessment model (Dichmont et al 2008) is used to dynamically estimate ERA risk of various species, taxa and functional groups (Milton et al. 2008a, Zhou et al. 2008, Zhou et al. 2009., AFMA 2007, 2008, 2009).

6.3.4.1 DATA

Fishing effort time series from commercial catch/effort data was extracted from the NPF's logbook data that include the history and target species assessment results for each season/year since 1970. The static biomass density and spatial distribution data for each functional group (FG) used in the GoC benthic EwE model were determined using a statistical biophysical species distribution model (Browne et al. unpubl. Ms). The Ecosim module of GoC EwE model was then used to forward predict the biomass estimates up the projected year of 2015.

6.3.4.2 METHOD

A 5-year sliding window was selected to average effort level (5 year backward to the assessed year) for all fished areas, delineating the fishing areas that overlap with that of individual species (or FGs) spatial distribution as:

$$U = q*(1-S) * (\sum^{n}_{i=1}(L_i*W)/(n*A_i) \quad \text{(Zhou et al, 2008)}$$

where **q** is catchability; **S** is escape rate; **n** is number of years; **Li** is yearly fishing trawled length; **W** is fishing trawl width. Modifying,

$$U = q*(1-S) *(\sum^{n}_{i=1}(\sum^{m}_{g}(a*E_{i,g}*d_g))/(n*\sum^{M}_{g}A_{g}*d_g)$$

If d_g = static (species distribution prediction) FG biomass density data, otherwise d_g = dynamic (EwE) projected FG density data:
where **a** is the mean swept area per boat day; $\mathbf{E_{i,g}}$ is annual effort for specific 6nm grid; **m** is number of fished grids of the year; $\mathbf{d_g}$ is species density of the grid; **M** is total number of grids of species distribution within considered habitat; $\mathbf{A_g}$ is area of each 6NM grid, which is a constant. The parameters **q** and **S** are available from Zhou et al (2008). For historical years, **m** can be derived by overlapping logbook data on static species distribution data. For project years, **m** will be deducted by MSE component – effective trawling model which allocated project annual efforts to the 6NM grids. $\mathbf{d_g}$ is the element of new projected dynamic (EwE) Functional Groups distribution data.

We then calculated the following reference points (from the Table for the functional groups based on individual species analyses in Zhou, 2007) to compare them with calculated fishing impact, U, in order to assess the status of ERA risks:

(1) *umsm*--fishing mortality rates corresponding to the maximum sustainable fishing mortality (MSM) at *Bmsm* (biomass that supports MSM, which is equivalent to MSY for target species);

(2) *ucrash*--minimum unsustainable fishing mortality rate that, in theory, may lead to population extinction in the long term. For convenience, we labelled the risk categories as follows:

- Low risk: fishing mortality rate *u* is less than *umsm*;
- Medium-High risk: fishing mortality rate is greater than *umsm* but less than *Ucrash*;
- Extreme high risk: fishing mortality rate is greater than *ucrash*.

The ERA model is coded with R language and embedded within the MSE framework. It interrogates a database where annual logbook data are stored; furthermore the database is continually updated as new information is received (better distribution protection, fishing effort, betters estimates of parameters, etc.).

6.3.5 PRAWN BIOECONOMICS

The technical descriptions of the operating prawn bieconomic models have been extensively described in various previous NPF-centric projects (e.g. FRDC 2004/022) and their resulting peer-reviewed publications (e.g. Wang et al. 1999, Dichmont et al. 2006a, b, c, Venables et al 2009, Dichmont et al. 2008, Kompas et al. 2009, Dichmont et al. 2010). For the purpose of this report, we have taken and modified these existing texts and adapted them to fit their use for this project's objectives. The prawn bioeconomics assessment consists of several parts: (i) a biological population dynamic prawn model, (ii) an effort allocation model, and (iii) the definition and evaluation of both biological and economic management strategies.

6.3.5.1 PRAWN BIOLOGICAL MODEL

The population dynamics of prawns was modelled in the operating model in the same way as in Dichmont et al. (2006a), although unlike Dichmont et al. (2006a) two species of endeavour prawns (*Metapenaeus endeavouri* and *M. ensis*) were taken into account as well as the two species of tiger prawns (*Penaeus esculentus* and *P. semisulcatus*). All four species are represented by multiple stocks. Although hypotheses exist which suggest that there may be up to seven stocks of each species in the NPF, this model focuses on the Tiger-Endeavour season 2 of the NPF (from early April to mid June of each year) and is centred in the proposed stock regions contained within and around the Gulf of Carpentaria (Figure 4). All four prawn species occur in the modelled area and much of the historical and current fishing effort is exerted during the tiger-endeavour season.

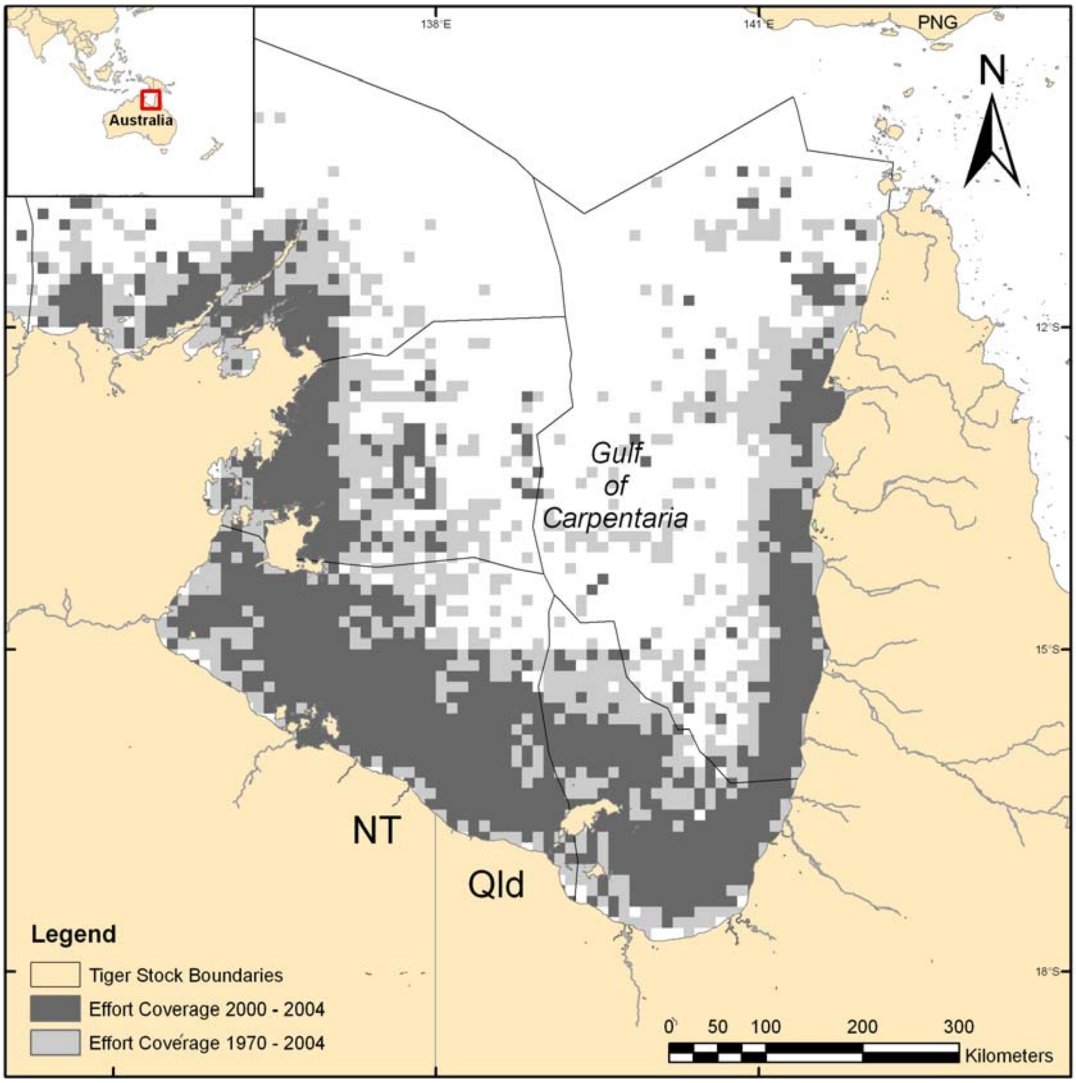


Figure 4. Map of northern Australia showing the seven regions and the 6-minute grids used in the prawn and benthic impact model. The three westernmost regions are combined in the operating model.

The population dynamics of each species in each region that they occur in are governed by a delay-difference model that operates on a weekly time step. The conditioning of the operating model involved estimating annual recruitments from data on catches and standardised catch-rates and using these to estimate the parameters of a Ricker stock-recruitment relationship (c.f. Dichmont et al., 2003). The values for natural mortality, growth, and weekly availability, recruitment and spawning proportion were based on tagging data, on analyses of experimental data, and on analyses of survey data.

Targeted effort for one species in a given region leads to catches of all species that are found in that region (i.e. technical interactions are included in the operating model). Unlike tiger prawns, there is no directed (target) fishery for endeavour prawns (although, anecdotally, operators sometimes target endeavour prawns towards the end of the fishing season) so the fishing mortality for endeavour prawns is the fishing effort targeted at the two tiger prawn species multiplied by species-specific catchability coefficients (computed using the methods of Wang (1999)).

6.3.5.2 FLEET DYNAMICS MODEL

The fleet dynamics model consists of a large-scale and a fine-scale component. The large-scale component takes the total annual tiger prawn effort set by management, allows for implementation error (i.e. scientific recommendations for changes in effort limits may be ignored so the total effort for year *y* may be same as that for year *y*–1 even though the management strategy suggested a change in effort (Dichmont et al., 2006a)), generates a banana prawn season, calculates the total available effort on both banana and tiger prawns, and allocates this total effort to week, and then target species (*P. semisulcatus* and *P. esculentus*, banana prawns), and region. The fine-scale component allocates this effort to the grid cells on which the benthic impacts model is based.

6.3.5.3 LARGE-SCALE SUB-MODEL

In recent years, the fishery has consisted of two sub-seasons split by a mid-year spawning closure. The first sub-season targets (common) banana prawns (scientific name) and the effort during this sub-season consequently depends very little on tiger or endeavour prawn abundance. The fleet switches to the tiger prawn fishery, for which catch-rates are lower but less variable, during the second sub-season although, if banana prawns are still available in large enough numbers, they will be fished by some of the fleet. As the length of the second sub-season is generally fixed, any effort expended on banana prawns is effort effectively taken away from tiger prawns. It is not possible to model the dynamics of banana prawns and hence include it in the operating model explicitly owing to an inability to characterize the stock-recruitment relationship for this species. Consequently, the amount of effort expended on banana prawns in any future year is selected by choosing a fishing pattern at random from those for past years.

The movement of individual vessels from one day of the season to the next is governed by a time-inhomogeneous Markov chain. Vessels are independently assigned to regions on a daily basis and regional effort is aggregated to week to match the time-step of the operating model. The transition probabilities of the Markov chain are functions of a number of key drivers, principally the absolute time of year, the elapsed fishing time within the season, aggregate catch rate measures for each region for the preceding week, and the average cost of travel that the transition in question implies. The parameters used to determine the transition probabilities are estimated from logbook data, using transitions that have actually occurred. Separate models are used for, and calibrated from, first and second sub-seasons.

Once the transition for a particular virtual vessel has been generated by the Markov chain, a decision is made whether the effort applies to the banana or tiger fishery based on empirical probabilities. This process imitates the action of the real fleet in a realistic way, and can respond to changes in local abundance in time and space, as well as more gradual changes in the cost of travel.

6.3.5.4 FINE-SCALE SUB-MODEL

The total effort in a region is subsequently allocated to about 3000 6-minute grid cells on an annual time step. The effort at this finer scale is subsequently used by the EwE food web model (section 6.3.3), the ERA model (section 6.3.4) and the EoT assessment model (section 6.3.1). Instead of a probabilistic transition model, which would be unworkable at this scale, an empirical approach is used that bases the effort distribution on historical patterns. Benthic impacts are sensitive not only to the total trawling effort, but also to the degree of aggregation: higher effort obviously results in higher impacts, but increasing aggregation tends to reduce the impact because effort is applied to areas that have already been depleted. Historically the degree of aggregation has been negatively correlated with total effort, implying that the vessels take fewer risks and stay close to known hot-spots when effort is low, and vice versa. Using the area index of Prince (pers. comm.) to quantify the degree of aggregation, we fit a power-law relationship between area index and total regional effort from effort data for 1970–2004.

To allocate effort by region to grid cells in each projection year, we selected a spatial distribution of effort from a randomly chosen but recent historical year, applied a power-law to the effort distribution to reflect the relationship between total effort and the degree of spatial aggregation of the effort, and then scaled the resulting “effort field” so that the total effort by grid cell equalled the total effort from the first component of the effort allocation model. Dorn (2001) and Hutton et al (2004) outline related approaches to effort allocation using probabilistic methods.

6.3.5.5 MANAGEMENT STRATEGIES

Dichmont et al. 2006 (b, c) and 2008 tested different prawn centric management strategies that, at their smallest spatial scale, were at the prawn stock region scale. They tested Maximum Sustainable Yield (MSY) and Maximum Economic Yield (MEY) related strategies. The Base Case strategy for this report is the MEY strategy presently used in the fishery and described in Dichmont et al. (2009).

This Base Case strategy is based on setting effort to maximise profit thereby moving the fishery to Maximum Economic Yield (MEY). Given a new assessment (undertaken every alternate year), this involves first determining whether any of the species is “overfished” (defined as the average spawning stock size over the most recent five years being below 0.5 S_{MSY}), in which case the fishery is closed until that species has recovered to above 0.5 S_{MSY}. If none of the species is assessed to be overfished, the sequence of future effort levels is calculated so that profit over a 50-year projection period is maximized (although the estimation has a lower bound so that effort in a given year is not less than half of that directed towards *P. esculentus* in 2006 (a value recommended by industry and management). The future efforts for the first seven years (with the effort for the eighth and all subsequent years set to that for the seventh year) are selected to maximise total profit. Future harvest levels and the spawning stock sizes for each of the four prawn species are projected from the most recent stock assessment and determining future recruitment from the deterministic component of the stock-recruitment relationship. As a result, each year that a bio-economic assessment is undertaken:

1. the MEY is recalculated i.e. the target is able to change over time based on the biological and economic inputs,
2. the most profitable pathway (as opposed to a linear pathway) to the target is calculated i.e., the optimal pathway is able to change over time, and
3. the effort for the next two years is set based on selecting the first and second year’s effort from the seven-year effort series, since an assessment is calculated biennially.

The profit function, which forms the basis for estimating the time-trajectory of future effort, accounts for costs due to labour, capital, fuel and other causes:

$$\sum_{t>t_{\mathrm{cur}}} \pi_t = \beta_t \sum_w \sum_s \left[v^s_{t,w} H^s_{t,w} - (c_L v^s_{t,w} H^s_{t,w} + c_M H^s_{t,w} + c_K E^s_{t,w} + c_F E^s_{t,w}) \right]$$

where π_t is the profit in future year t,

$v^s_{t,w}$ is the average price per kilogram for species s during week w of (future year) t (assumed exogenous as the product is exported);

$H^s_{t,w}$ is the harvest (kg) of prawns of species s during week w of year t;

$E^s_{w,t}$ is the fishing effort targeted at species s during week w of year t;

c_L, c_M is the share cost of labour and other variable costs per weight of output;

c_K , c_F is the average repairs and maintenance, and fuel and grease costs per unit of effort;

β_t is a discount factor (the rate at which future income or expenditures is discounted relative to the present value (Grafton et al., 2006)):

$$\beta = 1/(1+i)^{(t-t_{\mathrm{cur}})}$$

i is the rate of interest (assumed to be 5% per annum in this study); and

t_{cur} is the current year.

Cost parameters were derived from economic surveys of the fishery undertaken by the Australian Bureau of Agricultural and Resource Economics (ABARE). The economic survey does not divide the NPF into tiger and banana prawn fisheries. Therefore, the average revenue and costs per vessel are computed from the NPF sample as a whole and then recalculated for the tiger prawn fishery considering that the banana fishery fishes for 24 hours a day whereas the tiger prawn fishery is restricted to 12 hours of night fishing. The values for the economic parameters from the most recent two surveys and those used in the analyses of this paper are summarized in Table 9. Most of the costs were similar in 2004–05 and 2005–06, although the increase in fuel costs from 2004–05 to 2005–06 is particularly noteworthy. Values used in the analyses were based on the survey data for 2004–05 and 2005–06, but also accounted for recent industry advice on costs (D. Carter, Austral Fisheries Pty Ltd., pers. comm.). Fuel and gear costs (c_F) per unit of effort are estimated by dividing total fuel and grease costs by total fishing effort (total tiger fishing day equivalent).

Table 9. Summary statistics of the NPF (average per boat) for 2004/05 2005/06, 2006/07 and the first half of 2007/08. Data taken from Dichmont et al. 2008 NPF-RAG Assessment report.

		04-05	**05-06**	**06-07**	**07-08(1)**
Number of observed vessels		**24**	**24**	**33**	**33**
Part 1. Revenue and costs (average NPF vessel)					
Total cash income, including	*$*	$992,582	$1,061,117	$1,069,278	$547,800
Tiger income	*$*			$516,852	$424,935
Banana income	*$*			$490,264	$128,992
Endeavour income	*$*			$58,970	$45,231
Labour costs	*$*	$267,447	$263,950	$271,434	$126,720
Total materials costs	*$*	$49,035	$57,080	$73,358	$45,755
Fuel and grease costs	*$*	$296,786	$367,105	$293,493	$225,673
Repair costs	*$*	$142,368	$131,559	$126,774	$55,917
Gear costs	*$*	$34,706	$23,869	$19,265	$3,972
Part 2. NPF Vessel summary (Average NPF vessel)					
Number of crews on board	*persons*	3.2	3.5	4.3	4.3
Number of skippers	*persons*			1.15	1.12
Total crew days	*days*			832	487
Vessel size	*meters*	22	23	23.2	23.2
Vessel tonnage	*tonnage*	148	170	167	122
Engine power	*horsepw*	346	363	349	333
Tiger prawn landed	*kg*	21,998	23,547	24,580	21,963
Banana prawn landed	*kg*	38,592	49,257	48,856	17,428
Endeavour prawn landed	*kg*	3,019	4,680	4,927	3,492
King prawn landed	*kg*			409	397
Total prawn landed	*kg*	65,801	79,264	78,772	43,280
Part 3. Economic parameters (Average NPF vessel)					
Share of labour per/$1 income	*$*	$0.29	$0.26	$0.25	$0.23
Share of other cost/ kg	*$/kg*	$0.68	$0.67	$0.93	$1.06
Tiger fishing					
Repair cost	*$/day*	$681	$802	$678.4	$483.8
Fuel cost	*$/day*	$1,440	$2,251	$1,570.6	$1,952.6
Gear cost	*$/day*	$167	$146	$103.10	$34.37
Banana fishing					
Repair cost	*$/day*			$1,356.87	$967.63
Fuel cost	*$/day*			$3,141.29	$3,905.25
Gear cost	*$/day*			$206.20	$68.74
Average price for tiger prawn	*$/kg*	$17	$21	$21.03	$19.35
Average price for bananas	*$/kg*			$10.03	$7.40
Average price for endeavour	*$/kg*			$11.97	$12.95

Notes: All values are in 2007-08 prices.

Endeavour prawns are essentially caught as a bycatch of targeting tiger prawns. This means that management is essentially aimed at tiger prawns and endeavour prawns are only considered in the bio-economic model as added revenue while the catches of endeavour prawns only increase costs through catch-associated costs (i.e. packaging, labour etc.).

All other strategies are compared to the Base Case which has uses no spatial management beyond those already in place in the fishery for small prawns and habitat protection.

6.3.5.6 SPATIAL MANAGEMENT SUB-MODELS

The spatial management sub-models apply management at the scale of the 6-minute management cells by marking cells as closed to fishing. The management can be fixed, where closures are decided in advance and do not change, or adaptive, where the location of closures depends on the information from the EoT assessment sub-model.

When the management is fixed the models are very simple: any cells inside a closure are marked as closed to fishing. This information is passed to the fine-scale fleet dynamics sub-model, which distributes the effort whilst respecting the closures.

When the management is adaptive, however, the selection of cells is more complicated. The EoT assessment sub-model passes its habitat indicators to the spatial management sub-models. If any indicator I_{hs} falls below a threshold T, the spatial strategy model signals the spatial closure module to apply a partial closure in the habitat h. The formal decision rule is as follows:

> Rule 1: if $I_{hs} < T$ for any species s, then apply a partial closure to habitat h. If habitat h is already closed, keep it closed with the same closure as before.

The spatial closure module uses recent spatial effort pattern $\overline{C}^{hg}(t)$ (section 6.3.1) to decide which cells in the habitat to close. All cells in which the recent coverage is below the median coverage within habitat h are closed permanently. The formal decision rule is as follows:

> Rule 2: if $\overline{C}^{hg}(t) < \text{median}_g\{\overline{C}^{hg}(t)\}$ for grid cell g, then close the cell to trawling permanently.

Note the closed grid cells are determined once, only at time t, when the action is triggered. In addition, it is only the less fished cells that are closed: the consequence of the management action is to aggregate the fishing further into the fishing hot spots. It is assumed that effort at the stock level remains the same with or without a closure and therefore the small-scale fleet dynamics model is used to shift the effort from the closed grids to the remaining grids within a stock based on historical patterns, which therefore considered that certain grids are more favoured than others (Dichmont et al. 2008).

In this project, six spatial scenarios were considered: a base case with no closures; three fixed closures based on interactions between the distribution of effort and the distribution of tiger prawn derived from a risk assessment; a fixed closure based on the concept of a marine protected area; and the adaptive closures based on $T = 0.7$ and the four most vulnerable species groups (Table 10)

Table 10. Scenarios used to test various spatial management options thereby describing the operating model, which management scenario is applied and which additional spatial closures were used. Low spot= low fishing effort; Hot spot= high fishing effort; L= low ERA species biomass; H= high ERA species biomass; MPA = conservation closure; EoT70 = 70% biomass change threshold in the EoT model.

Scenario	Name	Acronym	Management scenario	Closures
Status quo. No changes in fishing and set as 1990 as reference year	Base Case	BC	Bio-economic model	None
Examples of representative closure of benthic biodiversity for conservation reasons	Marine Protected Area	MPA	As above	Ad-hoc closures
Adaptive closure via simulating the effect of trawling (EoT model) with % threshold. Once cells are closed remains closed.	EoT 70%	EoT70	Effects of trawling model with a limit reference point of 70% change of initial biomass for target functional group in a habitat	Set closures adaptively.
ERA closure on low biomass density (low spot) and low fishing effort. Cells close low fishing effort and externally derived biomass density cells	Low spot Low	LdLf	ERA closures	Close cells when ERA threshold criteria are reached
ERA closures on low biomass density (low spot) and high fishing effort. Only close cells from those with low density and high effort	Low spot High	LdHf	As above	Close cells when ERA threshold criteria are reached
ERA closure on high biomass density (hot spot) and low fishing effort. Close cells from high biomass density and low fishing effort cells	Hot spot Low	HdLf	As above	Close cells when ERA threshold criteria are reached

6.4 OBJECTIVE 4

6.4.1 SPATIAL MSE AND THE SOFTWARE FRAMEWORK

For the delivery of a spatially explicit management framework for the NPF, we chose to use a Management Strategy Evaluation (MSE) simulation framework (Smith 1994, 1999, Dichmont et al. 2008, Ellis et al. 2008) because it allowed the integration of the various models into a single framework. The spatial MSE models the whole management and biological systems together allowing comparison and evaluation of the relative performance of different management strategies (see also Sainsbury et al 2000). MSE comprises two parts: a management model and an operating model. The operating model can be considered as a 'virtual' resource and is seen as a representation of the 'true' underlying dynamics of the system and the fishery. The operating model in this case includes the biology of the benthos and ecosystem (including prawns) and all the processes that control the dynamics of that system. The management model uses information obtained from the operating model to make decisions on how to manage the fishing fleet. The management model remains 'ignorant' of the 'truths' included in the operating model other than the data given it. Each combination of the types of data used, the assessment related analysis method applied, and the decision rules used constitutes a different management strategy. The outcome of the management strategy (e.g. the level of effort, which areas and times are open to fishing etc.) is fed back to the operating model and is used to determine the dynamics of the 'true' situation being managed. The key component that links the management strategy and the operating model is a spatial fleet dynamic model that applies the effort spatially and temporally.

The diagram in Figure 5 describes the main components of the MSE system. Each box represents a separate sub-model of the system that is implemented in software. The outputs of one model become the inputs of the next model in the system. The system comprises a pair of loops. As a time step in the simulation proceeds, data are passed from one model to the next in sequence. A single orbit of a loop corresponds to a single timestep in the simulation. The upper loop is the large-scale sub-model, which describes the system at the scale of stock regions. The lower loop is the fine-scale sub-model, describing the system at the scale of 6-min grid cells. The models and their provenance in (section 6.3) and elsewhere are set out in Table 11.

The management and operating models are separated by the dashed line. This is to emphasise that there is no communication between the two, except by way of the observed catch and effort and the imposed total effort and closures. Models in the same column belong to the same category: the assessment models attempt to estimate the state of the real world, using the catch and effort information provided by the operating model; the decision models use the assessment results to decide on a strategy; the action models work out how to (or whether to) implement the strategy; the human response models determine the response of the fishing fleet; and the biological response model determine the effect of this fishing on the ecosystem.

6.4.1.1 THE SOFTWARE FRAMEWORK

The software framework used in this project evolved from a series of projects starting in the GBR (Ellis and Pantus, 2001), in which the MSE was a fairly monolithic application, and moving to the Torres Strait (Ellis et al, 2008) and the GoC (Dichmont et al, 2008), in which the code was improved by breaking it up into autonomous components. In these projects the communication backbone was introduced, in which the separate components of the MSE interact using a messaging system (see Figure 6). The backbone was a breakthrough in the development of MSE software because it made the system much more robust and flexible, with the potential to extend to a wide variety of models support by a wealth of services. Furthermore, the framework is still undergoing vigorous development in other projects (Catchment to Coast MSE in SE Queensland).

The software framework provides certain operations that are always required by MSE:

a. MSE specification, running of MSE scenarios and reporting.

Specification involves the setting of input parameter values and the permanent storage of those values. Input parameters comprise all the data the MSE needs to run, including high-level parameters (e.g., the name of the MSE, how many replicates, start and end times), low-level parameters (e.g., depletion rates, closures) and also which quantities to output (e.g., names of performance measures and state variables). In our framework all parameter values are stored in a relational database, with an MSE table at the top of the hierarchy, and with all parameters joined by cascading links to a single record in the MSE table. The framework also provides a graphical user interface (Figure 7) with which the expert user communicates with the database. This user interface was designed to work with any model specification, and so it is not customised for user-friendliness.

Running the MSE scenarios comprises a parsing stage, in which the specification is retrieved from the database and the appropriate software modules are instantiated, and a simulation stage, in which a master program initialises the modules with data and runs them sequentially within a time-stepping loop.

Reporting involves the synthesis of information output by the MSE. All outputs (which have been earmarked for output at the specification stage) are stored in an output database, together with the context information of which MSE they belong to. The form of the table is highly standardised, and is geared towards storing scalars, time series, spatial data and spatio-temporal data. This makes it very simple to interrogate.

The MSE framework also provides some services such as data access, a state-variable monitor for debugging and validation, and some utilities such as matrix libraries. The intent is for the framework to allow modellers to concentrate on refining their own models without the distraction of having to build the supporting MSE infrastructure from scratch.

b. The use of the framework in this project

In this project, each of the yellow boxes in Figure 5 comprised a separate module. As we have said, modules communicate with one another via a generalised communication interface, provided by the software framework. The advantage of the communication interface is that in principle two different modellers (say a fishery economics expert and a stock assessment expert) could plug their models into the same MSE. To allow the models to work, the experts only need to ensure their modules conform to the communication interface. This was demonstrated in this project as shown by the following examples.

The large-scale components (the brown boxes in Figure 5) were originally implemented as stand-alone Fortran code. This code was then modularized into a management model component and an operating model component. We then built a cross-language service to allow the framework to invoke the Fortran code and vice versa. Finally, we created two wrapper modules inside the framework that used the cross-language service to call the Fortran models. The framework only communicates with the (relatively simple) wrapper modules, but the real work is done in the Fortran code, which the framework does not need to know about.

A second example is the EwE module. Normally EwE is used as a stand-alone application with everything controlled from its own user interface. However, the authors of EwE have provided an alternative way to access to their software via the dynamic link libraries (DLLs). This makes it possible to use EwE as a server to client application, such as our MSE framework. The EwE module in the framework is another wrapper that passes data to the DLLs and receives data from the DLLs. Again, the framework does not need to know the details of how the module runs EwE; it only needs to be able to exchange data with the module via the communication backbone.

The actual implementation of the EwE module is interesting technically. The EwE DLL is set off running in its own processing thread, separate from the main thread of the EwE module. At certain points in its execution, the DLL thread raises events, which can be handled by the module. For example, events are raised when the EwE scenario is loaded, at the start and end of each time step, and just after the spatial gravity model has been calculated. By intercepting these events, the module can retrieve data from the DLL thread or change data on the thread. In our case, we used the ‘gravity model’ event to overwrite the tiger fleet effort with effort from our fine-scale fleet dynamics model, and the ‘end time step’ event to extract the grid cell densities for each functional group.

In principle, all the modules in Figure 5 can be run simultaneously within the framework. However, it is not necessary to do this because the brown modules not depend on the yellow modules. This means that the large-scale part of the MSE can be run in the absence of the fine-scale part. This proved to be very convenient because we then only needed to run the large-scale part once, collecting the regional effort to a data base. To run the fine-scale part we replaced the large-scale components with a single stub module that fed the regional effort to the fine-scale fleet dynamics module as in Figure 8.

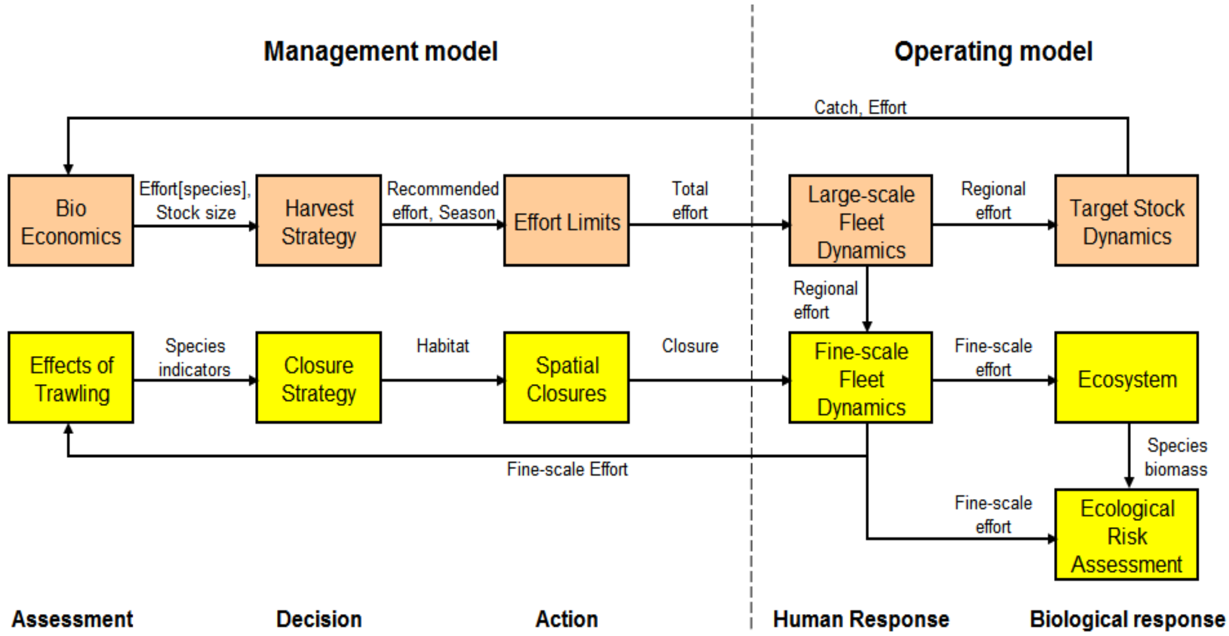


Figure 5. Schematic diagram of the MSE system. Each box represents a separate sub-model. Communication between sub-models and the kind of data supplied are shown by the labelled arrows. The dashed line separates the management model from the operating model. The models also splits into large-scale components (*brown*) on the scale of stock regions, and fine-scale components (*yellow*) on the scale of 6-minute management cells. The only link between the large-scale and fine-scale components is the regional effort from the large-scale fleet dynamics sub-model.

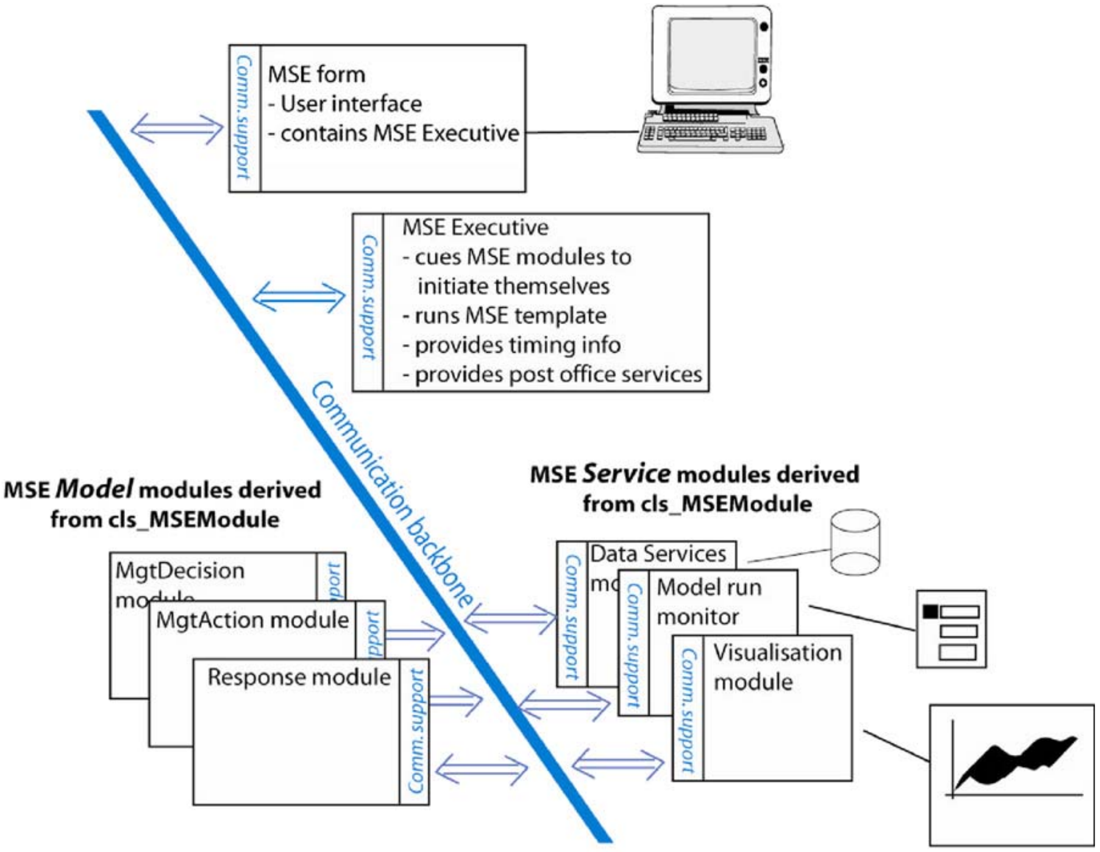
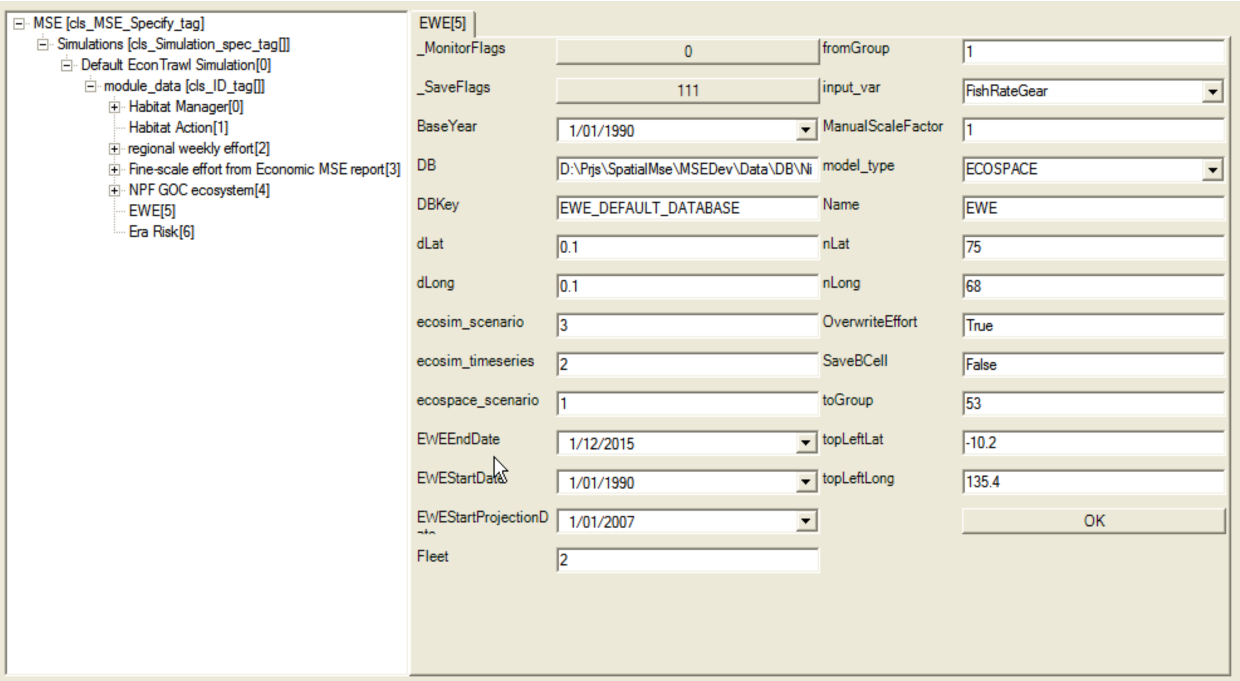


Figure 6. The architecture of the MSE Framework. All sub-models and all services are encapsulated in software classes (or modules) derived from a single base class (cls_MSEModule). The base class supports the communication interface that is the messaging system. A separate executive class controls the running of the simulation and the sequencing of the modules. Operations that are essential to any MSE such as saving results, visualisation and diagnostics are provided by the service modules

Table 11. The components of the MSE and where they are described.

Component	Section	Reference
Bio Economics	9.3.5	Dichmont et al. 2008; Kompas et al. 2010
Harvest Strategy	9.3.5	Jarrett and Dichmont 2007, Dichmont et al. 2008
Effort Limits	9.3.5	As above
Large-scale Fleet Dynamics	9.3.5	Venables et al. (2009)
Target Stock Dynamics	9.3.5	Dichmont et al. (2003)
Effects of Trawling	9.3.1	Dichmont et al. (2008)
Closure Strategy	9.3.6	this project
Spatial Closures	9.3.6	this project
Fine-scale Fleet Dynamics	9.3.5	Venables et al. (2009)
Ecosystem	9.3.3 and 9.3.2	Christensen & Pauly 1992, Pauly et al. 2001 and Christensen & Walters 2004
Ecological Risk Assessment	9.3.4 and 9.3.2	this project, Zhou et al. 2008, Browne et al. unpub. ms.



_MonitorFlags	0	fromGroup	1
_SaveFlags	111	input_var	FishRateGear
BaseYear	1/01/1990	ManualScaleFactor	1
DB	D:\Prjs\SpatialMse\MSEDev\Data\DB\Ni	model_type	ECOSPACE
DBKey	EWE_DEFAULT_DATABASE	Name	EWE
dLat	0.1	nLat	75
dLong	0.1	nLong	68
ecosim_scenario	3	OverwriteEffort	True
ecosim_timeseries	2	SaveBCell	False
ecospace_scenario	1	toGroup	53
EWEEndDate	1/12/2015	topLeftLat	-10.2
EWEStartDate	1/01/1990	topLeftLong	135.4
EWEStartProjectionD	1/01/2007		
Fleet	2		

Figure 7. Part of the expert user interface. The scenario is represented as a tree structure on the left hand side showing the seven modules (Habitat Manager to Era Risk). Selecting an item on the left displays a detailed form on the right where individual values can be set. Here the EwE module specification form is shown.

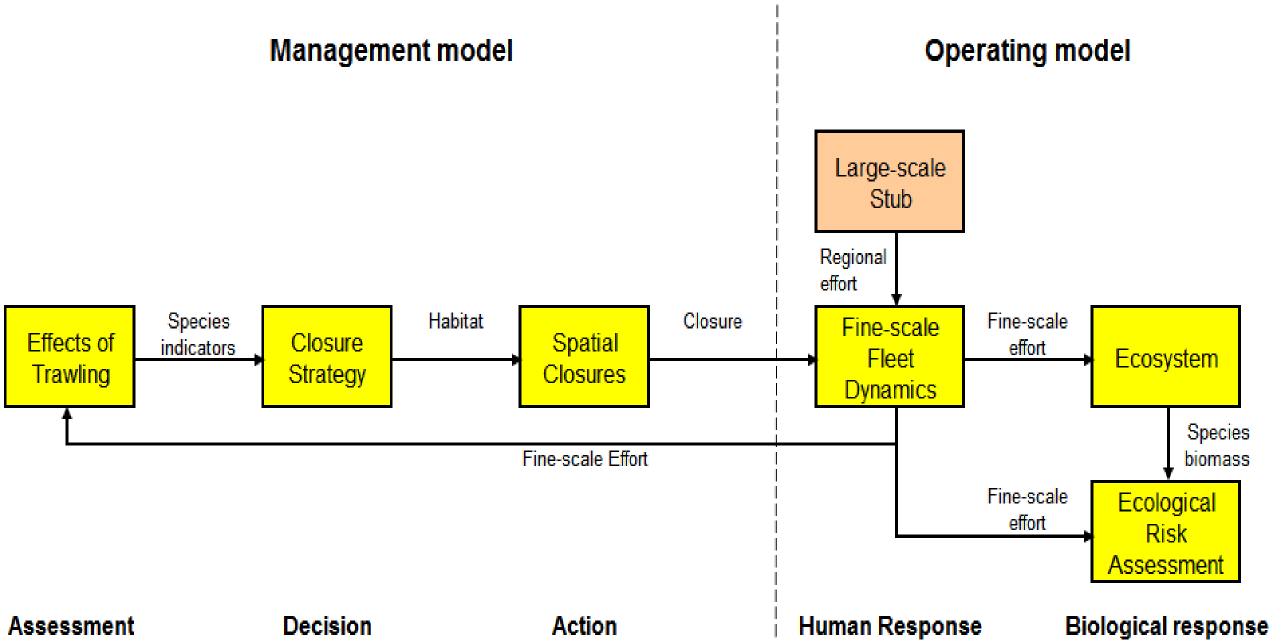


Figure 8. Schematic diagram of the MSE system with the large-scale components replaced by a stub module. These seven modules were executed together in the same MSE run.

7 RESULTS

This section, as do the methods, provides a succinct presentation of the major findings against the stated objectives. It presents only a synthesis of the main results that in our opinion satisfy the proposed objective. We have also included a brief discussion. Most of the details of analyses and related results are presented in their respective appendices.

7.1 OBJECTIVE 1: DETERMINE THE ACCUMULATED EFFECTS OF TRAWLING ON BENTHIC COMMUNITY STATE AND COMPOSITION

Appendix 6 presents the overall scientific manuscript with the detailed analyses and results obtained in this project to achieve this objective. In this section, we present the data derived from the natural experiment component of the SS 03/2005 survey in the southwest of the GoC. This work was carried out under the assumption that the data on state and composition of the benthic communities collected during this survey represents the cumulative effects of NPF trawling over its 30+ years of history. Thus, we evaluated the state (numbers and biomass) and composition (diversity, similarity) of the three major benthic components of the GoC benthic communities: (i) the demersal fish, (ii) the epibenthic invertebrates and (iii) the infaunal invertebrates.

7.1.1 COMMUNITY-WIDE EFFECTS

We found that the relationships between biomass, density and diversity of the benthic community in relation to factors included in the survey design strata varied among the three biotic components. Overall, the main effects explaining most of the variation in community state and composition were region (i.e. Mornington, Vanderlins, Groote) and, for fish invertebrates only, day versus night trawling (Table 12). Trawling intensity explained some variation, but was not the most important factor. The factorial analyses of variance (Table 12; Appendix 3) revealed no consistent relationship between the different levels of trawling (intensities) and the abundance, biomass and diversity of the three community components evaluated here. Trawling intensity as a single factor was only weakly, but significantly, related to the variation of invertebrate biomass and the equitability of benthic infauna (Table 12). Trawling becomes somehow important when combined with the regional effects, where their interactive effect is significant in explaining the variation of fish density, and infaunal biomass and equitability (Table 12).

Table 12. Summary of statistical probabilities derived from the factorial ANOVA applied to the SS 03/2005 density, biomass, and measures of diversity for fish, epibenthic and infaunal invertebrates. Significant values are in bold ($P \leq 0.05$). *S* = species richness, *E* = equitability.

	Fish			**Invertebrate**			**Infauna**		
	Density (ni/ha)	***S***	***E***	**Biomass (kg/ha)**	***S***	***E***	**Density (ni/ha)**	***S***	***E***
Trawl (T)	0.0632	0.5996	0.5161	**0.0400**	0.1724	0.5452	0.1035	0.4438	**0.0006**
Region (R)	**0.0001**	**0.0000**	0.1279	0.1258	0.7239	**0.0038**	**0.0041**	**0.0000**	**0.0000**
Day/Night (D)	0.1184	**0.0000**	**0.0070**	0.6416	0.3917	**0.0154**	0.9101	0.6168	0.8728
T:R	**0.0032**	0.7364	0.8939	0.2954	0.4420	0.1561	**0.0000**	0.0991	**0.0000**
T:D	0.1952	0.1990	0.5047	0.2654	0.2040	0.8379	0.3359	0.0111	0.7634
R:D	0.3466	0.0649	0.7242	0.8788	0.6244	0.9607	0.8226	0.7570	0.7651
T:R:D	0.2938	0.8038	0.9917	0.8105	0.6594	0.2848	0.8867	0.0623	0.4879

Figure 9 depicts the box-plot values of community state in terms of the densities and biomass for fish, and epibenthic and infaunal invertebrates aggregated by region and trawl intensity. This figure clearly shows that there is great variability among the median values (± 95% CI) for all three biotic groups. It also confirms the ANOVA results where the median biomass for the epibenthic invertebrates is only slightly higher at low trawling intensities compared with the high and medium trawling (Figure 9). However, this pattern was not shown with the densities of fish and the infauna (Figure 9).

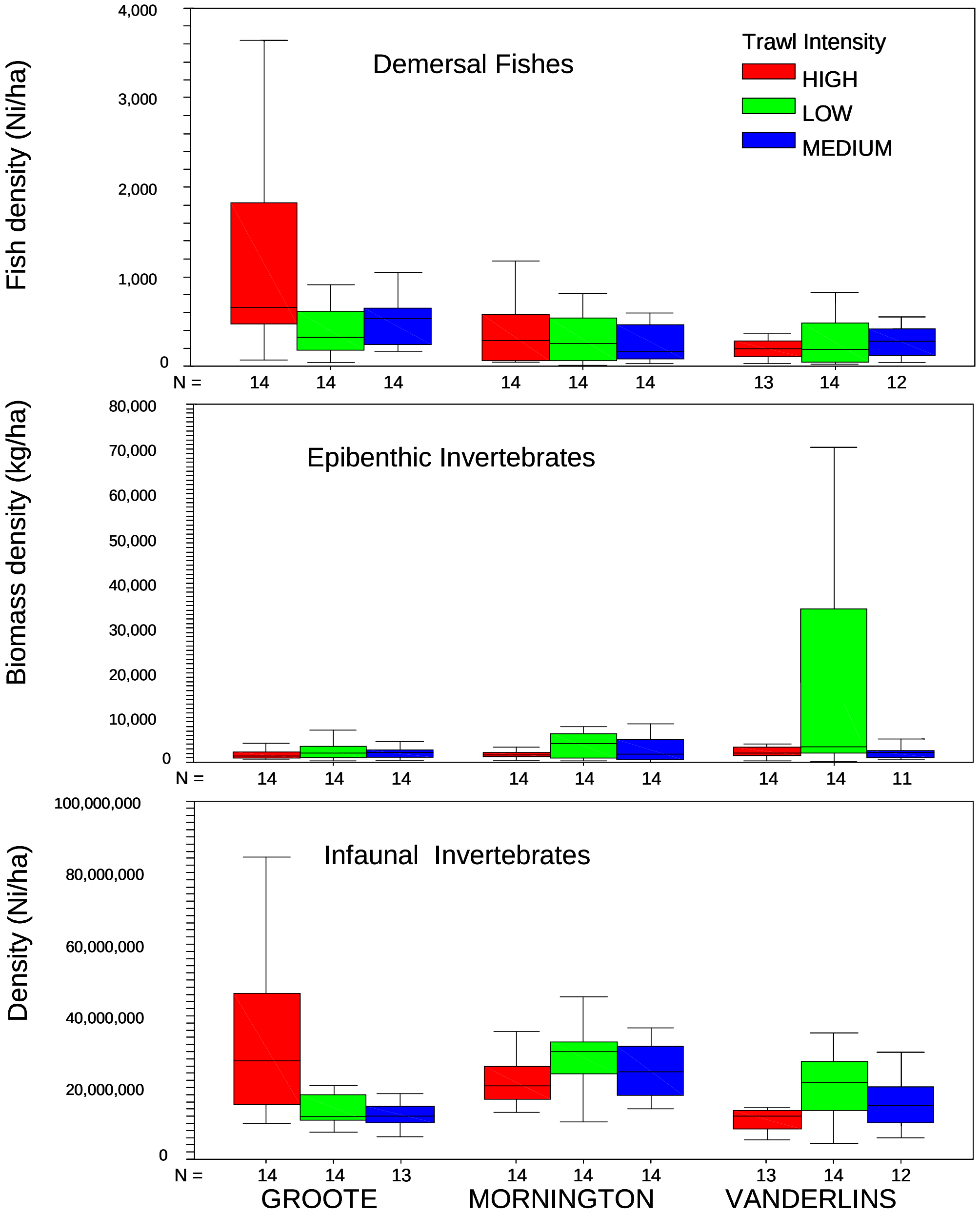


Figure 9. Median ± 95%CI (box) and range (lines) for the biomass and densities of fish, epibenthic and infaunal invertebrates in the three regions of the GoC obtained in the SS 03/2005 survey. N indicates the number sampling stations.

The results of the multivariate analyses (see Figure 24, Figure 25 and Figure 26 in Appendix 3) are also consistent with the univariate factorial ANOVA (Table 12), where demersal fish variation was explained largely by regional and day and night differences and weakly by trawl intensity (Figure 24 and Table 19 in Appendix 3). For the epibenthic invertebrates, a benthic group that includes sessile and mobile species, again the most important factors were the regional differences, which explained up to 12% of the biomass density variations, while trawl intensity, although significant, only explained 2% of the biomass density variation (see Figure 25 and

Table 21 in Appendix 3). For the infaunal invertebrates, the regional differences also explained most of the variation and trawling intensity explained at most 1% of the density variation, but only when interacting with the regions (see Figure 26 and Table 23, in Appendix 3).

In summary, it is clear that:

- the composition and density of demersal fish was related to region and day/night, and weakly related to trawling;
- the composition and the biomass of epibenthic invertebrates was largely related to region and weakly to trawling, but not to day/night variation;
- the composition of infauna was related to region and weakly to trawling, but not to day/night; and
- the relationships between the amount of bycatch and trawling intensity detected in these analyses do not in themselves indicate that trawling affects biodiversity.

These results are consistent with previous similar and related works conducted in tropical sedimentary ecosystems in Australia (e.g. Poiner et al. 1998, Hill et al. 2002, Burridge et al. 2003, Pitcher et al. 2004, Haywood et al. 2005, Pitcher et al. 2009).

At least some of the assemblage variation related to trawling may be caused by environmental factors that are correlated with trawling rather than trawling itself. To minimise that problem, our survey design minimised environmental variation within (but intentionally not among) the three sampling regions. The remaining environmental variation may include (a) factors that we know about and have data for (e.g. bathymetry), (b) factors that we know about but do not have data for (e.g. proximity to reefs or species mobility), and (c) factors that we do not know are important. We can therefore not completely address the confounding effects of environmental variation and trawling with additional statistical analysis. For that we would need to have conducted a before–after, control–impact (BACI) study rather than the control–impact study that was only possible for this analysis of the long-term, broad scale impacts of trawling. However, we can partially address the effects by performing additional analyses on environmental factors we have data for and by incorporating knowledge of life history into the design and interpretation of analyses.

Results from the analyses are consistent with partial effects of both trawling and habitat. The association of fragile animals such as bryozoans and brittle stars with low trawling is consistent with a trawling effect. The association of reef-associated fish with low trawling, and prawn predators with high trawling is consistent with a habitat effect.

The results demonstrate that the effects of trawling on the benthic biodiversity are not clearly demonstrated at whole-community level. There are however a number species and taxa that were negatively affected by high trawling intensities (e.g. sharks, rays, sponges, bryozoans, and gorgonians), but there were also as many taxa that seem to be more abundant or prevalent in areas with high trawling (e.g. dollar and pony fishes, some crabs and bivalves). This demonstrates that, at the current levels of trawling, management concerns should focus on those taxa that have been assessed to be at risk and that are specially protected, such as endangered and protected species.

7.2 OBJECTIVE 2. QUANTIFY KEY BENTHIC ECOSYSTEM PROCESSES OF IMPORTANCE TO PRAWN PRODUCTION AND BIODIVERSITY ALONG A TRAWL INTENSITY GRADIENT.

To achieve this project’s objective we evaluate two types of variables, (i) water column and benthos, and (ii) trophic.

7.2.1 WATER COLUMN AND BENTHIC SEDIMENT VARIABLES

The water column processes include various oceanographic variables among which are dissolved nutrient concentrations (ammonia, nitrate plus nitrite, phosphate, silicate), as well as chlorophyll-a concentrations. These variables were found to be relatively similar among in all regions (e.g. Figure 10, Figure 11). The major change in meteorological conditions halfway through the SS 03/2005 voyage (during and after the category 4-5 Cyclone Ingrid, 9-11 march 2005), and the subsequent mixing effects on the chemical and physical attributes of the water column, made inter-region and inter-trawling strata comparisons difficult and created some local temporal effects that influenced comparisons. Therefore the water column data only provides basic and descriptive information for the system and the results of the statistical analyses should be interpreted with care. Despite that, the limited and noisy water column data in general did not show any strong contrasts between the three regions and trawl intensities. Figure 10 shows the contrasting mean value of photosynthetic yield (as measured by a PHYTOPAM) at the bottom and surface of the GoC, that various regions and the effects post cyclone Ingrid. Production was significantly higher in the bottom sediments and there were no major differences among the regions. The lower yields in surface waters reflect photo inhibition of algal cells, and higher yields in the bottom waters reflect the capacity of cells from the bottom to photosynthesize. Cyclone Ingrid seems to have mixed up the waters well around Groote Eyland, but at the bottom, chlorophyll concentration was not significantly different from before the cyclone (Figure 10).

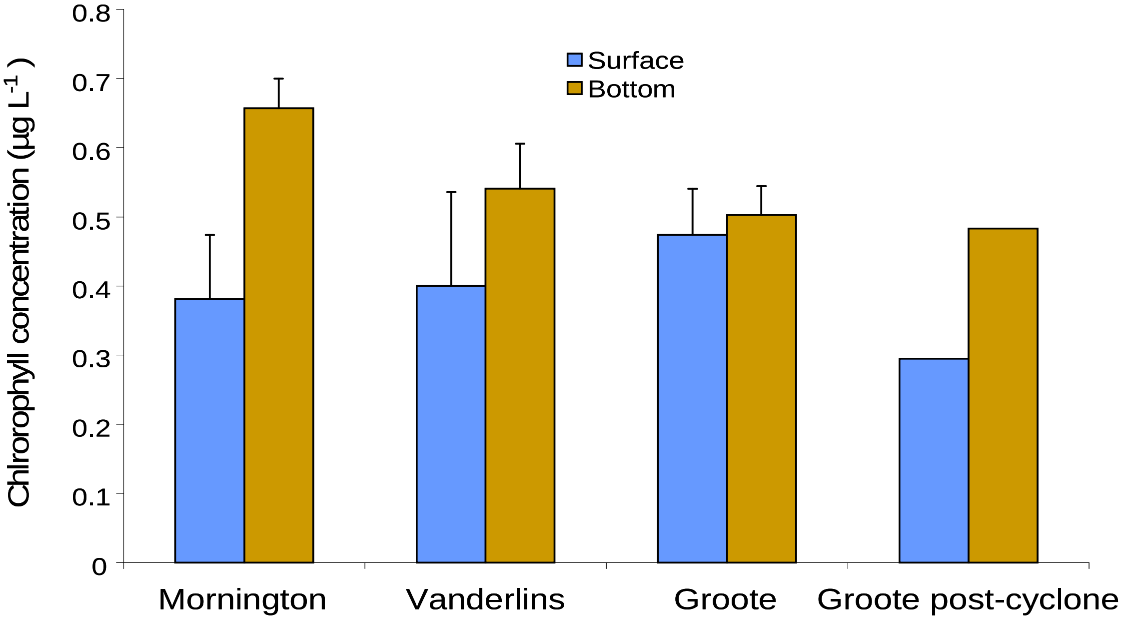


Figure 10. Mean (±SD) photosynthetic yield for phytoplankton collected from the surface and bottom at three regions in the Gulf of Carpentaria

As with the photosynthetic yield, there was in general a high regional variability for all water column and bottom sediment variables. The Figure 11 and Figure 12 provide a few of illustrative examples of such variability for selected water column variables (Figure 11) and bottom sediment variables (Figure 12). The major changes and differences were found among and within regions (e.g. for oxygen and ammonia concentrations) followed by the variations derived from the survey design along trawl intensities (Figure 11). However, despite the variation among trawl intensities, there was no discernible pattern and the variations were most likely explained by inter-and-intra site differences rather that trawl effects. This was similar for all other water column variables such as salinity, phosphates, temperature, nitrites/nitrate, silica (for more details see Appendix 4). Similar results were obtained for the bottom sediment carbon contents and carbon isotope variables, where the greatest variation was found at regional and intra-and-inter site levels while no consistent pattern was found with the various trawl intensities (Figure 12). Again, none of the other bottom sediment variables (%gravel, %mud, %sand, carbonates, denitrification) showed a clear and unequivocal relation with trawling intensity and all variation was explained by regional and inter-and-intra sites differences (see details in Appendix 4).

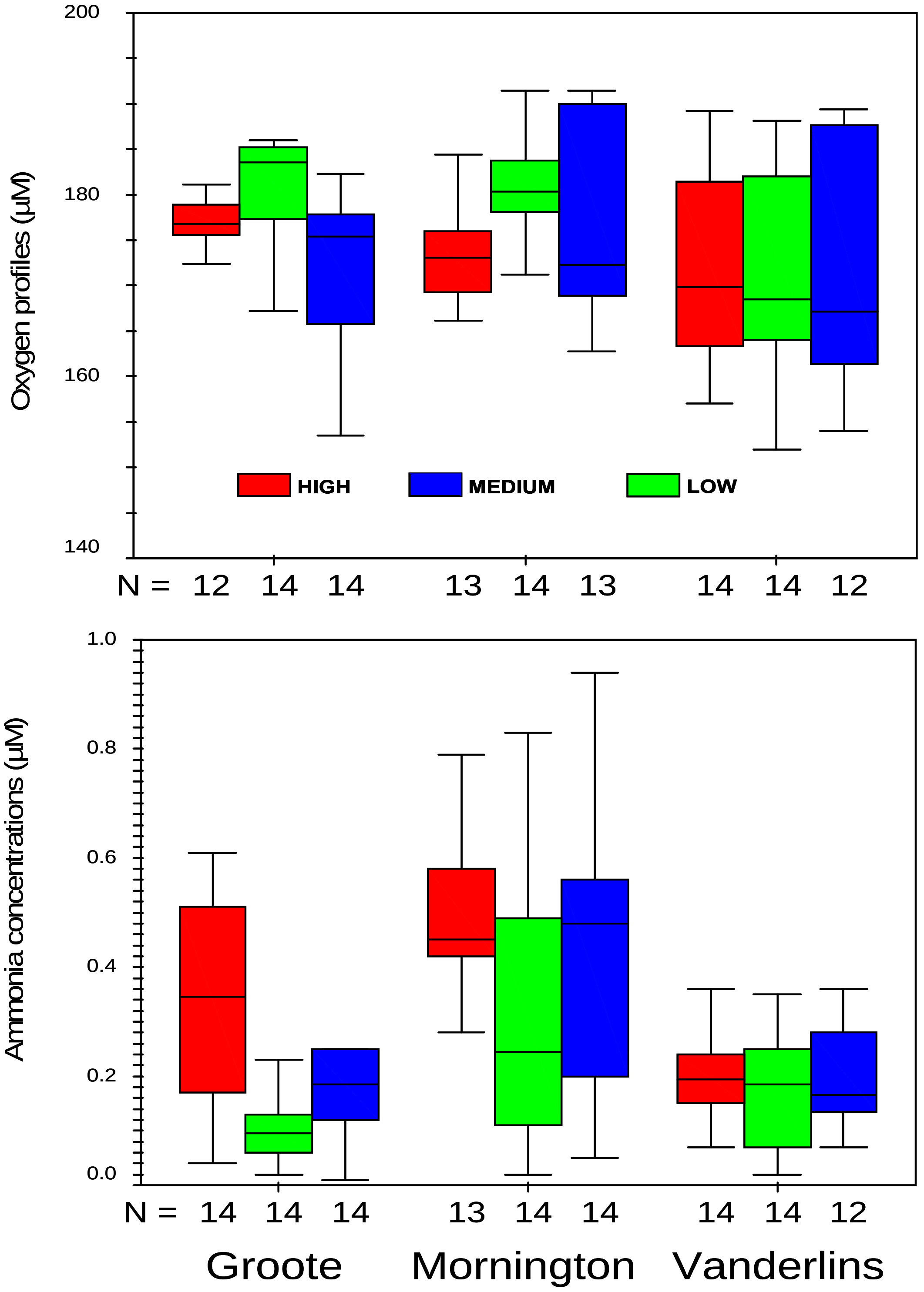


Figure 11. Median ±95%CI (box) and range (lines) for water column oxygen and ammonia concentrations for each trawl intensity in the three regions of the GoC obtained in the SS03/2005 survey. N indicates the number sampling stations.

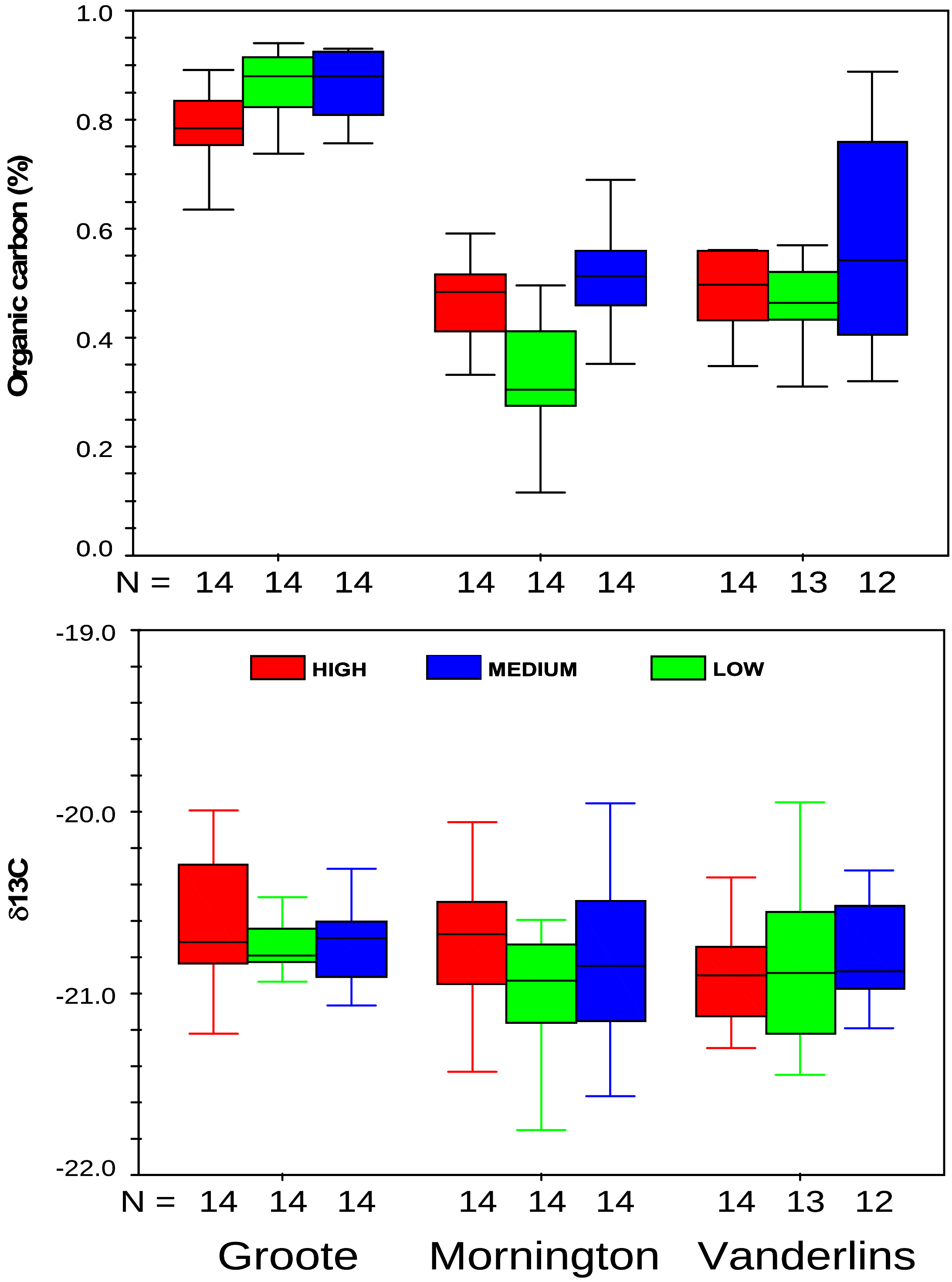


Figure 12. Median ±95%CI (box) and range (lines) for bottom sediment organic carbon content and carbon isotope concentrations for each trawl intensity in the three regions of the GoC obtained during the SS03/2005 survey. N indicates the number sampling stations.

When all water column and bottom sediment variables were included as covariates within a full factorial multivariate analysis of covariance (MANCOVA), and various measures of benthic biodiversity (density, biomass, richness, diversity, equitability) were used as responses variables, still no overall consistency in the trawling effects as explanatory factors for all biodiversity variability was evident (Table 13). Only the infaunal invertebrates showed significant effects of trawling in terms of density, diversity and evenness, but trawling had smaller effects compared with the variance explained by the regions and their interaction with trawling (Table 13). Overall, only few environmental covariates were significant, but there was some consistent effect of the three sediment composition variables (i.e. %mud, %sand and %gravel) on the density and biomass of invertebrates (Table 13). Similarly, only the nitrates/nitrate (Ni) concentrations in seawater consistently and significantly explained the variability in the species richness (*S*) of all faunal groups considered and of biomass of epibenthic invertebrates (Table 13). This can be explained through regional differences (higher nitrates/nitrate concentration in Groote region, see Appendix 5).

Table 13. Summary of statistical probabilities derived from the factorial MANCOVA applied to the SS 03/2005 density, biomass, and measures of diversity for fish, epibenthic and infaunal invertebrates. Twelve water column and bottom environmental variables were used as co-variables to the main design factors. Significant values are in bold ($P \leq 0.05$). *S* = species richness, *H* = Diversity, and *E* = equitability (or evenness).

	Fishes				**Epibenthic Invertebrates**				**Infaunal Invertebrates**			
Main factors	Density (Ni/ha)	*S*	*H*	*E*	Biomass (kg/ha)	*S*	*H*	*E*	Density (Ni/ha)	*S*	*H*	*E*
Multiple R^2	***0.583***	***0.740***	***0.477***	***0.335***	***0.341***	***0.740***	***0.420***	***0.401***	***0.673***	***0.696***	***0.738***	***0.707***
Trawl Strata (T)	0.625	0.956	0.871	0.916	0.622	0.956	0.270	0.792	**0.012***	0.486	**0.029***	**0.004****
Day/Night (D)	0.128	**0.000*****	**0.000*****	**0.014***	0.128	**0.000*****	0.084	0.087	0.949	0.865	0.418	0.424
Regions (R)	**0.008****	**0.000*****	0.583	0.489	0.307	**0.000*****	0.416	0.703	0.031	0.677	**0.001****	**0.001****
T:D	0.259	0.732	0.366	0.329	0.215	0.732	0.985	0.878	0.404	**0.042***	0.107	0.241
T:R	0.585	0.519	0.443	0.572	0.335	0.519	0.279	0.400	**0.000*****	**0.016***	**0.000*****	**0.000*****
D:R	0.842	0.706	0.697	0.595	0.509	0.706	0.947	0.236	0.362	0.624	0.525	0.116
T:D:R	**0.005****	0.224	0.335	0.377	0.505	0.224	0.790	0.593	0.644	0.113	**0.019***	0.181
Covariates												
%C	0.265	0.344	0.778	0.377	0.606	0.344	0.116	0.311	**0.022***	**0.030***	0.456	0.298
δC_{13}	**0.002****	0.083	0.567	0.816	0.859	0.083	0.260	0.345	0.711	0.070	0.243	0.912
S ‰	0.347	0.715	0.756	0.754	0.383	0.715	0.589	0.253	0.821	0.953	0.432	0.299
O_2	0.969	0.055	0.410	0.850	**0.044***	0.055	0.233	0.418	0.259	0.417	0.740	0.133
PHOS	0.117	0.362	0.776	0.908	0.589	0.362	0.052	0.358	0.991	0.420	0.170	0.230
Ni	0.217	**0.022***	0.269	0.886	**0.024***	**0.022**	0.933	0.360	0.115	**0.024***	0.103	0.975
SiO_4	0.145	0.385	0.714	0.969	0.633	0.385	0.837	0.750	0.467	0.373	0.462	0.713
NH_3	0.050	0.753	0.682	0.477	0.774	0.753	0.112	0.147	0.484	0.195	0.894	0.365
% Gravel	0.399	0.639	0.600	0.439	**0.048***	0.639	0.563	0.019	**0.010***	0.396	0.289	0.047
% Sand	0.399	0.639	0.600	0.440	**0.048***	0.639	0.563	0.019	**0.010***	0.395	0.289	0.047
% Mud	0.399	0.638	0.600	0.440	**0.048***	0.638	0.563	0.019	**0.010***	0.395	0.290	0.047
% $CACO_3$	0.351	0.301	0.462	0.869	0.772	0.301	0.287	0.557	0.208	0.380	0.934	0.550

7.2.2 TROPHIC VARIABLES

Using the body's isotopic signature of various infaunal functional groups as an indicator of possible trophic relations, we found that there were no consistent differences in the mean signature of δ13C and δ15N ratios between regions (Figure 13). Indeed, there was a wider range of δ13C and δ15N ratios between the various taxonomic groups than there was between regions, suggesting a range of food sources and trophic positions (Figure 13). These results do not suggest any relation with the trawling intensities. However, the low numbers of individual samples collected from each site prevent us from reaching major conclusions.

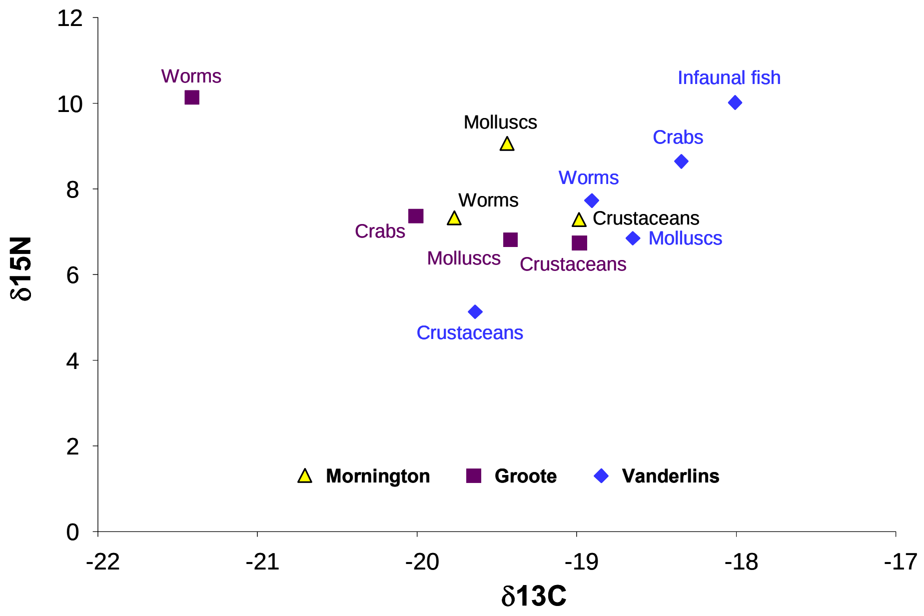


Figure 13. Mean δ13C and δ15N ratios for infaunal taxonomic groups in the three regions in the Gulf of Carpentaria

The diet and trophic similarity of selected benthic predatory fishes were different between regions, with 7 out of 8 fish species showing significant differences in their diet similarity (Table 14). This demonstrates, again, that the strong role of regional differences observed in biodiversity and environmental variables is also expressed in the food web of the GoC benthic predatory fishes. One species alone, the ornate threadfin bream *Nemipterus hexodon*, had a similar diet across the GoC (Table 14).

Table 14. ANOSIM results for the diet of 8 benthic predatory fish species: evaluation of the effect of regional differences on diet similarity.

Predator Species	**Global R**	**Regions significantly different to one another**	**Significance**
Caranx bucculentus #	0.604	Gro, Mor	**
Nemipterus hexodon #	0.406	--------------------	NS
Pentaprion longimanus	0.657	Gro, Mor, Van	***
Priacanthus tayenus	0.405	Gro, Mor, Van	***
Saurida macrolepis	0.418	Gro, Mor, Van	***
Selaroides leptolepis	0.655	Gro, Mor, Van	***
Terapon theraps #	0.745	Gro, Mor	***
Upeneus sulphureus	0.416	Gro, Mor, Van	***
# indicates species for which a comparison with the Vanderlin region was not possible			
NS Not Significant; * p < 0.05; ** p < 0.01; *** p < 0.001. Gro = Groote; Mor = Mornington; Van = Vanderlin			

In addition to regional effects, we found that five out of the eight fish species analysed exhibited clear differences in their diets in relation to trawling intensity (Figure 14). This is evidence that the benthic fishes are experiencing some dietary alterations in areas subjected to various degrees of trawling. This result is important since it suggests that the disturbance of trawling, whether because of prawn removal, bycatch and discards, and/or disturbance of bottom sediments, is affecting benthic trophic processes. This result reinforces and adds to the findings of other similar studies carried out in tropical Australia that have shown how trawling has detectable effects at the species and vulnerable taxa levels (Poiner et al. 1998, Hill et al. 2002. Burridge et al. 2003, Pitcher et al. 2004, Haywood et al. 2005, Pitcher et al. 2009).

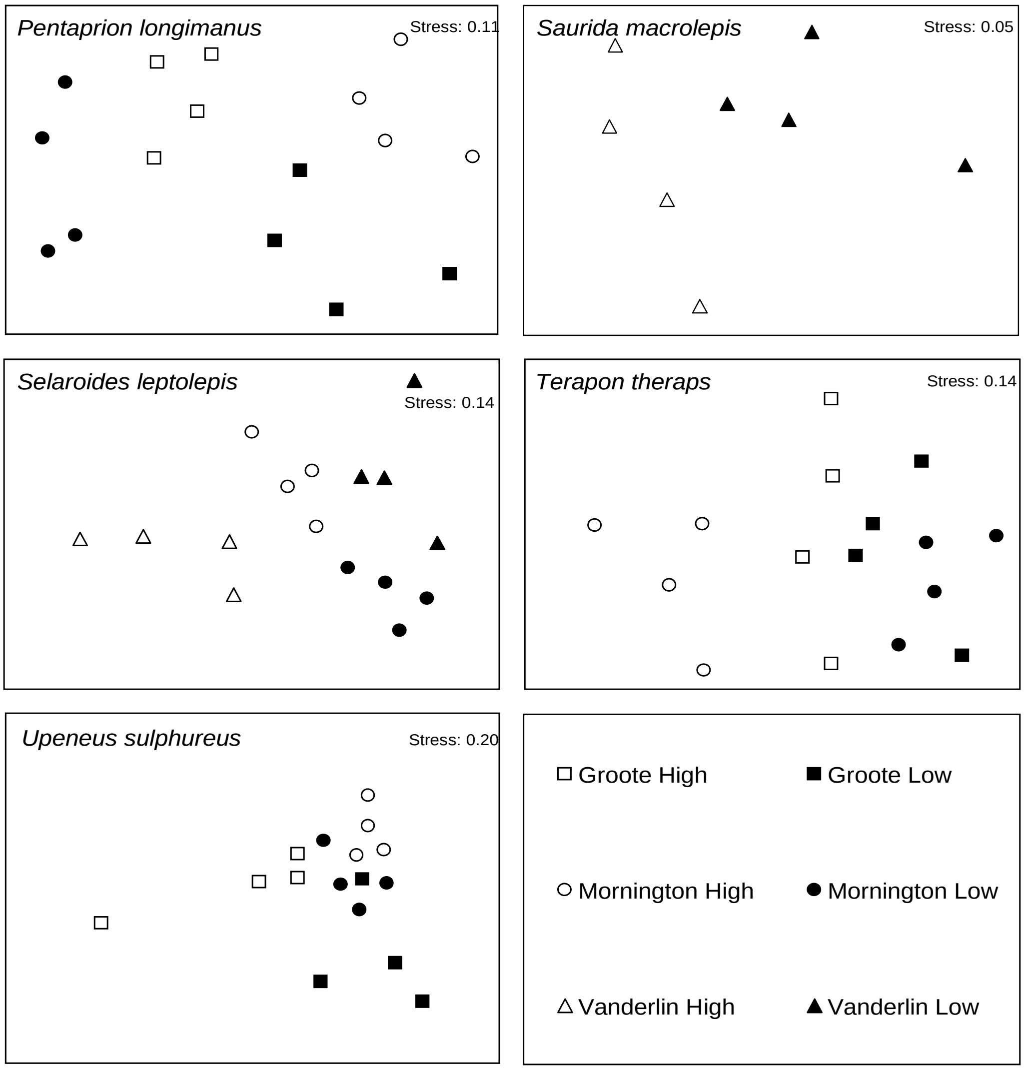


Figure 14. n-MDS ordinations plots for the diet of key and common benthic predatory fishes of the GoC whose diets were found to be significantly different. High and low trawling intensities are compared for each region.

In summary, for the environmental (water column and bottom sediment variables) and trophic processes we found that:

- the water column variable data were affected by cyclone Ingrid which caused some homogenisation in particular times and regions, allowing for limited comparisons,
- there was higher primary production in the bottom waters rather than at the surface or in the water column,
- surface phytoplankton were photoinhibited whereas bottom phytoplankton had the capacity to photosynthesize,
- the variation among and within regions is likely to explain most of the environmental variability of their associated processes. Only nitrates were clearly higher in the Groote Eyland region,
- trawling intensities did not explain the main environmental variations within the GoC,
- there was no general and unambiguous evidence of differences in sediment processes between trawling strata or regions. The exception to this was organic carbon concentration which was higher in the Groote Eylandt region than in the southern Gulf, but exhibited no differentiation according trawling intensity,
- there was no clear differentiation among the stable isotope signatures of infaunal biodiversity functional groups, with no discernible regional or trawling effects,

- the diets of several predator fish species differed at various regions,
- trawling effects were found to explain the dissimilarities of the prey composition of benthic fish diets, with diet differences found in five out of eight most common predatory fishes,
- overall, trawling effects appear not affect biophysical processes, but could be affecting the trophic processes at the species and taxon levels.

7.3 OBJECTIVES 3 AND 4 DEVELOP, AND PROVIDE FOR ADOPTION MANAGEMENT STRATEGY EVALUATION TOOLS FOR BENTHIC ECOSYSTEM IMPACTS. DESIGN AND DELIVERY OF A SPATIALLY EXPLICIT MANAGEMENT FRAMEWORK FOR THE NPF

Since objective 3 of this project was to develop and provide operational tools for integration within an overall spatial management framework (objective 4), there are no specific results other than the operational tools and models themselves (see section 6.3 above). Only the food web mass-balance model (EwE6) was used as a stand-alone application whose results are presented in Appendix 9.

The summary of the main results of the integrated spatial Management Strategy Evaluation framework (s-MSE), which represent the overall achievement of the two objectives, is presented below. We believe we have achieved the objectives by providing an operational way to evaluate ecosystem-based performance measures and by allowing to identify the trade-offs of multiple and alternative spatial management practices –i.e. the fundamental role of MSEs.

The outputs were multiple and of computational large sizes. Outputs could represent the mathematical product of several combinations of space-time splits in hierarchical simulations of: 6 spatial management scenarios, 12 habitats, 4 regions, 53 functional groups, 5 performance measures, 12 months, 9 years, 3,211 grid cells of 6-minutes, and 30 replicate simulations.

For example, for the evaluation of a biomass performance measure in the GoC grid, the results could be at least as big as 6 x 53 x 12 x 9 x 3211 x30 = 3,308,357,520 or more than 3 billion. The choice and selection of results will depend on the questions and evaluations required, so we choose to represent a range of outputs that we believe demonstrate the delivery of the operational capabilities of the spatial MSE.

7.3.1 SPATIAL MANAGEMENT SCENARIOS

The six spatial management strategy scenarios developed here were designed with the aim of modifying and altering the spatial distribution of trawling affecting the benthos while achieving the NPF's fisheries management targets (stock and economics). In essence, the resulting scenarios created a range of fisheries closures of various sizes that were designed to illustrate the operational implementation of ecosystem criteria and objectives to likely spatial restrictions of trawling in the GoC. The criteria used here include conservation and ecological risk criteria that in turn satisfy the current fishery management objectives.

Table 15 depicts the resulting fishing areas being excluded from trawling according to the spatial management scenarios. In the modelled area of the GoC (396,483 km^2 of surface), the trawling for tiger and endeavour prawn species is concentrated in only 20.8% of largely the southern and western shallows (<40-45m) shores of the GoC (Figure 15, Table 15). It was clear that the closures imposed by the scenarios are really small when the whole GoC is considered, being the maximum closed of 6.6% in the case of the MPAs (Table 15). In terms of closures sizes, the overall four MPA scenario had the largest closure with ~31% of the tiger-endeavour trawling grounds being closed, while the EoT70 scenario had the smallest closure with only ~6% of fishing ground closed (Table 15).

Table 15. Descriptive values for the various spatial management scenarios in relation to the total GoC modelled area and the tiger-endeavour fishing grounds. MPA = Marine Protected Area; BC= Base case (status quo); MPA-Grt- MPA Groote Eyland; MPA-Vnd= MPA Vanderlins; MPA- MPA Weipa;

MPA-Kar= MPA Karumba; LdHf= Low density High fishing effort; HdLf= High density Low fishing effort; LdLf= Low density Low fishing effort; EoT70= Effect of Trawling 70% threshold.

Scenarios	Acronym	# grid cells	Area (km^2)	Closure % of Trawling ground [a]	Closure % of GoC
Global modelled area GoC	GoC	3,211	396,483	-	-
Base case (status quo)	BC	669	82,606[a]	0	0
MPAs	MPA (total)	211	26,053	31.5	6.6
	MPA-Grt	37	4,569	5.5	1.2
	MPA-Vnd	52	6,421	7.8	1.6
	MPA-Wei	50	6,174	7.5	1.6
	MPA-Kar	72	8,890	10.8	2.2
ERA closures	LdHf	129	15,928	19.3	4
	HdLf	79	9,755	11.8	2.5
	LdLf	53	6,544	7.9	1.7
EoT 70% threshold	EoT70[b]	49±29	6,050±3,581	7.3±4.3	1.5±0.9

[a]= total area of trawling grounds; [b]= median ± standard deviation

Figure 15 (a-f) show the resulting spatial closures for each simulated management scenario. Closures were defined as individual or clusters of 6-minute grid cells closed to trawling. The number of closed cells varied according to the scenario and for all but the Eot70 scenario, the closed cells remain the same in all 30 replicated simulations. Being the Eot70 a dynamically adaptive scenario –i.e. closed cells are triggered when the threshold of 70% biomass change is reached in a given group-habitat, the number of closed grid cells varied from replicate to replicate and the ones depicted in Figure 15 (f) correspond to an illustrative example of a single replicated simulation.

The MPA ad-hoc scenario clustered all closed cells into compact closures while the ERA and EoT70 scenarios were more scattered (Figure 15). The number and clustering on each region depended on whether any given cell reached the closing thresholds for the various ERA taxa defined in Table 42and Table 44 in Appendix 10.

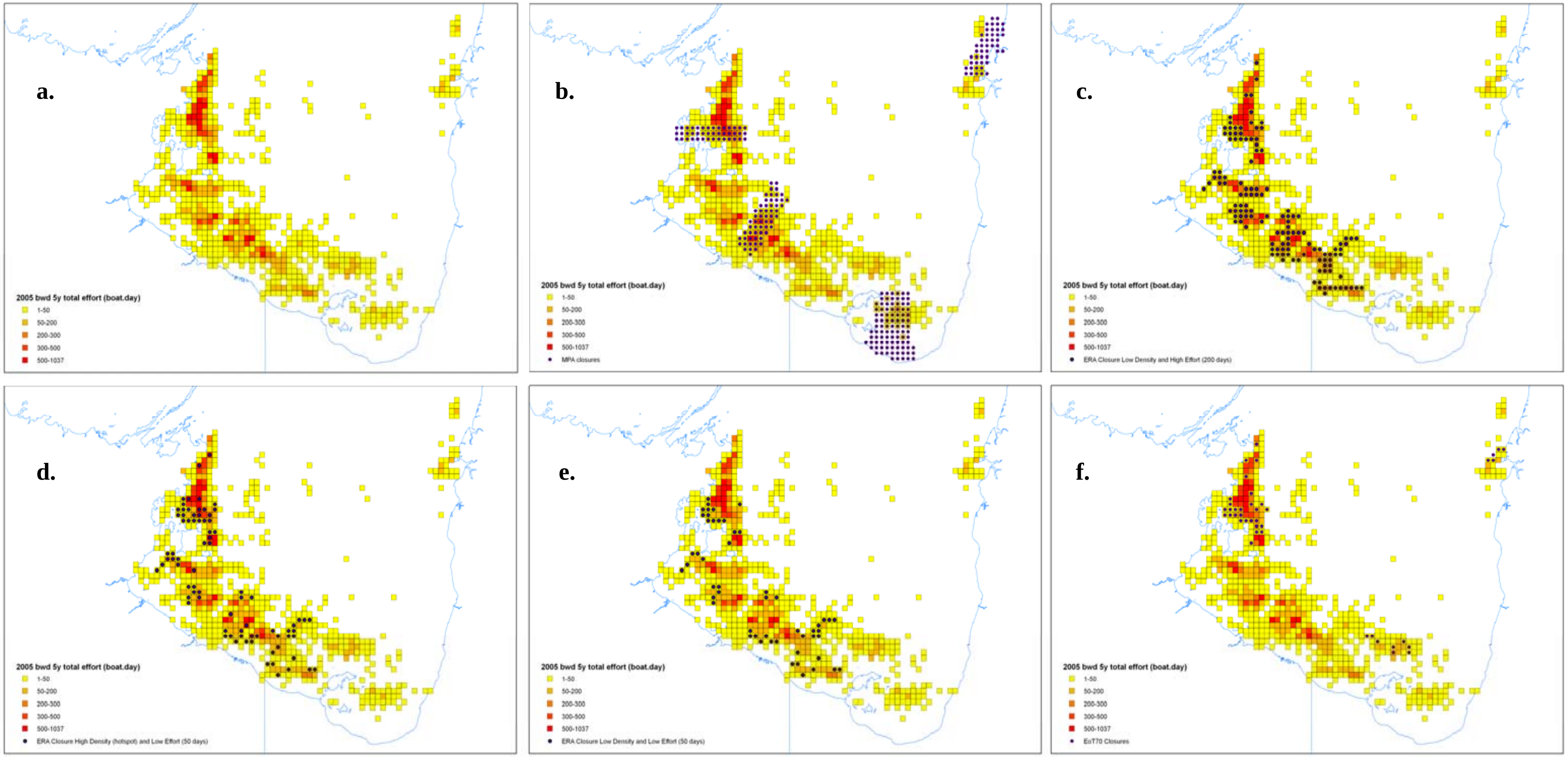


Figure 15. Maps the Gulf of Carpentaria depicting the 5-year average fishing effort from the NPF's logbook data (boat days) in 6-minute grid cells and showing the spatial closures scenarios for: (a.) Base Case (no closures), (b.). MPA), (c.) Low spot High, LdHf, (d.) Hot spot Low HdLf, (e) Low spot Low, LdLf, and (f.) EoT70 (illustrative example for a single replicate).

7.3.2 EVALUATION OF ECOSYSTEM CONSEQUENCES FOR ACHIEVING FISHERIES MANAGEMENT TARGETS

7.3.2.1 BIOMASS INDICATOR RELATIVE TO E_{MEY}

Using the Effect of Trawling operative model (EoT) in an adaptive manner, we evaluated the consequences of achieving the NPF's bioeconomic target (MEY) on target functional groups across all habitats and under all management scenarios. The target groups shown here were chosen to (i) demonstrate the capability of the spatial MSE and (ii) because they had been previously shown to contain some of the most susceptible taxa in the GoC to the effects of trawling (Haywood et al. 2005, Dichmont et al. 2008). Figure 16 depicts the temporal trajectories of the median predicted relative biomass of the sessile epibenthos and large gastropods carnivore functional groups (Figure 16 a and b, respectively). The dotted line in Figure 16 crossing at 0.7 (or 70%) indicates the trigger or threshold point that defines a trawling closure. This evaluation shows that overall, the performance of the indicator showed small variations that did not differ greatly across scenarios and habitats (<10% change), and only from 2010 onwards there was some variation in the indicator's trajectories (Figure 16 a,b). In the case of sessile epibenthos, no scenario triggers a fishing closure in any of the 12 habitats and 6 scenarios (Figure 16 a). Only in the Weipa and Mornington north fishing habitats there were some larger changes, corresponding to >10% to >30%% of the biomass (Figure 16 a). The trajectories of the indicator for the large gastropod carnivore group showed more changes –i.e. overall there were larger changes through time, and in the Weipa fishing habitat all but the MPA scenario crossed the set threshold (Figure 16 b).

This evaluation also shows that at the start (2008) the value of the indicator tends to be around 1 (or 100%), with hardly any variation among the scenarios, but from 2010 onwards the indicator does vary among of the various habitats and scenarios (Figure 16 a,b). These changes are likely due to the increased fishing effort, as predicted by the bioeconomic stock assessment (see Figure 65 in Appendix 10).

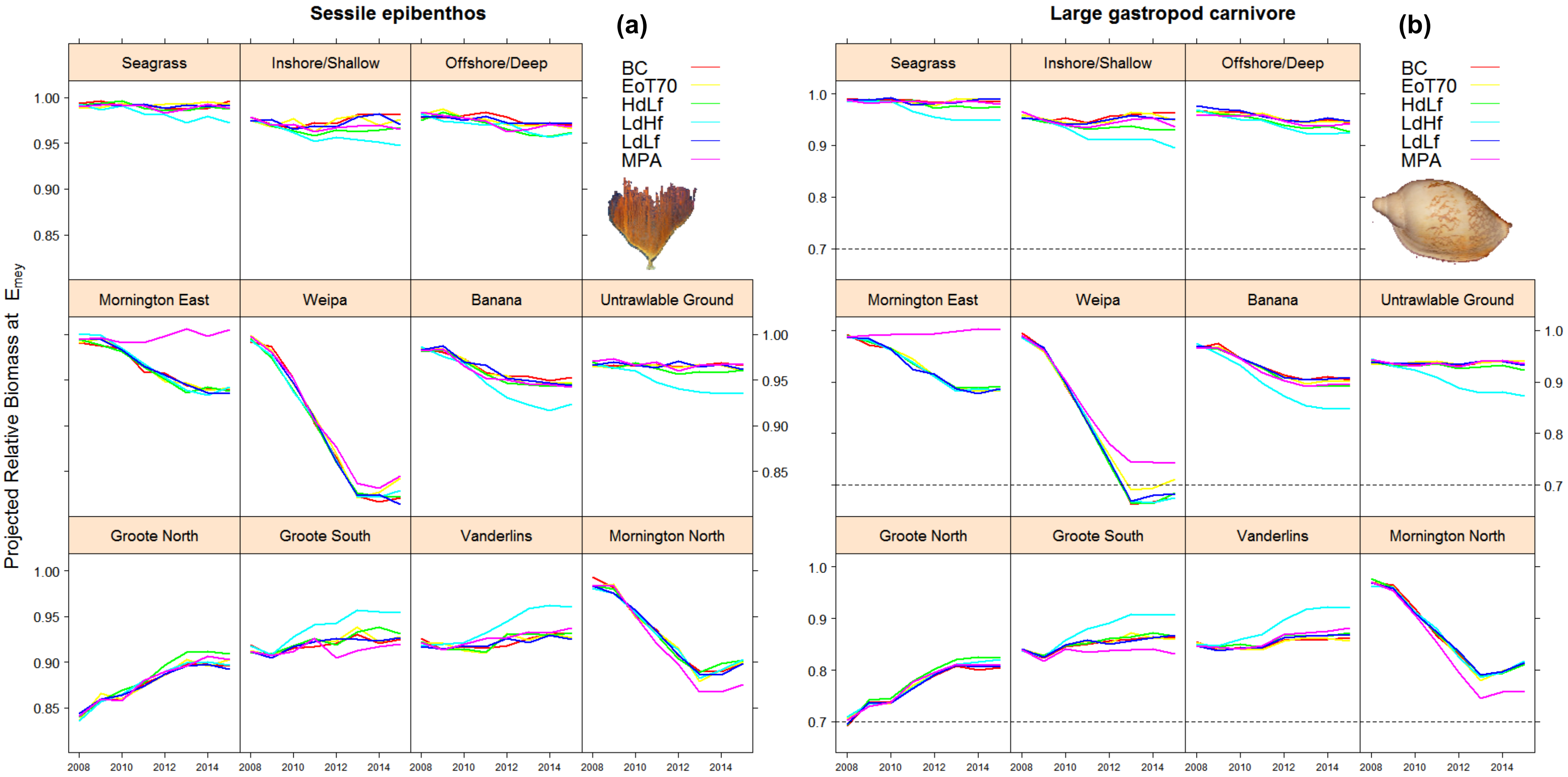


Figure 16. Performance of the relative biomass indicator in relation to the fishery management target E_{MEY} for all spatial management scenarios and for two illustrative functional groups (a) sessile epibenthos and (b) large gastropod carnivore. The evaluation is done by the EoF operative model.

7.3.3 REGIONAL EFFECTS OF FISHING – TEMPORAL CHANGES IN BIOMASS

To evaluate the spatial footprint of trawling under multiple spatial management strategies (closures), the modelled area of the GoC was divided into 12 major habitats (see Figure 55 and Table 37 in Appendix 9), where trawling occurs mostly in a few habitat types (see 5-year average fishing effort in Figure 15). Therefore there was no added information in listing (no) results for the areas not affected by trawling. For these reasons in this section we are present the results of the spatial MSE for only 4 major regions where trawling does occur in some form and intensity.

Figure 17 depicts the performance of the predicted relative biomass of all functional groups under all six spatial closures management scenarios for the simulated period 2008-2016 in the main tiger-endeavour fishing region. The results are expressed as the proportional change under a given scenario in relation to the base case (BC) and values are shown only for the year 2016, the end of the simulated period.

It is clear that no changes or relatively low and small changes in biomass were experienced for most functional groups (Figure 17) under current and predicted trawling efforts in relation to the BC (status quo). These results also confirm that most of the significant changes, both positive and negative, occur in the tiger region (Figure 17), where most of the trawling for tiger-endeavour prawns occurs. Negligible changes did occur in the inshore and offshore regions (not shown here but see Appendix 10) where there is very little or no trawling (see Figure 15). The highest values of the predicted median biomass changes did not exceed 2% beyond what was experienced in the BC scenario, and corresponded the ERA LdHf scenario which showed the most changes, followed by MPAs and ERA HdLf (Figure 17). The adaptive spatial closures of EoT70 and the ERA LdLf scenarios caused minimal changes across all taxa groups and regions. Despite the small changes detected across all scenarios, the relative biomass of prawns were slightly affected –e.g. adult tiger prawns were positively affected by ERA LdHf closures but negatively by MPAs (Figure 17). In general all closures tend to predict increases in top predator groups (e.g. sharks), that in turn feed on secondary consumers (e.g. prawns), fact that could explain the decrease in prawn biomass within MPAs (Figure 17).

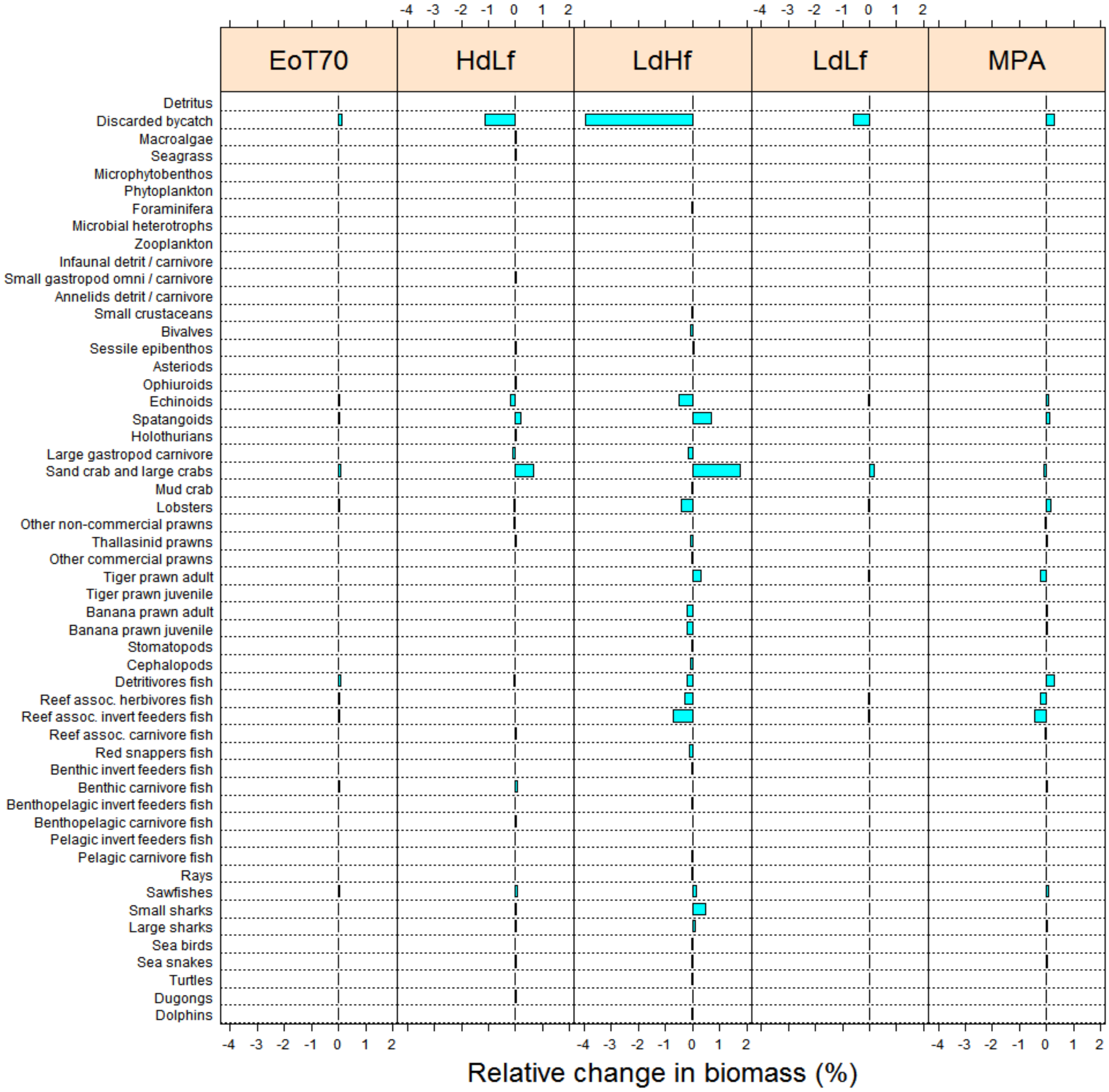


Figure 17. Changes in relative median biomass predicted for all FGs for all six modelled scenarios projected to 2016 in relation to the base case (BC) in 1990. Values are expressed in % and reported for the tiger-endeavour fishing habitat.

7.3.4 BIOMASS VARIATION PER HABITATS

The spatial MSE also allowed zooming spatially and temporally to asses the distributions of the predicted biomass for each habitat throughout the simulated period. This assessment was made for the median biomass under the various spatial management strategies in relation to the BC for individual functional groups impacted by the trawling and for the simulated period 2008-2016. Few examples out of the 53 functional groups are presented below and these were selected from the aggregated results presented in Figure 17 for the groups of: sand crabs, (Figure 18a), adult tiger prawns (Figure 18b), large sharks (Figure 19a) and large gastropods (Figure 19b). All temporal predictive figures biomass per habitat for all functional groups are presented in the Appendix 13A.

This assessment shows clearly that the spatial effects of the different modelled habitats are highly variable through time and highlights their importance for the predicted biomass for each group (Figure 18, Figure 19). The fact that trawling and its derived impacts occurs mostly on specific fishing habitats was also evident, where the least changes for most groups were found to be in the inshore, the seagrass and the offshore habitats while the tiger-endeavour fishing habitats showed the greatest variation (Figure 18, Figure 19). When changes were observed in habitats with little or no trawling, these variations could be explained by the trophic-induced (or predator-prey) dynamics of the predicted biomass of the various functional groups (e.g. large sharks in Figure 19a).

The performance of the various spatial management strategies was also evident at the temporal and habitat scales for the simulated period. It was clear that for each group and each habitat, and where the trawling effect is strong (on fishing grounds), most scenarios remain similar to the BC, with ±5% variation from the BC as the biggest inter-scenario variation (e.g. sand crabs Figure 18a). In general, the scenarios that showed most differentiation were the LdHf and MPAs (Figure 18, Figure 19). Despite the inter group and scenario variation some habitats showed similar trends –e.g. in Weipa most scenarios differed above and below the BC, in Vanderlins most scenarios were equal or above the BC, and in Mornington north most scenarios were equal or below the BC (Figure 18, Figure 19). These results were overall consistent for most other (not shown) functional groups. These results also show that, despite the scenario, variations increase towards the end of the simulated period, and this is likely to be the result of the recent and expected increase in fishing effort from 2010 onwards (Figure 18, Figure 19).

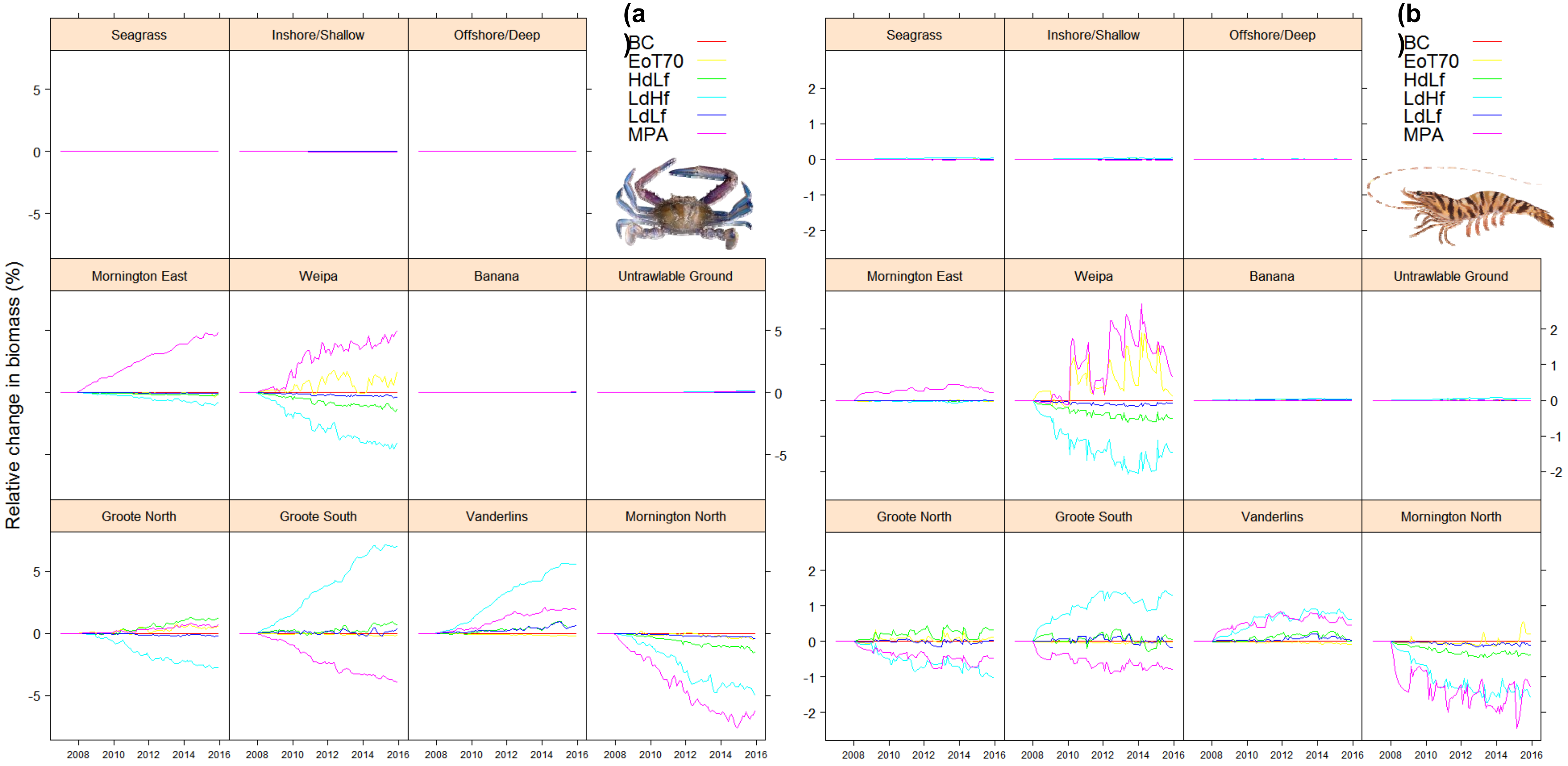


Figure 18 **a-b**. Changes in the predicted relative biomass for all six modelled scenarios in relation to the base case through time for (a.) sand crabs and other large crabs; (b.) adult tiger prawn groups.

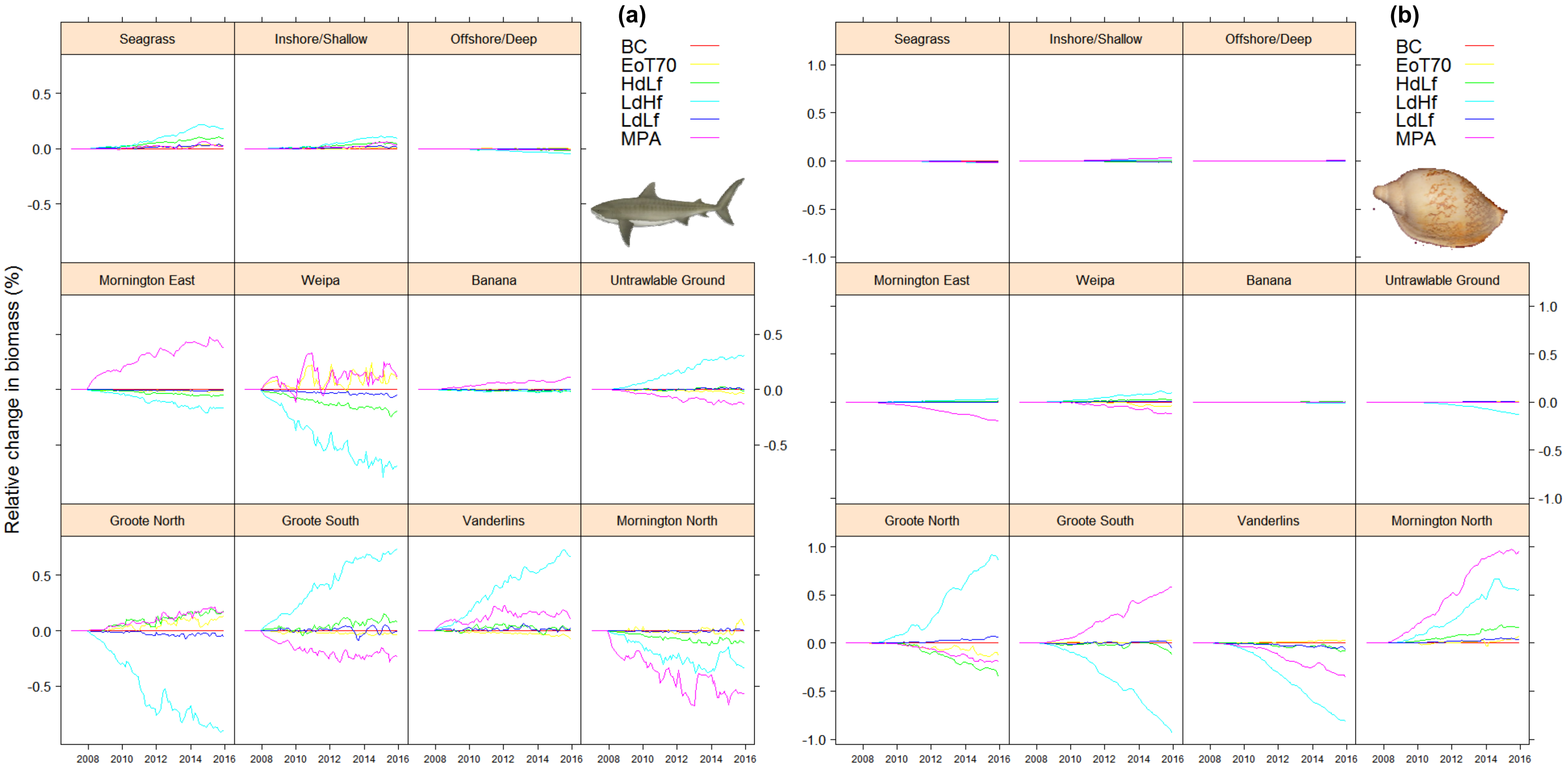


Figure 19 **a-b.** Changes in the predicted relative biomass for all six modelled scenarios in relation to the base case through time for (a.) large shark; and (b.) large gastropod carnivore groups. All habitats are shown in different panels.

7.3.5 ASSESSING THE SPATIAL PREDICTIONS OF THE ECOSYSTEM BIOMASS

The spatial MSE also allows assessing the detailed spatial distribution of the predicted biomass for any functional group across any time step. Figure 20 a-d give illustrative examples, for various functional groups, of their relative biomass for each management scenario, in relation to the BC (status quo) for the end-of-simulation year 2016. All fine-scale spatial predictive biomass for all functional groups are presented in the Appendix 13B. Is important to note that the results depicted below are all at different scales, so the colour prediction for one group is no the same for another –i.e. each figure allows comparison among the scenarios but not across functional groups. As with other biomass evaluations (regional, habitats), these spatial predictions clearly show that positive (towards red) and negative (towards light blue) responses were found for all functional groups in response to the spatial management scenarios (Figure 20 a-d). This is suggesting that some groups are being more, nil or less affected by trawling. These results are also confirming that different groups respond spatially differently among the scenarios. Some groups, such as rays (Figure 20a) and large sharks (Figure 20 b), are predicted to have their biomass increased within spatial closures, whilst the biomass of other groups, such as cephalopods (Figure 20c) and small crustaceans (Figure 20d), is predicted to decrease. We also found that most groups that respond negatively to closures are those that are in general preys of secondary consumers and predatory groups, e.g. cephalopods (Figure 20c) and small crustaceans (Figure 20d). Conversely, most top predatory groups increased their biomass inside and around closures: rays (Figure 20a) and large sharks (Figure 20b) are two examples. These findings are consistent with the expected changes in the underlying food web dynamics that are being affected by trawling. Spatially, these results also confirm that the LdHf and MPAs scenarios showed the most spatial variation in relation the BC (status-quo) across all functional groups, while the EoT70 is the scenario causing the least changes (Figure 20 a-d). In addition, different functional groups respond spatially differently to the regional and habitat variation of the trawling effects, the only changing factor in the simulations. It is interesting to note, that the imposed closures have in some cases a clear spilling-over effect into the surrounding non-closed cells, like in the cases of the rays and large sharks (Figure 20 a and b, respectively). This spilling over is normally found cells subjected to most intensive trawling, e.g. in the fishing hot-spot north of Groote Eyland (Figure 20 a, b). The converse effect was also evident –i.e. when groups decrease in closures they tend to increase in fishing grounds (Figure 20 c, e).

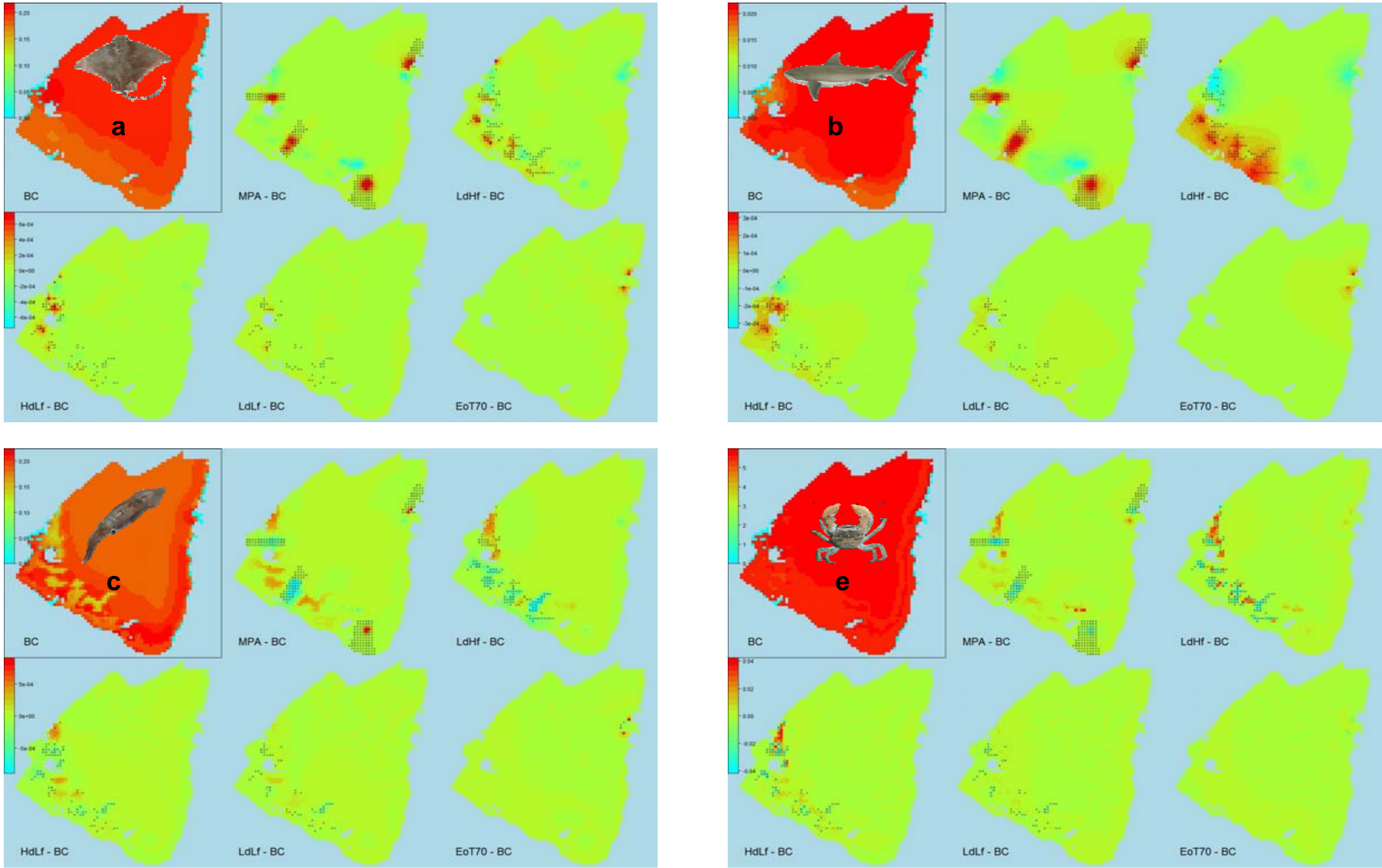


Figure 20**.** Predicted changes to 2016 of the relative biomass of the functional groups; (a) rays, (b), large sharks (c), cephalopods, and (d) and small crustaceans. Different panels show the spatial prediction of each modelled scenario in relation to the base case (biomass - biomass BC). Spatial closures are shown as black crosses.

7.3.6 MEAN TROPHIC LEVEL

The relative performance and trajectories of the mean trophic level of the ecosystem under the various scenarios in relation to the BC (Figure 21) are evaluated in this section. Our results show that the different scenarios in the current simulation period and under current and predicted fishing efforts, induce few and very small changes (of less than 1%) in the mean trophic level of ecosystem under trawling (Figure 21). This is not unexpected for this type of performance measure, since the historical variability (i.e. 1970 to 2010) of the NPF's mean trophic level ranged between 3.77 and 3.38 (see Figure 61, Appendix 9). What is important here, are not the absolute values but their temporal trends and the variability of the performance measure under the different management scenarios.

Despite the small changes, we found pronounced differences among the scenarios (Figure 21). As with other biomass results, the HdLf, LdLf, and EoT70 scenarios showed high variability but closer to the flatline of the BC, although they showed increasing variability through time (Figure 21). The LdHf, and to a lesser extent the HdLf scenarios, show steady and increasing trajectories, while the MPA scenario showed the most inter-annual fluctuations and a decreasing trend in mean trophic level for the last two years of the simulation (Figure 21). These differences can also be explained by the expected food dynamics –i.e. the spatial management scenarios are inducing changes in the benthic ecosystem that in turn is altering its trophic structure and function.

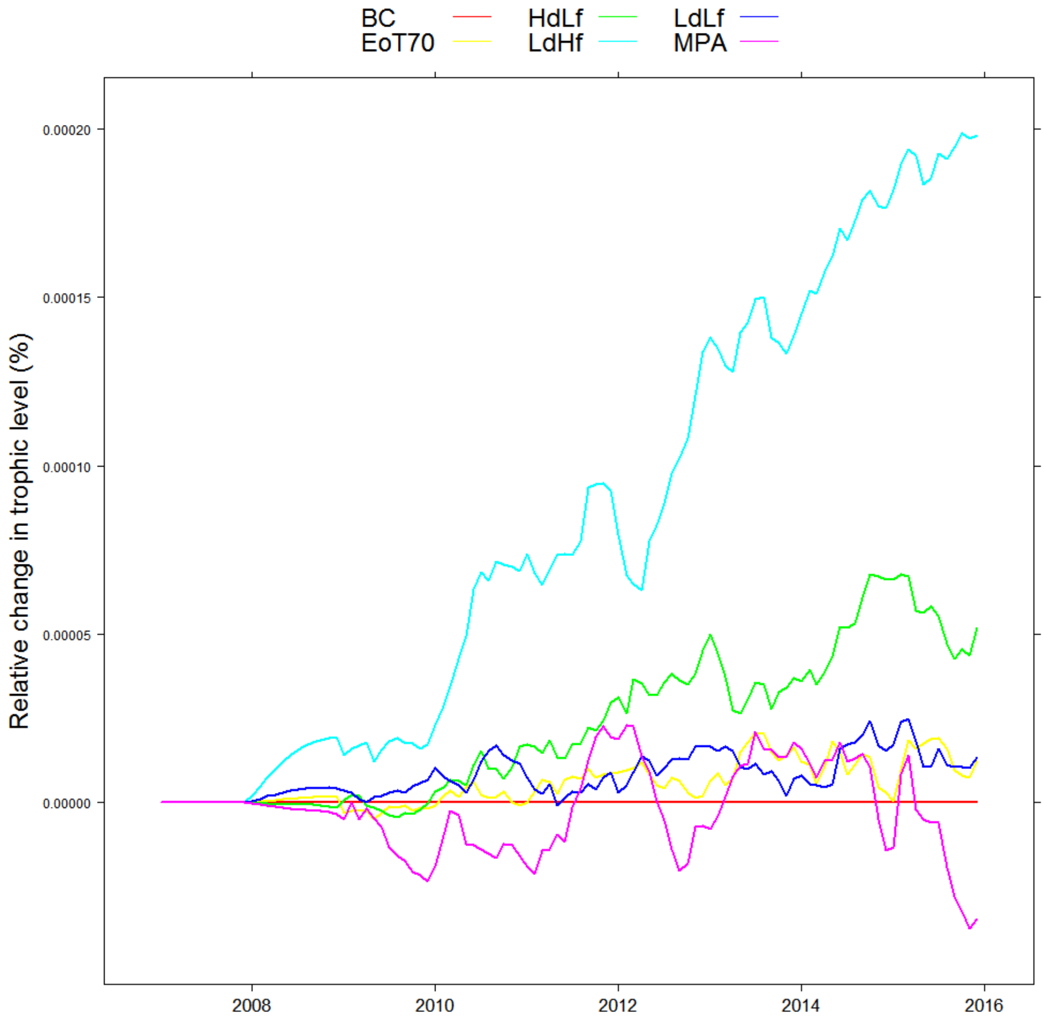


Figure 21. Changes in the mean trophic level of the ecosystem under all spatial management scenarios in relation to the base case (BC status quo).

7.3.7 SUMMARY ASSESSMENT

A summary of the cost-benefits among the simulated spatial management scenarios for a range of likely management objectives and a few selected key functional groups are presented in Table 16. This summary presents the best-and-worst-case scenarios, their pro's-and-con's, and comments regarding the results of the alternative simulated spatial closures, once the fisheries stock and economic objectives have been achieved as advised by the bioeconomic stock assessment. This assessment shows clearly that different scenarios can act as the worst or the best case scenarios depending on the performance measure used and the management objective –i.e. there is no single spatial closure scenario that satisfies or dissatisfies all management objectives (Table 16). Similarly, there was no a single scenario that was consistently the worst, and surprisingly the base case BC (do nothing in the case of the status quo) was not necessarily the worst-case scenario (Table 16).

This assessment also clearly showed that if the management objective is the sustainability of byproducts such as squids and bugs (or cephalopods and lobsters in the model), measured as increasing or stable relative biomass, the MPA scenarios provide the best case because they provide the biggest area protected from fishing (Table 16). If the management objective is to protect or increase the relative biomass of habitat forming species, such as seagrasses and sessile epibenthos (sponges, corals, gorgonians, etc.), then best cases differ depending the group in question. Similar findings occurred when the objective was to increase the biomass of threatened, endangered and protected species (TE&P) such as turtles, dugongs, sawfishes or sea snakes. In this case, the best and worst-case scenarios varied widely but those that foresaw a closure of the greatest area of fishing grounds (LdHf and MPA) tended to perform better (Table 16). This variation in the TE&P responses to closures is likely to be affected by other-than-trawling factors such as local movements, migration, predation and edge effect of closures.

Interestingly, on one hand we found that some spatially coarser measures (e.g. overall biomass on tiger fishing grounds) did not change across all scenarios simply because the variations at larger scales were greater than the impacts and changes induced by trawling (that operates normally at small and local scales). On the other hand, other coarser measures (e.g. the overall number of taxa at risk (from ERA) and the mean trophic level of the whole fisheries ecosystem) did show varied responses across scenarios, with the LdHf being the best case and the BC the worst (Table 16). The mean trophic level showed that the LdHf closure was the best-case scenario while the MPA scenarios were the worst because larger closures tend to favour predatory groups, which in turn, consume groups further down in the trophic web (Table 16).

If spatial management also requires that closures to fishing have the explicit objective of minimising effort displacement and/or the number (or %) of closed fishing grounds, then the smaller closures are the best-case scenarios (EoT70) whilst the contrary is true for the bigger ones –i.e. MPA or LdHf (Table 16)

All these simulated scenarios proved to be highly informative towards the achievement of a range of management objectives whose cost-benefit evaluation is only possible through the use of the proposed spatial MSE delivered in this project.

Lastly, is important to mention that the NPF has a long history of using spatial fishing closures (daylight, seasonal or protected area) in its management –e.g. to protect nursery grounds (such as seagrass beds) to improve biological or economic performance of the fishery (for example, by reducing catches of small prawns or targeting of spawning prawns). This modelling work did not considered such closures because they are mostly inshore and shallow habitats that are no longer impacted by fishing. Additionally, these areas have not been surveyed at all and their biodiversity values status and trends remain unknown.

Table 16. Comparative cost-benefit summary of the best- and-worst-case scenarios for various performance measures against a set of likely management objectives for key functional groups.

Management Objectives	Performance measure	Best scenario	Worst scenario	Pro’s	Con’s	Comments
Sustainability of byproduct						
• Cephalopods	Relative biomass	MPA	LdHf	– higher biomass	– fewer catches	– MPA spatial design will be crucial to achieve NPF’s least effort displacement
• Lobsters (mostly bugs)	“	MPA	LdHf	– higher biomass	– higher displaced effort – higher costs ($)	
Protecting habitat-forming taxa						– Seagrass protected by historical closures
• Seagrasses	Relative biomass	LdLf	HdLf	- higher biomass	– non–adaptive closures	
• Sessile epibenthos	“	LdHf	MPA	- higher biomass	– more displaced effort	– MPA more echinoids and sea turtles that less epibenthos
Reducing “at risk” species						– ERA need to be at smaller spatial scales (habitats or less) otherwise no taxa is detected at risk.
• Overall number	µ and # taxa at risk	LdHf	BC	– effective to remove all taxa at risk	– need to close more fishing grounds	
• Sharks	“	all	BC	– zero risk	– no clear differentiation, need ancillary criteria	
• Rays	“	all	BC	– zero risk		
Protecting TE&P species						
• Turtles	Relative biomass	HdLf, LdLf, EoT70	LdHf	– less effort displaced	– “spotty” and small single closures.	– Closures will require post–hoc consolidation
• Dugong	“	LdHf,	MPA	– higher biomass	– need to close more fishing grounds	– MPA favour dugong predators
• Sawfishes	“	LdHf	none	– higher biomass	– need to close more fishing grounds	– All closures favour sawfishes
• Sea snakes	“	MPA	LdHf	– higher biomass	– need to close more fishing grounds	– Trawl bycatch is the biggest negative effect, so trawling closures provide full protection
Biodiversity						
• Biomass (increase)	Relative biomass	—	—	—	—	– Measure too spatially coarse, no differentiation among scenarios
• Trophic level (increase)	Median trophic level	LdHf	MPA	– higher TL in trend to pristine (1970)	– need to close more fishing grounds	– MPA protects predators that decrease prawns and TE&P
Minimize effort reduction/displacement						
• Tiger fishing grounds	% closed	EoT70	MPA	– minimal closures	– adaptive data needs	– Closures are not focussed on fishing but related to other measures (conservation or risk based)
• Displaced effort	Boat days	EoT70	LdHf		– poor performance in other measures	

In summary, for objectives 3 and 4, the results delivered here are:

- An operational spatially-explicit analytical framework and software that allowed the evaluation of multiple spatial management options (closures) that satisfy multiple objectives.
- A set of systematic and analytical tools that will allow the integration of ecosystem-based information into predictions of spatio-temporal management strategies and the assessment of the likely trade-offs of multiple objectives (e.g. conservation, ecological risk, and fisheries).
- A set of illustrative spatial management strategies, or scenarios, designed to alter the spatial distribution of trawling in order to evaluate its effects on the benthos over the period 2008-2016.
- The scenarios included one fishery dynamic closure and five static conservation and risk assessment closures.
- The closures evaluated the downstream ecosystem impacts of trawling while achieving the NPF's fisheries management targets (stock and economics).
- The spatial management scenarios created plausible fisheries closures of various sizes and shapes that ranged from 5% to 31% of overall fishing grounds.
- The scenarios did not affect fisheries targets and, overall, at the current fishing effort level, had smaller and minor effects in the benthic groups. The changes, though, increased over time.
- The largest fisheries closure scenarios corresponded to ad-hoc conservation MPAs, followed by the ecological risk-based closures, while the smallest were consequent of the EoT70 scenario that created dynamic closures triggered by the 70% threshold of biomass removal.
- The EoT70 scenario directly linked to the NPF's economic target of E_{MEY} showed that the threshold was triggered in very few occasions and associated to specific groups and fisheries habitats.
- All scenarios are shown to have a high spatial variability where most trawling effects on the benthos vary across habitats, showing both positive and negative changes in biomass.
- Most trawling effects varied across regions and this was more evident in the tiger-endeavour fisheries habitats, whereas the changes in the benthic biomass in other regions and habitats could be explained by trophic-induced (or predator-prey) dynamics of the simulated food web.
- On fishing habitats, the risk-based scenario that created closures in areas with high density of taxa at risk and high fishing effort (LdHf) was the spatial management scenario that induced most biomass variations on groups, regions and habitats, despite not being the one generating the most closed areas.
- The adaptive closure Eot70 scenario was consistently the scenario that induced the least biomass variations, largely because it creates the least closures.
- The trade-off in the responses of the various spatial management scenarios on biomass performance measures was largely affected by the interaction between affected groups and the amount of fishing effort removed.
- The different functional groups responded spatially differently to the regional and habitat variation of the trawling effects, the only changing factor in the simulations.
- Closures tended to increase the biomass of high level predators (large sharks, rays, carnivore fishes) with increased predation of prey taxa (small crustaceans, bivalves, echinoids), sometimes decreasing their biomass due to trophic interactions.
- Closures showed clear spilling-over effects into the surrounding non-closed cells –i.e. the positive and negative effects are transferred to the nearby cells subjected or not to fishing.

- Small changes occurred in the mean trophic level of the modelled ecosystem over the simulated period. The changes, though, increased over time.
- There were clear and large differences among the spatial management scenarios in the trajectories and variability of mean trophic level.
- The LdHf, and, to a lesser extent, the HdLf scenarios showed steady and increasing trajectories, while the MPA scenario showed a decreasing trend in the mean trophic level.
- Overall, although smaller changes in ecosystem biomass and trophic level were detected throughout the various simulations, the spatial management scenarios induced changes in the benthic ecosystem impacted by trawling.
- Although the current fishing effort of the NPF is low, it is clear that trawling is altering the trophic structure and functioning of the benthic ecosystem and spatial management could provide a method for mitigating the undesirable ecological effects of trawling.

8 BENEFITS AND ADOPTION

The results and outputs of this project contribute to the three main sectors:

8.1 MANAGERS (FISHERIES AND CONSERVATION).

We believe that managers will be those to benefit the most from the project's outputs because the project:

- Provides critical information on the biota and ecological processes towards a better understanding and management of the impacts of trawling on the benthic habitats and biota of the Northern Prawn Fishery.
- Provides further spatially-explicit information on the distribution and abundance of fish and invertebrates that, for example, will enhance the ecological risk assessment for the non-target taxa.
- Provides operational tools to assess the further ecological consequences of the ongoing fisheries management that focus largely on stock, economics and risk.
- Contributes to the development and proposal of key ecological reference points and indicators to assess the spatial impacts of, and changes in, alternative spatial management regimes.
- Provides evidence-based advice for the development and future implementation of spatial management policies.
- Allows managers to evaluate likely consequences and trade-offs of alternative spatial management provisions ahead of implementation.
- Provides the mangers with an integrated set of spatially-explicit tools and their derived advice for operational implementation of ecosystem-based conservation management.
- Enables managers to contrast and evaluate spatial management provisions generated by sometimes-perceived competing management objectives (e.g. conservation MPAs and fisheries closures).
- Provides an integrated analytical framework that effectively bridges between the management of target stocks and their ecological environment, yielding the analytical support for decision making and spatial planning frameworks for management across fisheries and conservation objectives.
- Provide support towards the conservation and regionalisation processes intended for the northern region, in particular the National Representative System of Marine Protected Areas to protect northern biological diversity and maintain essential ecological processes and life-support systems.
- Contributes methods to assess impacts and develop spatial conservation measures on the status and trends of habitats (e.g. seagrasses) and endangered and protected marine taxa (sawfishes, sea snakes, etc.).
- Provides further data and tools that will allow the development of ecosystem monitoring programs as well as baseline assessments for benthic and some basic water column variables (and their characterisation) as well as for habitats and biological communities of the Gulf of Carpentaria.

8.2 FISHING INDUSTRY

The fishing industry will benefit from this project for a number of reasons:

- Because we can now spatially predict the likely effects of trawling, the industry will have a better understanding of environmental policies designed for the management of perceived and effective impacts of fishing.
- Additional ecosystem-based tools will be available for integration onto regular environmental assessments required by DEWHA.

- A formal and science-based link between the current bieconomic target management and the environmental and conservation management will be available.
- Better visualisation of the spatial context of fisheries ecosystems will be achievable, and this may, in turn, guide improved spatial planning of fishing and the associated likely environmental costs and investments.

8.3 SCIENCE COMMUNITY AND PUBLIC

With respect to the scientific community and the public, the project provides:

- Improved knowledge and understanding of key bio-physical, biogeochemical and ecological drivers of biodiversity in tropical marine ecosystems, in particular northern Australia’s Gulf of Carpentaria.
- Improved and applied observational tools and technology for the identification and evaluation of environmental and ecosystem characterization.
- Innovative use of analytical and modelling tools to describe and predict ecosystem variability and changes.
- Sound baseline information on which ecosystem management could be assessed and trends detected, in particular regarding expected changes due to climate variability and changes.
- Ecosystem characterization and baseline information which could be used to assess environmental impacts of both natural and anthropogenic sources, and will include the other industries on land and coastal, estuarine and riverine systems as they develop in northern Australia.
- A potential model for assessing fishing impacts in other regions of northern Australia through an integrated approach, as well as the operative tools to assess them.

9 FURTHER DEVELOPMENT

We believe that the further developments of this project's results and outputs can be aggregated under three main categories.

9.1 TECHNICAL DEVELOPMENT

- Develop a specific proposal for further refinement of the spatial allocation fishing effort.
- Improve fine-scale effort allocation and enhance feedback between coarse-scale and fine scale allocation.
- Update the framework to the new versions of software and models being developed recently (e.g. Ecospace, ERA level 3).
- Transfer to the new spatial visualisation MSE framework developed for C2C-SEQ to deal with massive results and outputs.
- Develop and add further stochastic processes to some of the models (e.g. ERA, EoT, EwE).
- Develop and implement further the use of simulated survey and EoT and species distribution models for assessing TE&P species.
- Include ERA in the loop as an assessment model and not as a performance indicator.

9.2 COLLABORATIVE DEVELOPMENT

This project expects to:

- Expand and improve data sources and provide the spatially explicit information basis to have several iterations of improvement, using engagement with industry and managers (e.g. RAG, MAC).
- Agree and define specific spatial management and industry-driven spatial scenarios (e.g. effects of closures on: ITQs, TE&Ps, risk management, etc.) and investigate how these respond to the agreed multi user needs (managers, industry, others).
- Agree with managers, industry and environmental groups as to the overall process for likely uptake.
- Identify the needs of managers and stakeholders for further (or not) development and application.
- Inform and communicate across managing agencies (e.g. DEWAH, DAFF), to demonstrate the utility of the framework beyond fisheries management.
- Develop further alternatives to conservation-centric scenarios such alternative configuration of MPAs and closures for TE&P taxa.

9.3 COMMUNICATION AND DISSEMINATION

- Present project's results, recommendations and conclusions to AFMA and agree and design the overall communication process and steps for likely uptake and/or uses.
- Present the results to NPF's management RAG and MAC and receive additional feedback for further (or not) adjustments, uses and development.
- Present the project's results to DEWHA and conservation organisations to demonstrate the capabilities of spatial MSE to assess beyond fisheries objectives.
- Participate in science-based events such as congresses and symposia, and prepare and submit scientific manuscripts to peer-reviewed journals.

10 PLANNED OUTCOMES

In this section we are providing a synthesis of the project's results against the planned outcomes as stated in the original proposal (inserted in numerals and text in italics below). It is important to note that these outcomes were planned during the years leading to this project, largely between 2004 and 2005. Many things have changed since then, particularly in the industry and management but the relevance of the majority of the planned project's outcomes still remain. Some expected outcomes are already in place (e.g. risk assessments) despite this project, however the project adapted and adjusted its results to represent the current fisheries and management situation. Below we describe how this project has and will contribute towards these outcomes.

1. *AFMA, the industry, advisory groups and science agencies will be in a position to complete the collaborative development of a spatially explicit management framework. The prerequisite technical communication between relevant projects (stock assessment, economics, bycatch, and this project) will have occurred during the course of this project, and will have been facilitated by the Steering Committee.*

The spatial MSE delivered here (see section 7.3 and Appendix 10) provides, for the first time, an operational-analytical framework. This will allow managers and the industry in particular, to evaluate alternative spatial management options and their consequences when achieving simultaneously the fisheries economic and stock objectives as well as other conservation and environmental management ones. This project also establishes a clear understanding of the impacts of trawling on the benthic ecosystems of the GoC and how that information is or should be used in spatial management. The project also managed to build an effective technical communication between the various other existing science-based supports to the management of the NPF (economics, risk and stock assessments, bycatch management). This resulted in an integration of knowledge and a cost-efficient use of pre-existing tools towards the provision of one single spatial MSE framework for the NPF. The industry and management were kept directly involved and informed on the project's progresses and, in conjunction with AFMA, it is foreseen that managers will communicate and disseminate results to the NPF's RAG and MAC.

2. *The NPF will have a capacity for developing science-based objectives, targets, reference points and performance indicators for management of the impacts of trawling on benthic ecosystems.*

This project has provided a range of quantitative and numerical information. This will allow the NPF to develop and use a range of indicators, performance measures, and set model-based targets for the spatial monitoring and evaluation of the impacts of trawling. In particular, the evaluation of the extent, nature, and variability of the impacts of trawling on the benthic ecosystem and its processes confirm previous findings that indicate that trawling at the current NPF's fishing effort and GoC spatial scales do not in themselves affect biodiversity. There are still however some impacts at the small spatial scales (local) and at particular taxon/species levels that should be addressed in part by the current bycatch action plan and the ecological risk management responses for those taxa at risk. In short, this project provides the industry and managers with the tools and ability to respond to increasing environmental and conservation management policies, as well with an operative integration of all existing tools into a formal spatially-explicit ecosystem-based management framework.

3. *Managers of the NPF will be better able to delineate effective management stocks for use in spatial management. The knowledge and data from this project will include maps of prawn habitats and maps of environmental regions in the south-western Gulf of Carpentaria.*

The project made use of the existing prawn stock region boundaries already in use by the bioeconomic stock assessment (Dichmont et al. 2006, 2008, 2010), so there was no need to develop new ones as expected originally from this outcome set out in 2004-05. However, this project, in conjunction with a parallel and complementary spatial data project done for the northern region of Australia (Rochester et al. 2007), composed an extensive digital spatial library available at CMAR's http://www.marine.csiro.au/datacentre/ext_docs/mbp_north/characterisation/index.html whose output maps are included in Appendix 12. Therefore, this project did collate and integrate all spatially-explicit data available for the NPF providing managers with a detailed portrait and characterisation of the fisheries ecosystem under management, its state, composition and spatial variability. We believe that this information basis, in addition the spatial MSE and the collection of analytical tools, provides this fishery and the region with a unique and particularly strong science-based foundation needed for the management of its natural resources, its conservation and overall environmental sustainability.

4. *Risk assessment for benthic habitats, invertebrates and fish in the NPF, needed for sustainability assessment, will be enhanced. Quantitative estimates of the responses of benthic habitats and species will be provided by analysis of the Gulf survey samples.*

This project brought in the data, models and predictions of the current NPF's ecological risk assessment (ERA) as part of the core components to be integrated into the spatial MSE. The ERA models were used and enhanced, providing the managers with: (i) an independent spatial prediction for species at risk (species distribution model, see Appendix 8), (ii) the assessment itself, which is now part of integrated framework that bridges across various management tools, and (iii) the ability to use the ERA to create and define various spatial management scenarios or risk based closures –i.e. the scenarios of Low density High fishing effort (LdHf), the High density Low fishing effort (HdLf), and the Low density Low fishing effort (LdLf). The use of these ERA-based spatial closures has taken the current approach of ERA and how to inform the resulting AFMA ecological risk management plans (ERM), one step forward. Thus, the project has not only enhanced existing risk-based tools applied to biota, but has also integrated them into a holistic framework that provides this industry and the managers with a state-of-the-art operational spatial assessing tool for biota at risk.

The various estimates of the biological responses to the trawling impacts as derived from the performance measures, indicators and metrics (e.g. see section 10.3.9) are all quantitatively based, new and established in this project. This information is now readily available and will be provided to the industry and managers for use into the upcoming overall environmental assessment process.

5. *Sampling of benthic habitats and communities will be more cost-effective in the future, because of estimates of the spatial variation in benthic habitats and communities provided by this project.*

This outcome was identified hoping that, concurrently to the project and in the future, there be more biological and ecosystem field-based research in northern Australia that will impact and likely benefit the NPF environmental management. Towards that, we contributed substantially with the integration of most existing spatially-explicit biophysical datasets for the region and in particular the GoC. These data were used directly to plan and execute the SS 2005/03 and SS 2005/04 surveys (see Appendix 11). In addition, and based on the work developed by Rochester et al. (2007) and our own datasets, this project created a range of maps for habitats and its communities (see Appendices 6 and 9) and the ecosystem biophysical characterisation as well as detailed maps of the spatial distribution of the main functional groups of the fisheries ecosystem of the GoC (see Appendix 12). This range of spatial outcomes will, again, put managers, the industry and researchers in particular, in the best informed position for the planning and design of sampling, surveyor and, more importantly, monitoring and evaluation programs.

An additional outcome is that these spatial maps and data are, and will, be the core of the information basis for the bioregionalisation of the northern region (DEWHA 2008) and the design and placement of conservation MPAs within the areas of further assessment within the region (DEWHA 2009). Thus, delivery of this project regarding the required spatial data and evaluation tools is contributing substantially towards anticipating these conservation-based processes, if fisheries mangers and industry decide to do so.

11 CONCLUSIONS

In this section we present the main conclusions summarized against the established project's objectives.

11.1 OBJECTIVE 1

- The effects of trawling on the benthic biodiversity were not clearly demonstrated at the whole-community or whole fisheries ground levels. The identified impacts seem to operate at smaller local habitat and taxa/species levels.
- Overall, there was more variation in benthic biodiversity measures across the different regions of the Gulf of Carpentaria as well as within day and night variations than among the different trawling intensities.
- There are species, taxa and functional groups that are indeed negatively affected by high trawling intensities (e.g. sharks, rays, sponges, bryozoans, and gorgonians).
- There are also as many others biotic groups that seem to be more abundant or prevalent in areas with high trawling (e.g. dollar and pony fishes, some crabs and bivalves).
- At the current levels of trawling and fishing effort, the management concerns should focus on those taxa that have been assessed to be at risk and that are specially protected, such as endangered, threatened and protected species (TE&P).

11.2 OBJECTIVE 2

- For the surveyed fishing grounds, during the sampling period and within the constrains imposed by the sampling, the differences among water column and bottom sediment environmental variables and their associated ecosystem processes were largely explained by regional and spatial variability.
- The variability of water column, sedimentary habitats and stable isotopes variables have no or little relation to trawling intensities.
- Trawling effects appear not affect biophysical processes, but could be affecting the small-scale trophic processes at the species and taxa levels.
- A simulation of the food web dynamics s over the past 40 years of trawling showed a clear ecosystem impact by lowering the mean trophic levels of the catches (TL). That effect largely occurred during the fishery expansion in 70s to mid 80s.
- When fishing effort was reduced (starting in the mid to late 80s) the TL increased steadily to predicted values in 2010 close to the ones estimated for the mid 70s.
- As concluded in Objective 1, the simulated historical effects of trawling showed both positive and negative effects on the relative biomass of the 53 functional groups of the simulated benthic ecosystem.
- The evaluation of the recent 2005-2010 reductions of fishing showed that current impacts of fishing on the fishing grounds have very small effects in the overall biomass of all affected biotic groups.
- The food web simulations suggest that indeed the ecosystem in the Gulf of Carpentaria is still influenced by trawling, but due to the drastic reduction of fishing (from 286 vessels in 1981 to

52 vessels in 2009) these impacts on biomass removal and trophic levels have been largely reduced but will increase as fishing effort increases.

11.3 OBJECTIVES 3 AND 4

- The existing information basis and analytical tools and models currently used to support the management of the NPF were successfully integrated into a single operational spatially-explicit management framework –i.e. a spatial management strategy evaluation simulation approach (spatial MSE).
- This spatial MSE demonstrates that can be effectively used to evaluate the trade-offs of multiple and sometimes perceived conflicting fisheries and conservation management objectives.
- The delivered spatial MSE is able to evaluate simultaneously the ecosystem consequence of multiple spatial management fisheries closures and conservation MPAs scenarios while achieving the NPF's fisheries management targets (stock and economics).
- The effects of the spatial management scenarios did not affect fisheries targets and overall, and at the current fishing effort levels, the trawling-induced changes are small and with little net variation across the range of performance measures and metrics, evidencing to have smaller and minor changes in the benthic ecosystems. These changes though are in an increasing trajectory as fishing effort increases over time.
- All spatial management scenarios tested showed to have a high spatial variability where most trawling effects on the benthos were found on the tiger-endeavour fishing-affected habitats, but also varied regionally across habitats, showing both positive and negative changes.
- Changes in the simulated spatial management scenario not affected by trawling can be only explained by trophically-induced (or predator-prey) dynamics of the simulated food web.
- Depending on the management objectives, the various simulated spatial management scenarios (largely fisheries closures) can act as best, nil and worst case scenarios for any particular group, performance measure or evaluation metric.
- In general and in relation the base case BC for example, the scenarios that have the greatest spatial closures -i.e. the conservation MPA and risk-based closures the closes fishing hotspots (LdHf), have the best and worst biological responses across all measures and metrics.
- Comparing across simulated scenarios, the more the fishing closures the more responses across the whole benthic ecosystem, while the scenarios that have the least closures will have no or relatively little responses in relation to the base case BC (status quo).
- The coarser spatial scale of the evaluated performance measure or metric, like in the case of overall relative biomass for the whole tiger-endeavour fishing grounds, the less informative and sensitive to changes. This demonstrates that they seem to be too spatially coarse to detect any difference across alternative spatial management scenarios.
- No single simulated spatial management scenario satisfied all or most management objectives, evidencing that if any fishing closures (or MPA network) are to be established in the NPF, these need to be thoroughly assessed and tested against multiple criteria, performance measures and metrics.
- The spatial MSE delivered by this project can perform the above-mentioned tasks, incorporating industry, managers and stakeholders' inputs to the desirable performance measures and in developing the likely spatial management scenarios.

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APPENDIX 1 INTELLECTUAL PROPERTY

The intellectual property of this project resides largely on the information, data and tools developed during the project's implementation. This include all digital records of all historical CSIRO data holdings for the northern region of Australia and relevant to the NPF, including the data generated in the SS 2005/03 – 04 research cruises (see Appendix 11). The metadata for these data seta are publically available at:

SS 2005/03

http://www.marine.csiro.au/marq/edd_search.search_choice?tFre=ss+03/2005&ch1=freetext&cSub=>>

SS 2005/04

http://www.marine.csiro.au/marq/edd_search.search_choice?tFre=ss+04/2005&ch1=freetext&cSub=>>

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APPENDIX 3 RELATIONSHIPS BETWEEN BENTHIC BIOTA AND TRAWLING INTENSITY

W.A. Rochester, R.H. Bustamante, T. Wassenberg, & G. Fry

3.1 ABSTRACT

The target sites for voyage SS 03/2005 were located and visited in a factorial design with three factors: region, day/night and trawling. Biota at the sites were sampled with a prawn trawl, an epibenthic sled and a box core. Variation in biodiversity among the survey strata was described and tested with univariate analysis of ecological indicators and multivariate analysis of catch composition. The ecological indicators were abundance and diversity indices. The multivariate methods were principal components analysis, redundancy analysis, variation partitioning and nonparametric multivariate analysis of variance. Analyses were performed on three sets of data: fish from the prawn trawl, invertebrates from the epibenthic sled, and infauna from the box core.

The relationships between catch species composition and the survey design strata varied among the three analysis datasets. The composition of fish from prawn trawl samples was related to region and day/night, and was weakly related to trawling. The composition of invertebrates from epibenthic sled samples was related to region and trawling, but not to day/night. The composition of infauna from box core samples was related to region and weakly to trawling, but not to day/night.

Although empirical relationships between catch composition and trawling intensity do not in themselves indicate that trawling affects biodiversity, the analysis results are consistent with partial effects of both trawling and habitat. The association of fragile animals such as bryozoans and brittle stars with low trawling is consistent with a trawling effect. The association of reef-associated fish with low trawling and prawn predators with high trawling is consistent with a habitat effect.

3.2 INTRODUCTION

Evaluating the ecosystem effects of demersal trawling has attracted substantial research efforts in the past decades (Jennings and Kaiser. 1998, Hall 1999, Gislason et al. 2000, Collie et al. 2000, Thrush and Dayton 2002, Kaiser et al. 2006, Watling and Norse 2008, Trush and Dayton 2010, Thurstan et al. 2010). These efforts have been largely biased towards studies in the northern hemisphere and temperate and deep water ecosystems, where the impacts have been demonstrated to be substantial (e.g. Thrush et al. 1998, Cryer et al. 2002, Heath 2005, Heifetz et al 2009). The results of such studies have strongly influenced the perceptions and the management responses for the trawling impacts.

Similar studies elsewhere however have not occurred for tropical and sub-tropical soft-sediment systems, normally associated with fishing for Peaneids and crustaceans species. The comparatively few studies conducted in tropical and sub-tropical ecosystems have been largely focus on bycatch assessments (e.g. Harris and Poiner 1990, Hendrickson and Griffin 1993; Brewer et al. 1998; Ortiz 1998; Robins et al. 1999; Stobutzki et al. 2001a,b). Foster (2009) made the interesting point that most of these impact studies are concentrated in developed countries, creating a strong knowledge bias to simile ecosystems in developing nations, where most trawling is conducted in soft-sediments habitats. Australia has led the assessment of ecosystem impacts of fishing in particular the impacts of trawling, with a range of studies in tropical sub-tropical (e.g. Pitcher et al. 2000, Sobutski et al. 2001, Wassenberg eta al 2002, Sumpton et al. 2005, Courtney et al. 2006, Pitcher et al. 2009), temperate (e.g. Bax and Williams 2001, Tanner 2003, Williams et al. 2006, Svane et al. 2009) and deep ocean ecosystems (e.g. Koslow et al. 2001, Althaus et al. 2009, Williams et al. 2010). The findings of such studies has been varied, ranging from irreversible (Williams 2010), to strong (Svane et al 2009), and to less obvious and in cases undetectable impacts (Burridge et al. 2005). Despite these wide variations, trawling does have local and specific impacts, particularly when the fishing grounds overlap with vulnerable biota and the impact assessments are done at the appropriate temporal and spatial scales (Pitcher et al. 2009). The significance of these works is that they have set scientific basis and ecological knowledge needed for this study.

The target sites for voyage SS 03/2005 were located and visited in a factorial design with three factors: region, day/night and trawling (Table 17). At each target site we attempted to complete a prawn trawl sample, an epibenthic sled sample, a CTD cast and a few box cores, and we generally managed that. This design enabled us to study the relationships between observations made on the survey and the survey stratification factors and the interactions among these factors. In this analysis we examine the assemblages of organisms found in the prawn trawl, epibenthic sled and box core samples.

Table 17. Target sites visited as part of the natural experiment component of RV *Southern Surveyor* voyage SS 03/2005

	Trawling						
	Low		Medium		High		
Region	Day	Night	Day	Night	Day	Night	Total
Groote	7	7	7	7	7	7	42
Vanderlins	7	7	5	7	7	7	40
Mornington	7	7	7	7	7	7	42
Total	21	21	19	21	21	21	124

3.3 METHODS

Variation in biodiversity among the survey strata was described and tested with univariate analysis of ecological indicators and multivariate analysis of catch composition. The ecological indicators were abundance and diversity indices. The multivariate methods were principal components analysis (PCA), redundancy analysis (RDA), variation partitioning and nonparametric multivariate analysis of variance (NPMANOVA).

Analyses were performed on three sets of data: fish from the prawn trawl, invertebrates from the epibenthic sled, and infauna from the box core. For fish and infauna, catches were represented by count densities (individuals per hectare). For epibenthic invertebrates, catches were represented by weight densities (kilograms per hectare), which are more appropriate for colonial animals such as ascidians and bryozoans.

The primary datasets required for the analyses were sample by species matrices of the catches. The samples included in the matrices were those that were included in the factorial design part of the survey and that had no data quality problems (most samples). The choice of taxa to include was more complicated because different species were identified to different taxonomic levels depending on available taxonomic skills and difficulty of identification. For example, if we have a genus for which we have identified some species to species and some only to genus, then we need to choose between aggregating the species level records up to genus level or discarding the genus level records. For this analysis we used the following rule: if the number of deployment by taxon records for a taxonomic group was more than 25% of those for the group and lower taxa within the group, then we aggregated the records for the lower taxa, otherwise we discarded the group level records and retained those for the lower taxa within the group. The rule was applied to all taxa, starting with the kingdoms and working up the tree to species.

For the multivariate analysis (but not the univariate indicator analysis), a second taxon filter was applied after the sample by species matrix (or deployment by taxon matrix in this case) was compiled for each sampling device. To exclude infrequently observed taxa, we dropped those present in fewer than 5% of deployments from the matrices.

The ecological indicators were total catch, species richness and Shannon's equitability (equal to Shannon's diversity divided by the Shannon's diversity of an assemblage with the same number of species but with equal abundances for all species). For total catch and equitability, the abundance measure was count for the trawl and box core and weight for the epibenthic sled.

For PCA, RDA, variation partitioning and NPMANOVA, catches were transformed firstly with the fourth-root transformation to reduce the skewness in the catch distributions, and secondly with the Hellinger transformation to enable the use of PCA, RDA and other methods based on Euclidean distance (Legendre and Gallagher 2001). The Hellinger transformation enables the use of Euclidean methods because Euclidean distances calculated from Hellinger-transformed data equal Hellinger distances calculated from the original data (Legendre and Gallagher 2001). The Hellinger distance performs well with community composition data. Compared with the chi-square distance, which is used in correspondence analysis, it places less weight on rare species (Legendre and Gallagher 2001). Reducing the weight on rare species is desirable here because rare species can destabilise analyses when observation error is relatively high. A number of other distance measures, including the chi-square distance, also have corresponding transformations that effectively enable them to be used with Euclidean methods (Legendre and Gallagher 2001).

To visually examine whether the survey strata were related to the general pattern of variation in catch assemblage, we summarised the assemblage variation with PCA plots and labelled the PCA plots with the strata (region, day/night and trawling). The analysis was performed with the **RDA** function of the **VEGAN** library of the R computer program (R Development Core Team 2007).

To examine the amount and pattern of variation of catch assemblage that was related to the survey strata, we used RDA with region, day/night and trawling level as constraining variables. To specifically examine variation in catch assemblage among the trawling levels we used RDA constrained on trawling level and conditioned on region and day/night. The analyses were performed with the **RDA** function of the **VEGAN** R library.

To test whether catch assemblage varied significantly among the survey strata we performed an NPMANOVA of catch assemblage on region, day/night and trawling level. The analysis was performed with the **RDA** and **ANOVA.CCA** functions of the **VEGAN** R library. To estimate the proportion of catch assemblage variation explained by each variable, we used variation partitioning analysis with the **VARPART** function of the **VEGAN** R library.

3.4 RESULTS

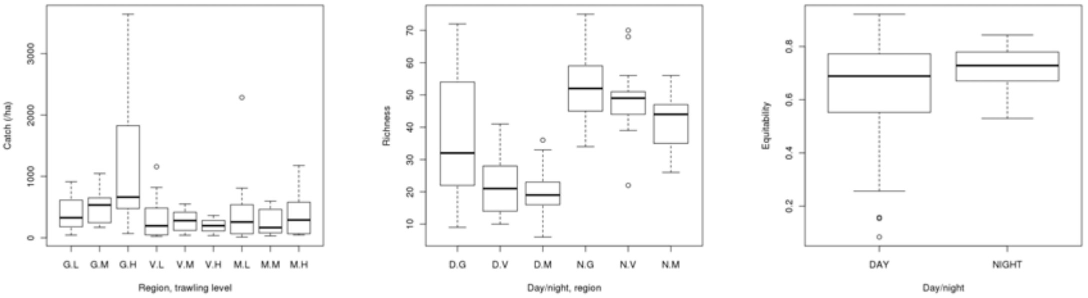
3.4.1 FISH FROM PRAWN TRAWL SAMPLES

For fish from prawn trawl samples, there was no consistent relationship between any of the ecological indicators and trawling (Figure 22. Figure 23, Table 18). Total catch varied among trawling levels, but the pattern of variation varied among the regions. Total catch and species richness varied among regions, and species richness and equitability varied between day and night (Figure 22. Figure 23, Table 18).

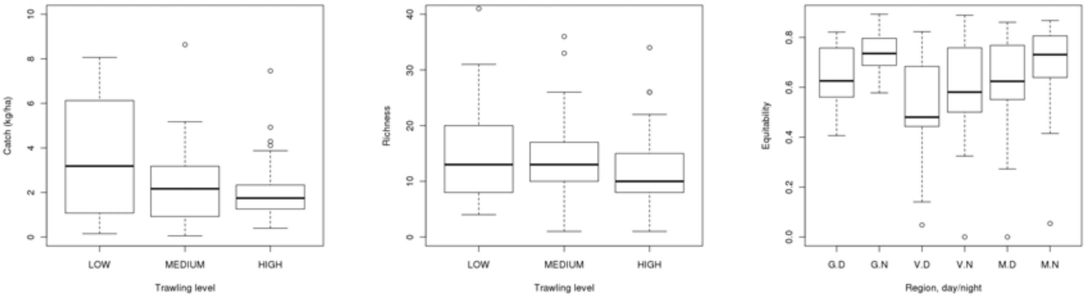
For multivariate analysis, the sample by species matrix for fish from prawn trawl samples included 123 deployments and 128 taxa (127 species and 1 genus).

Fish catch assemblage varied among regions and between day and night, but only weakly among trawling levels. Regions and day/night were clearly segregated by the PCA ordination, but trawling strata were interspersed on at least the first five axes (Figure 24a–c). In the RDA constrained by region, day/night and trawling, the first three axes were dominated by day/night and region (e.g. Figure 24d). Trawling levels were not strongly segregated until the fourth axis. In the NPMANOVA, fish catch assemblage varied significantly among regions, day/night and trawling levels, but note that the actual variation explained by trawling was very small (Table 19).

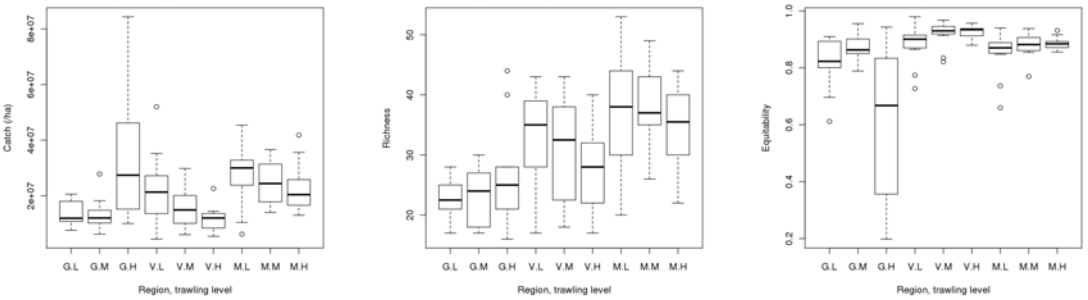
In the RDA constrained by trawling and conditioned on region and day/night, the first axis separated the low, medium and high trawling levels in that order, although the medium and high levels were close together. The second axis separated the medium and high trawling levels. Fishes associated with medium to high trawling (first axis) included the blue-spotted trevally (*Caranx bucculentus*) (Table 20), an important predator of commercial penaeid prawns (Brewer et al. 1994). Those associated with low trawling included the longfin silverbiddy (*Pentaprion longimanus*), which is a common NPF bycatch species but also generally common throughout the GOC (Blaber et al. 1994), and reef-associated species such as the gulf damsel (*Pristotis obtusirostris*), the onion trevally (*Carangoides caeruleopinnatus*) and the brownstripe snapper (*Lutjanus vitta*)). If the relationship between species composition and trawling was causal, then we might expect that the species scores would be correlated with the trawl sustainability rankings estimated by Stobutzki et al. (2001). However, there was no such trend for the top and bottom five ranked species of the first or second axes.



(a) Trawl, fish, catch (b) Trawl, fish, richness (c) Trawl, fish, equitability



(d) Sled, invert., catch (e) Sled, invert., richness (f) Sled, invert., equitability



(g) Core, infauna, catch (h) Core, infauna, richness (i) Core, infauna, equitability

Figure 22. Figure 23 Box-and-whisker plots of catch, species richness and equitability against survey strata. The boxes indicate the 25th and 75th percentiles. The whiskers extend at most 1.5 times the interquartile ranges from the boxes. The strata included in each plot are generally those to which the indicator was significantly related (). Catch and equitability were calculated with count data for fish and infauna (a–c, g–i) and with weight data for invertebrates (d–f). The survey strata are region (G, Groote; V, Vanderlins; M, Mornington), day/night (D, day; N, night) and trawling (L, low; M, medium; H, high).

Table 18. ANOVA tests for relationships between three indicators (total catch, species richness and equitability) and the survey strata (T, trawling; R, region; D, day/night)

Fish from prawn trawl samples

```
           Total catch (count density)                                Richness                                  Equitability
----------------------------------------------------  ----------------------------------------  ----------------------------------------
            Df   Sum Sq  Mean Sq F value  Pr(>F)        Df  Sum Sq Mean Sq F value  Pr(>F)        Df  Sum Sq Mean Sq F value  Pr(>F)
T            2 1.16e+06 5.81e+05  2.8358  0.0632 .       2   133.5    66.8  0.5141  0.5996         2 0.02938 0.01469  0.6657  0.5161
R            2 4.15e+06 2.08e+06 10.1422  0.0001 ***     2  4237.8  2118.9 16.3153  0.0000 ***     2 0.09256 0.04628  2.0970  0.1279
D            1 5.08e+05 5.08e+05  2.4787  0.1184         1 12716.6 12716.6 97.9170  0.0000 ***     1 0.16713 0.16713  7.5731  0.0070 **
T:R          4 3.47e+06 8.67e+05  4.2336  0.0032 **      4   259.3    64.8  0.4991  0.7364         4 0.02423 0.00606  0.2744  0.8939
T:D          2 6.80e+05 3.40e+05  1.6596  0.1952         2   425.9   212.9  1.6396  0.1990         2 0.03038 0.01519  0.6883  0.5047
R:D          2 4.38e+05 2.19e+05  1.0704  0.3466         2   729.4   364.7  2.8080  0.0649 .       2 0.01428 0.00714  0.3236  0.7242
T:R:D        4 1.03e+06 2.56e+05  1.2518  0.2938         4   211.0    52.7  0.4062  0.8038         4 0.00588 0.00147  0.0667  0.9917
Residuals  105 2.15e+07 2.05e+05                       105 13636.5   129.9                       105 2.31721 0.02207
```

Invertebrates from epibenthic sled samples

```
           Total catch (weight density)                               Richness                                  Equitability
----------------------------------------------------  ----------------------------------------  ----------------------------------------
            Df   Sum Sq  Mean Sq F value  Pr(>F)        Df  Sum Sq Mean Sq F value  Pr(>F)        Df  Sum Sq Mean Sq F value  Pr(>F)
T            2     3081     1540  3.3191  0.0400 *       2   407.9   204.0  1.7876  0.1724         2  0.0374  0.0187  0.6101  0.5452
R            2     1963      981  2.1145  0.1258         2    74.0    37.0  0.3241  0.7239         2  0.3610  0.1805  5.8906  0.0038 **
D            1      101      101  0.2179  0.6416         1    84.4    84.4  0.7398  0.3917         1  0.1860  0.1860  6.0713  0.0154 *
T:R          4     2316      579  1.2476  0.2954         4   430.5   107.6  0.9433  0.4420         4  0.2081  0.0520  1.6976  0.1561
T:D          2     1247      623  1.3435  0.2654         2   368.3   184.2  1.6141  0.2040         2  0.0109  0.0054  0.1772  0.8379
R:D          2      120       60  0.1293  0.8788         2   108.0    54.0  0.4731  0.6244         2  0.0025  0.0012  0.0401  0.9607
T:R:D        4      737      184  0.3969  0.8105         4   276.4    69.1  0.6057  0.6594         4  0.1562  0.0390  1.2742  0.2848
Residuals  105    48728      464                       105 11980.0   114.1                       105  3.2173  0.0306
```

Infauna from box core samples

```
           Total catch (count density)                                Richness                                  Equitability
----------------------------------------------------  ----------------------------------------  ----------------------------------------
            Df   Sum Sq  Mean Sq F value  Pr(>F)        Df  Sum Sq Mean Sq F value  Pr(>F)        Df  Sum Sq Mean Sq F value  Pr(>F)
T            2 5.67e+14 2.83e+14  2.3182  0.1035         2    73.9    37.0  0.8188  0.4438         2 0.16331 0.08166  7.9155  0.0006 ***
R            2 1.42e+15 7.08e+14  5.7923  0.0041 **      2  3287.7  1643.8 36.4110  0.0000 ***     2 0.48534 0.24267 23.5234  0.0000 ***
D            1 1.57e+12 1.57e+12  0.0128  0.9101         1    11.4    11.4  0.2519  0.6168         1 0.00027 0.00027  0.0258  0.8728
T:R          4 4.54e+15 1.13e+15  9.2780  0.0000 ***     4   362.2    90.5  2.0057  0.0991 .       4 0.44249 0.11062 10.7233  0.0000 ***
T:D          2 2.69e+14 1.35e+14  1.1023  0.3359         2   424.5   212.2  4.7011  0.0111 *       2 0.00558 0.00279  0.2706  0.7634
R:D          2 4.78e+13 2.39e+13  0.1956  0.8226         2    25.2    12.6  0.2792  0.7570         2 0.00554 0.00277  0.2685  0.7651
T:R:D        4 1.40e+14 3.49e+13  0.2857  0.8867         4   417.9   104.5  2.3140  0.0623 .       4 0.03568 0.00892  0.8646  0.4879
Residuals  104 1.27e+16 1.22e+14                       104  4695.3    45.1                       104 1.07287 0.01032
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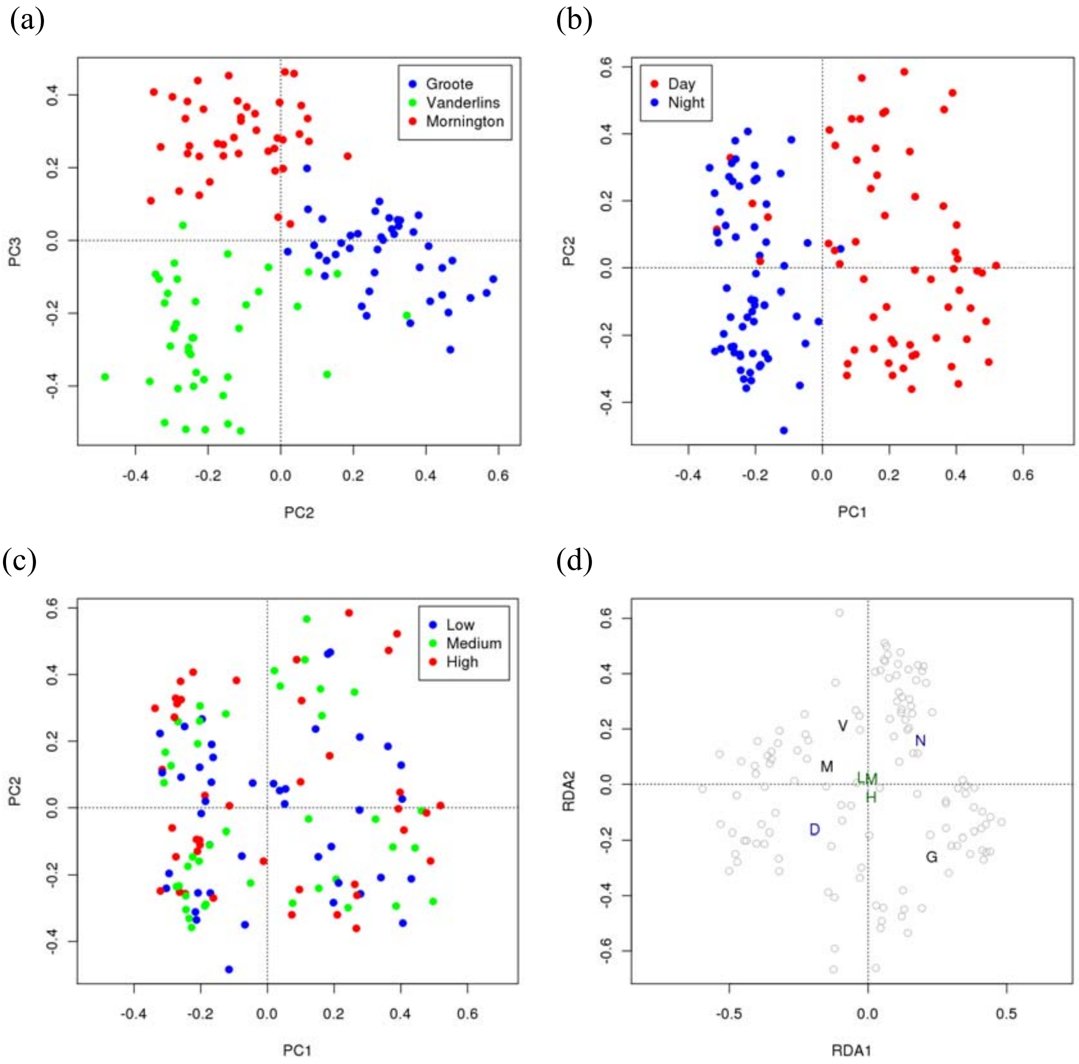

Figure 24. Ordination analyses of fish from prawn trawl samples. Figures a–c show sample scores from a PCA. Samples are coloured by (a) region, (b) day/night and (c) trawling intensity. The axes shown are those most informative for the displayed factor. Figure d shows sample scores from an RDA. The coloured letters indicate weighted average sample scores for samples with each value of each constraining variable (black, region; blue, day/night; green, trawling stratum)

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Table 19. NPMANOVA and variation partitioning analysis for fish from prawn trawl samples. The variation partitioning columns are percentages as follows: A, variation explained by the term; AB, A plus variation shared with the remaining terms; ABC, variation explained by the full model.

Source	d.f.	SS	MS	*F*	*p*	A	AB	ABC
Trawling	2	1.24	0.620	1.55	0.003	0.8	0.4	23.6
Day/night	1	5.30	5.296	13.25	0.001	8.8	7.6	23.6
Region	2	8.44	4.218	10.55	0.001	13.6	12.0	23.6
T × D	2	0.95	0.476	1.19	0.144	0.3	–0.1	23.6
T × R	4	2.24	0.560	1.40	0.007	1.1	0.3	23.6
D × R	2	1.96	0.981	2.45	0.001	2.1	1.5	23.6
T × D × R	4	1.57	0.392	0.98	0.515	–0.1	–0.7	23.6
Residual	105	41.96	0.400					
Total	122	63.85						

Table 20. Top and bottom ranked taxa on the first axis of an RDA of fish from prawn trawl samples. The analysis was constrained by trawling and conditioned on region and day/night. The axis runs from low to high trawling. Taxa associated with low trawling intensity are at the top of the table; those associated with high trawling intensity are at the bottom. The table includes the species score and the frequency of each taxon.

Taxon	Class	Order	Family	Score	Freq.
Pentaprion longimanus	Actinopterygii	Perciformes	Gerreidae	–0.06	78
Pristotis obtusirostris	Actinopterygii	Perciformes	Pomacentridae	–0.06	24
Carangoides caeruleopinnatus	Actinopterygii	Perciformes	Carangidae	–0.06	15
Lutjanus vitta	Actinopterygii	Perciformes	Lutjanidae	–0.05	20
Parupeneus heptacanthus	Actinopterygii	Perciformes	Mullidae	–0.05	8
Lutjanus malabaricus	Actinopterygii	Perciformes	Lutjanidae	–0.05	14
Nemipterus furcosus	Actinopterygii	Perciformes	Nemipteridae	–0.04	20
Epinephelus sexfasciatus	Actinopterygii	Perciformes	Serranidae	–0.04	32
Pomadasys maculatus	Actinopterygii	Perciformes	Haemulidae	–0.04	7
Choerodon monostigma	Actinopterygii	Perciformes	Labridae	–0.04	7
Polydactylus multiradiatus	Actinopterygii	Perciformes	Polynemidae	–0.04	8
Leiognathus bindus	Actinopterygii	Perciformes	Leiognathidae	–0.04	88
Pterocaesio chrysozona	Actinopterygii	Perciformes	Caesionidae	–0.04	7
Scolopsis taenioptera	Actinopterygii	Perciformes	Nemipteridae	–0.04	46
Johnius borneensis	Actinopterygii	Perciformes	Sciaenidae	–0.03	10
Chirocentrus dorab	Actinopterygii	Clupeiformes	Chirocentridae	–0.02	27
Choerodon sugillatum	Actinopterygii	Perciformes	Labridae	–0.02	10
Zabidius novemaculeatus	Actinopterygii	Perciformes	Ephippidae	–0.02	6
Apogon fasciatus	Actinopterygii	Perciformes	Apogonidae	–0.02	19
Ulua aurochs	Actinopterygii	Perciformes	Carangidae	–0.02	15
Carcharhinus dussumieri	Chondrichthyes	Carcharhiniformes	Carcharhinidae	0.03	11
Mene maculata	Actinopterygii	Perciformes	Menidae	0.03	20
Nemipterus hexodon	Actinopterygii	Perciformes	Nemipteridae	0.03	76
Carcharhinus tilstoni	Chondrichthyes	Carcharhiniformes	Carcharhinidae	0.03	6
Saurida argentea	Actinopterygii	Aulopiformes	Synodontidae	0.03	56
Saurida undosquamis	Actinopterygii	Aulopiformes	Synodontidae	0.03	77
Bregmaceros lanceolatus	Actinopterygii	Gadiformes	Bregmacerotidae	0.04	20
Stolephorus	Actinopterygii	Clupeiformes	Engraulidae	0.04	39
Sardinella gibbosa	Actinopterygii	Clupeiformes	Clupeidae	0.04	29
Apogon quadrifasciatus	Actinopterygii	Perciformes	Apogonidae	0.04	10
Leiognathus equulus	Actinopterygii	Perciformes	Leiognathidae	0.04	12
Alepes apercna	Actinopterygii	Perciformes	Carangidae	0.04	41
Parastromateus niger	Actinopterygii	Perciformes	Carangidae	0.04	15
Priacanthus tayenus	Actinopterygii	Perciformes	Priacanthidae	0.04	92
Pellona ditchela	Actinopterygii	Clupeiformes	Pristigasteridae	0.05	41
Carangoides humerosus	Actinopterygii	Perciformes	Carangidae	0.05	72
Yongeichthys nebulosus	Actinopterygii	Perciformes	Gobiidae	0.05	15
Carangoides talamparoides	Actinopterygii	Perciformes	Carangidae	0.06	53
Gazza minuta	Actinopterygii	Perciformes	Leiognathidae	0.06	21
Caranx bucculentus	Actinopterygii	Perciformes	Carangidae	0.11	76

3.4.2 INVERTEBRATES FROM EPIBENTHIC SLED SAMPLES

For invertebrates from epibenthic sled samples, total catch declined, although not highly significantly, from low to high trawling (Figure 22. Figure 23,Table 18). Species richness was not related to any of the strata, and equitability varied among regions and between day and night (Figure 22. Figure 23, Table 18).

For multivariate analysis, the sample by species matrix for invertebrates from epibenthic sled samples included 123 deployments and 73 taxa (42 species, 19 genera, 7 families, 2 orders, 1 subclass and 2 classes).

Invertebrate catch assemblage varied among regions and among trawling levels, but not between day and night. Regions were clearly segregated by the first two axes of the PCA ordination (Figure 25a). Day and night were interspersed on at least the first five axes (Figure 25b). Low trawling was separated from medium and high trawling on the third axis (Figure 25c). In the RDA constrained by region, day/night and trawling stratum, the first axis separated Groote from Vanderlins and Mornington (Figure 25d). The second axis separated low trawling from medium and high trawling, and Vanderlins from Mornington. In the NPMANOVA, variation in invertebrate catch assemblage among regions and trawling strata was significant, but variation between day and night was not significant (). The variation in species composition explained by trawling was small (2%) relative to that explained by region (12%) ().

In the RDA constrained by trawling stratum and conditioned on region and day/night, the first axis separated low trawling from medium and high trawling, and the second axis separated medium trawling from high trawling. Invertebrates associated with medium and high trawling included a hairy crab (*Cryptolutea arafurensis*), venus cockle (*Placamen tiara*), sea cucumbers (Holothuroidea) and an infaunal sponge (*Disyringa dissimilis*) (Table 22). Those associated with low trawling included a bryozoan (*Triphyllozoon*), brittle stars (Ophiurida) and a heart urchin (*Maretia planulata*) (Table 22)

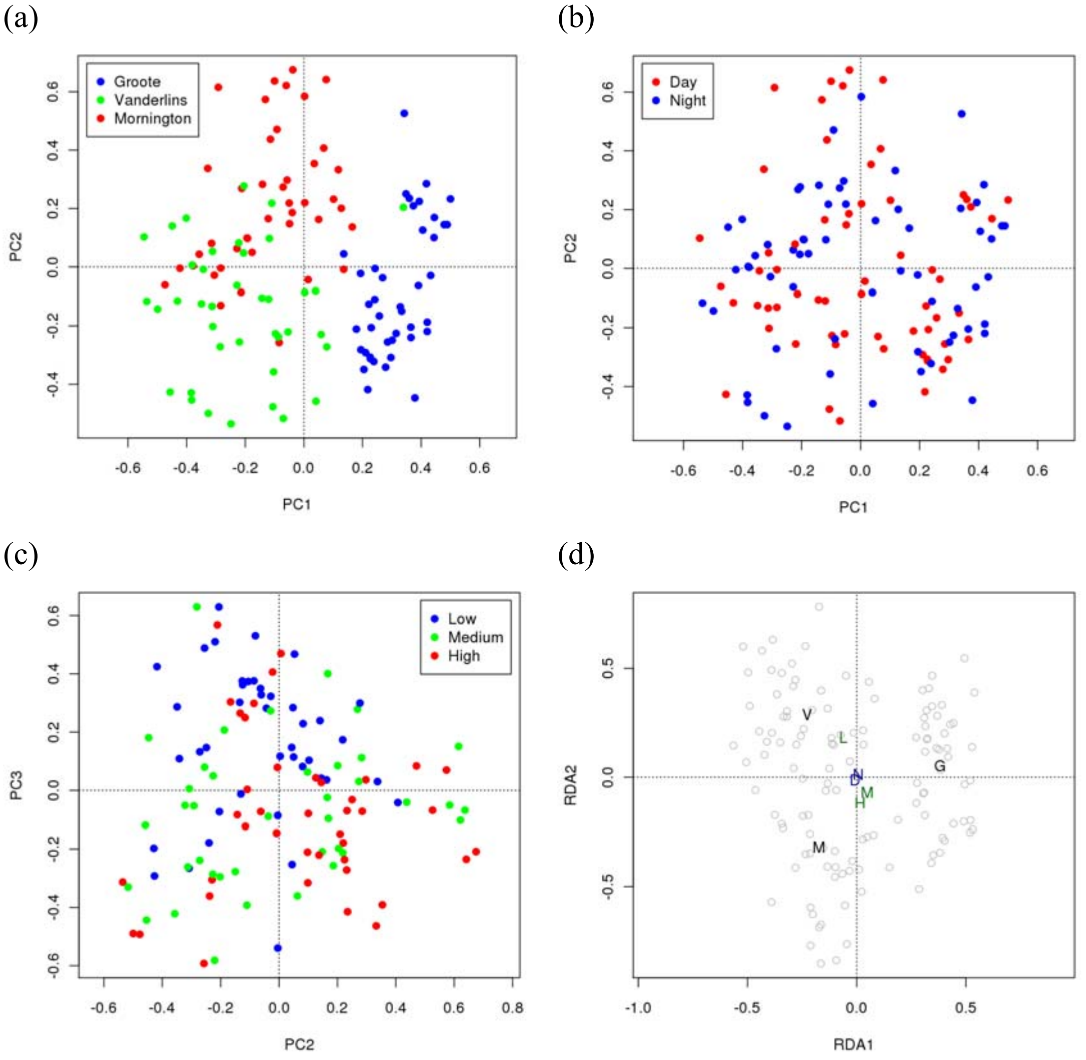

Figure 25. Ordination analyses of invertebrates from epibenthic sled samples. Figures a–c show sample scores from a PCA. Samples are coloured by (a) region, (b) day/night and (c) trawling intensity. The axes shown are those most informative for the displayed factor. Figure d shows sample scores from an RDA. The coloured letters indicate weighted average sample scores for samples with each value of each constraining variable (black, region; blue, day/night; green, trawling stratum).

Table 21. NPMANOVA and variation partitioning analysis for invertebrates from epibenthic sled samples. The variation partitioning columns are percentages as follows: A, variation explained by the term; AB, A plus variation shared with the remaining terms; ABC, variation explained by the full model.

Source	d.f.	SS	MS	*F*	*p*	A	AB	ABC
Trawling	2	3.01	1.506	2.33	0.001	2.1	1.7	14.6
Day/night	1	0.73	0.729	1.13	0.259	0.1	0.0	14.6
Region	2	10.92	5.459	8.44	0.001	11.9	10.5	14.6
T × D	2	1.28	0.640	0.99	0.510	–0.0	–0.3	14.6
T × R	4	4.53	1.132	1.75	0.001	2.4	1.7	14.6
D × R	2	1.52	0.758	1.17	0.158	0.3	0.0	14.6
T × D × R	4	2.48	0.620	0.96	0.574	–0.1	–0.7	14.6
Residual	105	67.89	0.647					
Total	122	92.39						

Table 22 Top and bottom ranked taxa on the first axis of an RDA of invertebrates from epibenthic sled samples. The analysis was constrained by trawling stratum and conditioned on region and day/night. The axis runs from low to high trawling. Taxa associated with low trawling intensity are at the top of the table; those associated with high trawling intensity are at the bottom. The table includes the species score and the frequency of each taxon.

Taxon	Class	Order	Family	Score	Freq
Triphyllozoon	Gymnolaemata	Cheilostomata	Phidoloporidae	–0.14	33
Ophiurida	Ophiuroidea	Ophiurida		–0.07	61
Maretia planulata	Echinoidea	Spatangoida	Spatangidae	–0.07	9
Arcania septemspinosa	Malacostraca	Decapoda	Leucosiidae	–0.07	7
Adeonella	Gymnolaemata	Cheilostomata	Adeonellidae	–0.06	8
Nursilia	Malacostraca	Decapoda	Leucosiidae	–0.06	7
Chicoreus cervicornis	Gastropoda	Neogastropoda	Muricidae	–0.06	6
Dendronephthya	Anthozoa	Alcyonacea	Nephtheidae	–0.05	14
Tellina	Bivalvia	Veneroida	Tellinidae	–0.05	7
Parthenope hoplonotus	Malacostraca	Decapoda	Parthenopidae	–0.05	7
Paguridae	Malacostraca	Decapoda	Paguridae	–0.05	7
Ascidiacea	Ascidiacea			–0.05	18
Upogebia	Malacostraca	Decapoda	Upogebiidae	–0.05	6
Inquisitor	Gastropoda	Neogastropoda	Turridae	–0.05	7
Diogenidae	Malacostraca	Decapoda	Diogenidae	–0.04	16
Vepricardium multispinosum	Bivalvia	Veneroida	Cardiidae	–0.04	12
Phalium bisulcatum	Gastropoda	Neotaenioglossa	Cassidae	–0.04	6
Peronella	Echinoidea	Clypeasteroida	Laganidae	–0.04	39
Luidiidae	Asteroidea	Platyasterida	Luidiidae	–0.04	8
Galene bispinosa	Malacostraca	Decapoda	Pilumnidae	–0.04	6
Liagore rubromaculata	Malacostraca	Decapoda	Carpiliidae	0.03	11
Bursa rana	Gastropoda	Neotaenioglossa	Bursidae	0.03	15
Semele casta	Bivalvia	Veneroida	Semelidae	0.03	7
Ensiculus cultellus	Bivalvia	Veneroida	Pharidae	0.04	11
Leucosia ocellata	Malacostraca	Decapoda	Leucosiidae	0.04	21
Trachypenaeus anchoralis	Malacostraca	Decapoda	Penaeidae	0.04	12
Scalopidia spinosipes	Malacostraca	Decapoda	Gonoplacidae	0.04	16
Leucosia pubescens	Malacostraca	Decapoda	Leucosiidae	0.04	24
Portunus spinipes	Malacostraca	Decapoda	Portunidae	0.04	24
Clorida	Malacostraca	Stomatopoda	Squillidae	0.05	24
Portunus hastatoides	Malacostraca	Decapoda	Portunidae	0.05	35
Metapenaeopsis	Malacostraca	Decapoda	Penaeidae	0.05	24
Parthenope longimanus	Malacostraca	Decapoda	Parthenopidae	0.05	23
Charybdis truncata	Malacostraca	Decapoda	Portunidae	0.06	20
Myra tumidospina	Malacostraca	Decapoda	Leucosiidae	0.06	38
Corbula	Bivalvia	Myoida	Corbulidae	0.07	37
Disyringa dissimilis	Demospongiae	Astrophorida	Ancorinidae	0.07	31
Holothuroidea	Holothuroidea			0.09	40
Placamen tiara	Bivalvia	Veneroida	Veneridae	0.12	33
Cryptolutea arafurensis	Malacostraca	Decapoda	Pilumnidae	0.29	54

3.4.3 INFAUNA FROM BOX CORE SAMPLES

For infauna from box core samples, all three ecological indicators varied among regions (Figure 22. Figure 23, Table 18). Total catch and equitability varied among trawling levels, although the pattern of variation varied among regions (Figure 22. Figure 23, Table 18). The high catch and low equitability in the high-trawling parts of the Groote region were mainly due to high catches of the tainaid crustacean *Apseudomorpha wagait*.

For multivariate analysis, the sample by species matrix for infauna from box core samples included 122 deployments and 92 taxa (22 species, 33 genera, 29 families, 4 orders, 4 classes and 2 phyla).

Infauna catch assemblage varied among regions and weakly among trawling levels, but not between day and night. Regions were clearly segregated on axes 1 and 4 of the PCA ordination, but day and night and trawling strata were interspersed Figure 26 a–c). In the RDA constrained by region, day/night and trawling stratum, the first axis separated Groote from Vanderlins and Mornington, and the second axis separated Vanderlins from Mornington (Figure 26d). In the NPMANOVA, variation in infauna catch assemblage among regions was significant (Table 23). Variation among trawling levels was significant only as an interaction with region, and the variation explained was very small (1%) (Table 23).

In the RDA constrained by trawling stratum and conditioned on region and day/night, the first axis separated low trawling from medium and high trawling, and the second axis separated medium trawling from high trawling. Infauna associated with medium and high trawling include a tanaid crustacean (*Apseudomorpha wagait*), polychaetes (*Terebellides* and *Notomastus*) and isopods (Anthuridae and *Cirolana*). Those associated with low trawling included some brittle stars (Ophiomyxidae), polychaetes (Nereididae, Chrysopetalidae, Acoetidae) and a beaked cockle (*Nuculana novaeguineensis*) (Table 24).

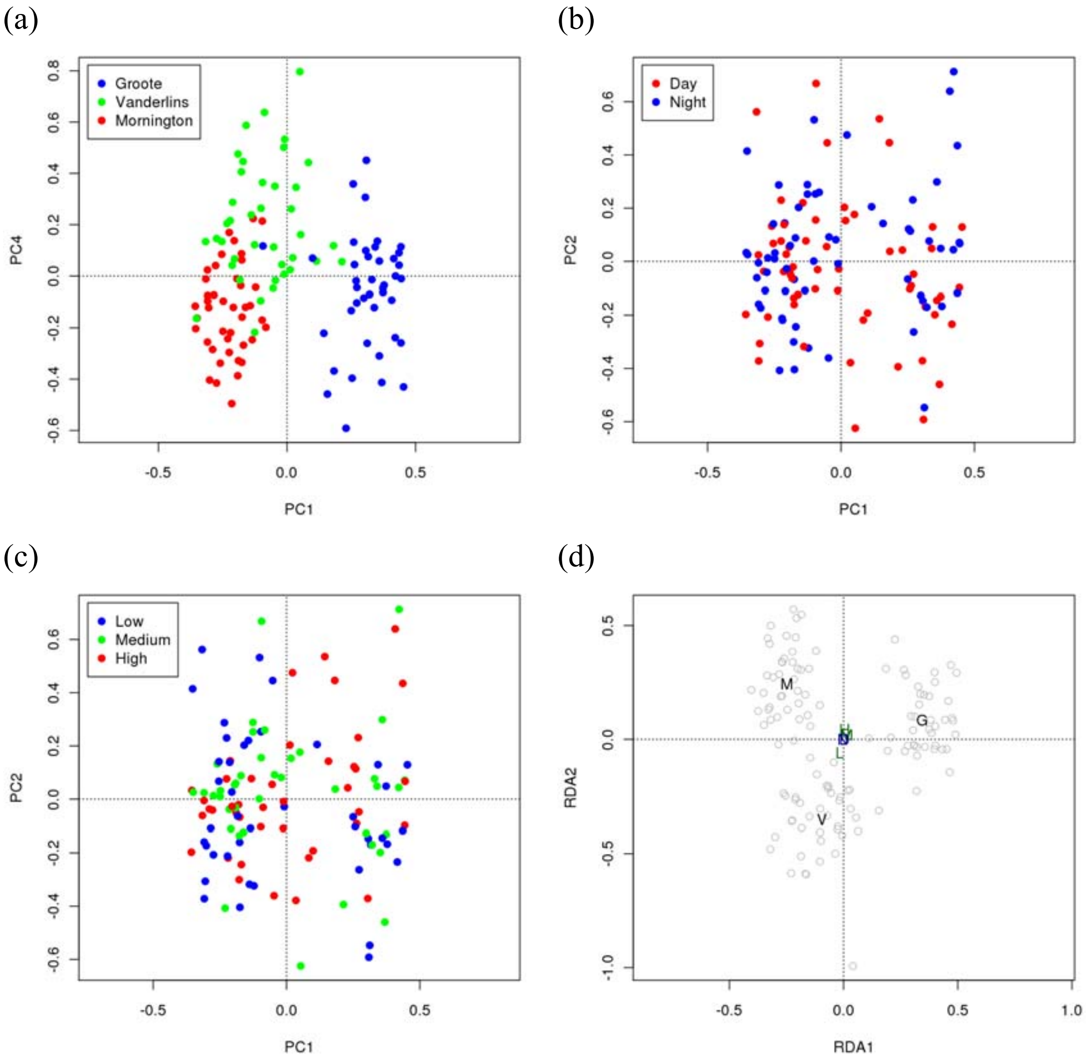

Figure 26. Ordination analyses of infauna from box core samples. Figures a–c show sample scores from a PCA. Samples are coloured by (a) region, (b) day/night and (c) trawling intensity. The axes shown are those most informative for the displayed factor. Figure d shows sample scores from an RDA. The coloured letters indicate weighted average sample scores for samples with each value of each constraining variable (black, region; blue, day/night; green, trawling stratum).

Table 23. NPMANOVA and variation partitioning analysis for infauna from box core samples. The variation partitioning columns are percentages as follows: A, variation explained by the term; AB, A plus variation shared with the remaining terms; ABC, variation explained by the full model.

Source	**d.f.**	**SS**	**MS**	***F***	***p***	**A**	**AB**	**ABC**
Trawling	2	1.13	0.563	1.26	0.056	0.4	0.2	12.3
Day/night	1	0.42	0.420	0.94	0.531	0.0	−0.2	12.3
Region	2	7.50	3.751	8.41	0.001	12.3	10.8	12.3
T × D	2	0.83	0.417	0.93	0.642	−0.1	−0.3	12.3
T × R	4	2.29	0.572	1.28	0.024	0.9	0.5	12.3
D × R	2	1.02	0.512	1.15	0.152	0.2	0.0	12.3
T × D × R	4	1.89	0.473	1.06	0.256	0.2	−0.2	12.3
Residual	104	46.41	0.446					
Total	121	61.54						

Table 24. Top and bottom ranked taxa on the first axis of an RDA of infauna from box core samples. The analysis was constrained by trawling stratum and conditioned on region and day/night. The axis runs from low to high trawling. Taxa associated with low trawling intensity are at the top of the table; those associated with high trawling intensity are at the bottom. The table includes the species score and the frequency of each taxon.

Taxon	Class	Order	Family	Score	Freq.
Ophiomyxidae	Ophiuroidea	Phrynophiurida	Ophiomyxidae	–0.06	28
Nereididae	Polychaeta	Aciculata	Nereididae	–0.06	33
Nuculana novaeguineensis	Bivalvia	Nuculoida	Nuculanidae	–0.06	15
Chrysopetalidae	Polychaeta	Aciculata	Chrysopetalidae	–0.05	18
Acoetidae	Polychaeta	Aciculata	Acoetidae	–0.05	25
Calanoida	Maxillopoda	Calanoida		–0.05	48
Nuculana darwini	Bivalvia	Nuculoida	Nuculanidae	–0.05	25
Ostracoda	Ostracoda			–0.05	79
Cumacea	Malacostraca	Cumacea		–0.05	66
Paraonidae	Polychaeta		Paraonidae	–0.04	35
Goniada tripartita	Polychaeta	Aciculata	Goniadidae	–0.04	19
Amphinomidae	Polychaeta	Aciculata	Amphinomidae	–0.04	26
Clorida	Malacostraca	Stomatopoda	Squillidae	–0.04	16
Syllidae	Polychaeta	Aciculata	Syllidae	–0.03	57
Rhynobrissus	Echinoidea	Spatangoida	Brissidae	–0.03	7
Naticidae	Gastropoda	Neotaenioglossa	Naticidae	–0.03	13
Leiochrides australis	Polychaeta		Capitellidae	–0.03	8
Sabellidae	Polychaeta	Canalipalpata	Sabellidae	–0.02	16
Venericardia	Bivalvia	Veneroida	Carditidae	–0.02	7
Nemata				–0.02	69
Tellina	Bivalvia	Veneroida	Tellinidae	0.02	41
Amphicteis philippinarum	Polychaeta	Canalipalpata	Ampharetidae	0.02	38
Corbula	Bivalvia	Myoida	Corbulidae	0.02	57
Paralacydonia weberi	Polychaeta	Aciculata	Paralacydoniidae	0.02	67
Coppingeria longisetosa	Polychaeta	Canalipalpata	Flabelligeridae	0.03	9
Sigalionidae	Polychaeta	Aciculata	Sigalionidae	0.03	20
Ophelina	Polychaeta		Opheliidae	0.03	16
Lumbrineridae	Polychaeta	Aciculata	Lumbrineridae	0.03	75
Sigambra tentaculata	Polychaeta	Aciculata	Pilargidae	0.03	25
Sphaeromatidae	Malacostraca	Isopoda	Sphaeromatidae	0.03	17
Sternaspis scutata	Polychaeta	Canalipalpata	Sternaspidae	0.03	52
Tucetona	Bivalvia	Arcoida	Glycymerididae	0.04	8
Lysilla	Polychaeta	Canalipalpata	Terebellidae	0.04	13
Onuphidae	Polychaeta	Aciculata	Onuphidae	0.04	49
Aglaophamus verrilli	Polychaeta	Aciculata	Nephtyidae	0.04	50
Cirolana	Malacostraca	Isopoda	Cirolanidae	0.04	16
Notomastus	Polychaeta	NA	Capitellidae	0.06	59
Terebellides	Polychaeta	Canalipalpata	Trichobranchidae	0.07	41
Anthuridae	Malacostraca	Isopoda	Anthuridae	0.08	62
Apseudomorpha wagait	Malacostraca	Tanaidacea	Metapseudidae	0.10	40

3.5 DISCUSSION

The relationships between catch species composition and the survey design strata varied among the three combinations of the sampling device and the group of taxa for which the analyses were performed. The composition of fish from prawn trawl samples was related to region and day/night, and weakly related to trawling. The composition of invertebrates from epibenthic sled samples was related to region and trawling, but not to day/night. The composition of infauna from box core samples was related to region and weakly to trawling, but not to day/night.

The empirical relationships between catch assemblage and trawling intensity detected by this analysis do not in themselves indicate that trawling affects biodiversity. At least some of the assemblage variation related to trawling may be caused by environmental factors that are correlated with trawling rather than trawling itself. To minimise that problem, our survey design minimised environmental variation within (but intentionally not among) the three sampling regions. The remaining environmental variation may include (a) factors that we know about and have data for (e.g. bathymetry), (b) factors that we know about but do not have data for (e.g. proximity to reefs), and (c) factors that we do not know are important. We can therefore not completely address the confounding effects of environmental variation and trawling with additional statistical analysis. For that we would need to have conducted a before–after, control–impact study rather than the control–impact study that was possible for this analysis of the long-term, broad scale impacts of trawling. However, we can partially address the effects by performing additional analyses on environmental factors we have data for and by incorporating knowledge of life history into the design and interpretation of analyses.

Results from the analysis are consistent with partial effects of both trawling and habitat. The association of fragile animals such as bryozoans and brittle stars with low trawling is consistent with a trawling effect. The association of reef-associated fish with low trawling and prawn predators with high trawling is consistent with a habitat effect.

3.6 REFERENCES

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APPENDIX 4 SEDIMENT AND WATER COLUMN PROCESSES IN THE GULF OF CARPENTARIA

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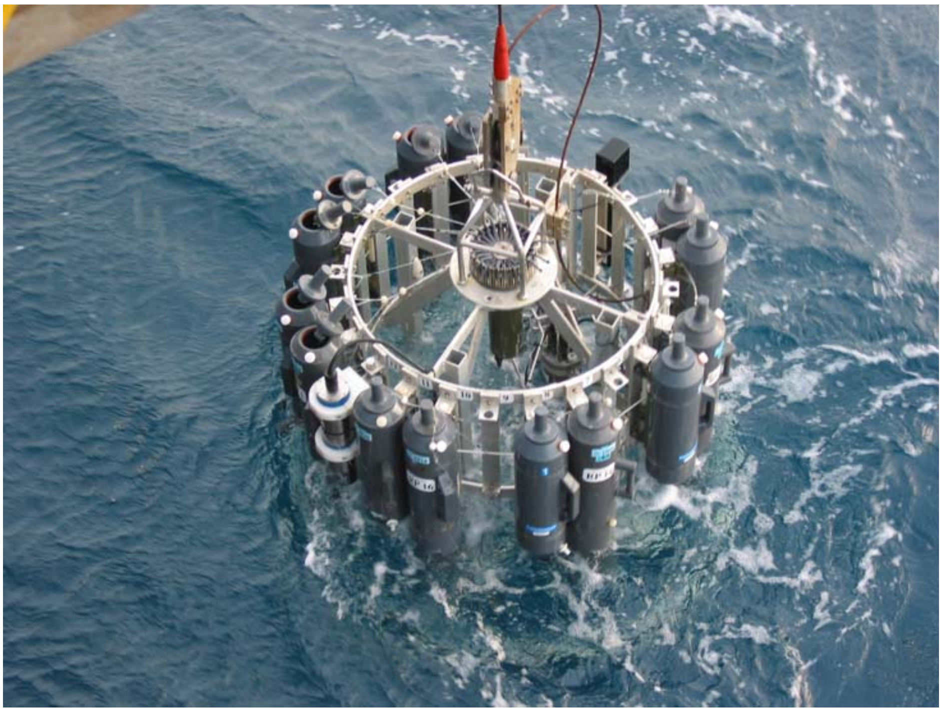

4.1 EXECUTIVE SUMMARY

The CMAR/FRDC-funded Gulf of Carpentaria voyage (SS0305) in summer 2005 aimed to examine the effect of trawling intensity on a range of physical, chemical and biological parameters in the sediments. Three regions were sampled – near Mornington Island in the southern Gulf, near the Vanderlins (southern Gulf), and near Groote Eylandt in the northwestern Gulf. During this voyage, the physico-chemical environment was also characterised. A component of the project findings is outlined in this document: namely characterisation of the physico-chemical environment during the voyage, and examination of the effect of region and trawling intensity on a range of biogeochemical and food web parameters.

Based on CTD data, early in the voyage the water temperatures were high and the water column was thermally stratified. This is typical for summer conditions in the Gulf. Dissolved nutrient (ammonia, nitrate plus nitrite, phosphate, silicate) concentrations, measured every 10 m through the water column, were typically very low both regionally and through the water column, but concentrations were higher in bottom waters. Chlorophyll *a* concentrations, measured periodically throughout the voyage, were also low in all regions but typically higher in bottom waters. During and after Cyclone Ingrid impacted the Groote Eylandt region the water column became well mixed with no gradients in temperature, oxygen or nutrients. The major change in meteorological conditions halfway through the voyage, and the subsequent effects on the chemical and physical attributes of the water column, made inter-region and inter-trawling strata comparisons impossible. Therefore the water column data only provides descriptive information for the voyage.

Sediment biogeochemical processes also measured using samples collected from box cores. Organic carbon and $\delta^{13}C$ ratios were measured at all sites, all trawl strata (low, medium, high) and all three regions. Denitrification (nitrogen gas removal via microbial processes) and sediment oxygen demand were also measured at some sites in each trawl strata and region. Organic carbon concentrations in the sediment were statistically different ($P<0.05$) between the regions (Mornington, Vanderlins, Groote) but not between trawling strata. $\delta^{13}C$ ratios, as a measure of microbial activity in the sediment, were not statistically different between trawl strata or regions. There was also no obvious difference in the sediment oxygen demand or denitrification rates between sampling strata or regions. However it must be acknowledged that this was based on a smaller sample size than for organic carbon or $\delta^{13}C$ ratios.

$\delta^{15}N$ ratios were also measured in the sediment and infauna throughout the Gulf, although replication was lower than for organic carbon or $\delta^{13}C$ ratios. Infaunal ratios were highly variable depending on the species group (molluscs, crustaceans, fish, worms) with crustaceans and molluscs typically having the lowest values. $\delta^{15}N$ ratios were lowest at Vanderlins, probably indicative of a bloom and subsequent decay of the nitrogen-fixing cyanobacterium, *Trichodesmium.* The bloom was not apparent after the wind mixing caused by the cyclone.

The data from both the SS0305 Gulf voyage and previous Gulf voyages was analysed to develop a nitrogen budget for the Gulf of Carpentaria. The analysis suggests that nitrogen-fixing cyanobacterium, *Trichodesmium*, plays a key role in providing a new source of nitrogen to fuel a nitrogen limited ecosystem. Coastal inputs of nutrients appear to be relatively minor.

To optimise the value of the data generated from this component of the SS0305 study, a statistical comparison of the sediment biogeochemical parameters with other components of the FRDC/CMAR study, i.e. sediment physical characteristics, infaunal and epifaunal communities is needed.

4.2 OCEANOGRAPHIC SURVEY

4.2.1 AIMS

This component of the study examined the physico-chemical environment in the water column during the R/V *Southern Surveyor* voyage (SS2005/03) in the Gulf of Carpentaria. A large dataset of oceanographic information was collected to provide the hydrological context for the studies of the benthos.

4.3 METHODS

CTD casts were performed at each sampling site to provide vertical profiles of temperature, salinity, oxygen and fluorescence throughout the water column. Additionally water samples were collected with Niskin bottles at the surface and at 10m intervals through the water column for dissolved nutrient (nitrate/nitrite, ammonia, phosphate, silicate) concentrations. Periodically chlorophyll *a* (9 sites) and photosynthetic activity (yield) (7 sites) of phytoplankton in surface and bottom waters was also measured.

Nutrients were analysed on board, with the exception of ammonia, using standard methods (Neale Johnson, CSIRO Marine Technology & Services). Ammonia was analysed in the laboratory after the voyage using standard methods. For chlorophyll *a* analysis, known volumes of water were filtered through glass fibre filters. The filters were then frozen until returned to the laboratory. Filters were extracted by sonicating for 1 min in cold 100% acetone, and extracts were measured spectrophotometrically after adjusting the acetone concentration to 90%, when there was sufficient chlorophyll, and with a spectrofluorometer when chlorophyll values were low (Jeffrey & Welshmeyer, 1997).

For measurement of photosynthetic yield, water samples collected from the Niskin bottles were placed in the dark for a minimum of 20 min. Samples were then read using a Walz PHYTOPAM to determine the yield resulting from a saturating pulse of light.

4.4 RESULTS & DISCUSSION

The water temperatures were highest and the water column was most highly stratified at Vanderlins with surface temperatures as high as 31.4ºC and a thermocline around 30m deep (Figure 27)

Mornington region, which was sampled early in the voyage, was also stratified with a thermocline around 20 m deep. Profiling in the Groote Eylandt region occurred during a period of cyclonic activity and resulted in a completely mixed water column with temperatures in surface waters as low as 30ºC.

Oxygen concentrations were highest in the surface waters in both Vanderlins and Mornington regions early in the voyage (Figure 28). Consistent with the stratified water column, oxygen concentrations were lowest in the bottom waters. With the advent of cyclonic winds and full mixing of the water column, both surface and bottom oxygen concentrations were lower.

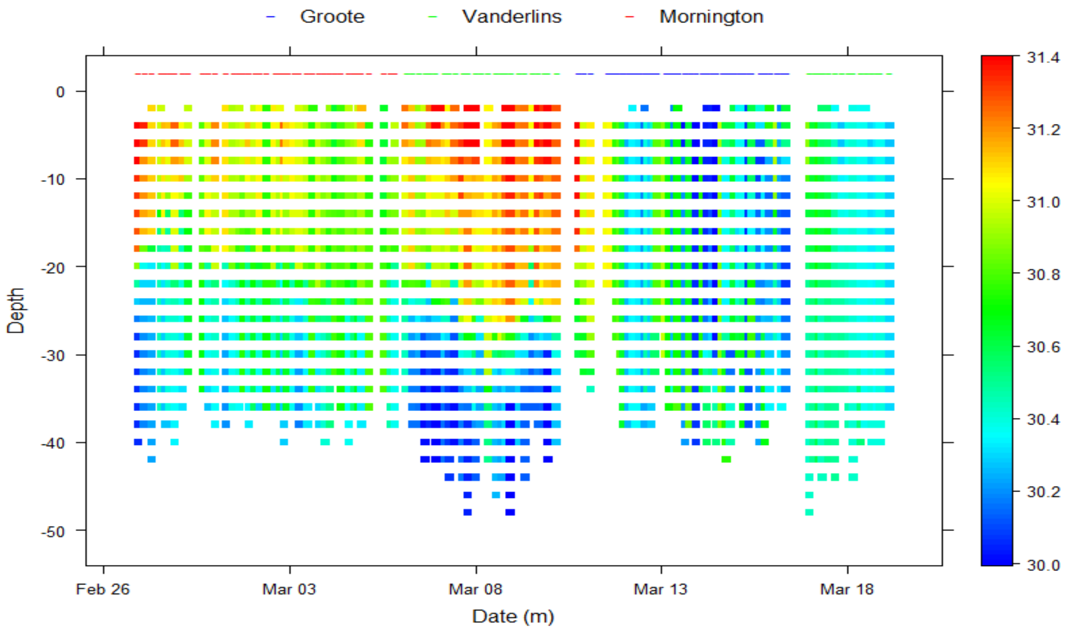

Figure 27. Temperature (^{o}C) profile throughout the water column at each sampling site. Profile maps generated by Wayne Rochester, CSIRO

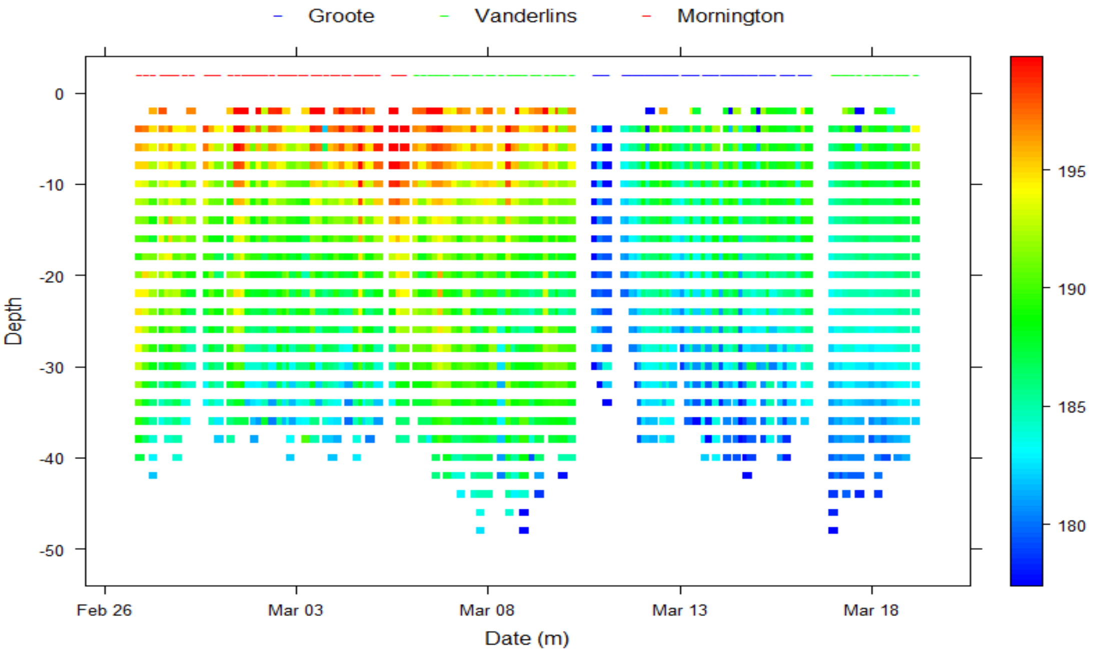

Figure 28: Oxygen profiles (µM) through the water column at each sampling site. Blue region = Groote Eylandt, Green region = Vanderlins, Red region = Mornington Island.

Mean concentrations of oxides of nitrogen, ammonia and phosphate were typically very low in all regions, but highly variable (Figure 29). Phosphate concentrations were similar to ammonia concentrations. Concentrations were comparable with those found in previous studies (Rothlisberg et al. 1989, Rothlisberg et al. 1994, Burford et al. 1994, 1995, Burford & Rothlisberg 1999). Additionally the low dissolved nitrogen, relative to phosphorus concentrations suggests that nitrogen, rather than phosphorus, is more likely to be limiting phytoplankton growth. This is consistent with studies by Burford & Rothlisberg (1999). Silicate does not appear to be a limiting nutrient in the Gulf, given mean concentrations of > 3 µM. This is also consistent with previous studies in the Gulf, which have shown that diatoms, which require silica for growth, are a significant proportion of the phytoplankton community (Hallegraeff & Jeffrey 1984, Burford et al. 1994, Hallegraeff & Burford 1996).

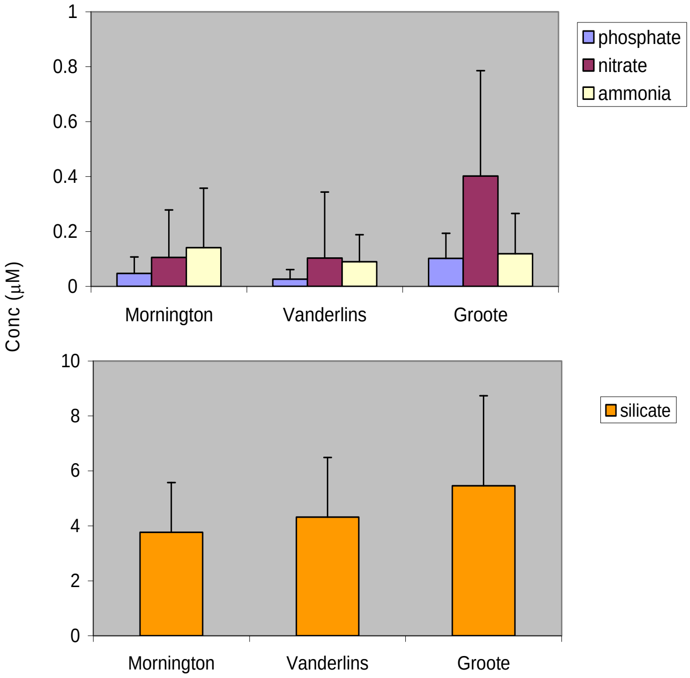

Figure 29. Mean (+ SD) nitrate, ammonia, phosphate and silicate concentrations (µM) through the water column at each of the three regions.

In Vanderlins and Mornington, nitrate, ammonia, phosphate and silicate concentrations were lowest in surface waters down to 20-30 m (Figure 30, Figure 31, Figure 32, Figure 33). Bottom waters had higher and more variable concentrations. In contrast, in the Groote Eylandt region, concentrations were higher and did not vary through the water column, coinciding with the period of high wind mixing. However after some days in this region, concentrations of all nutrients decreased throughout the entire water column.

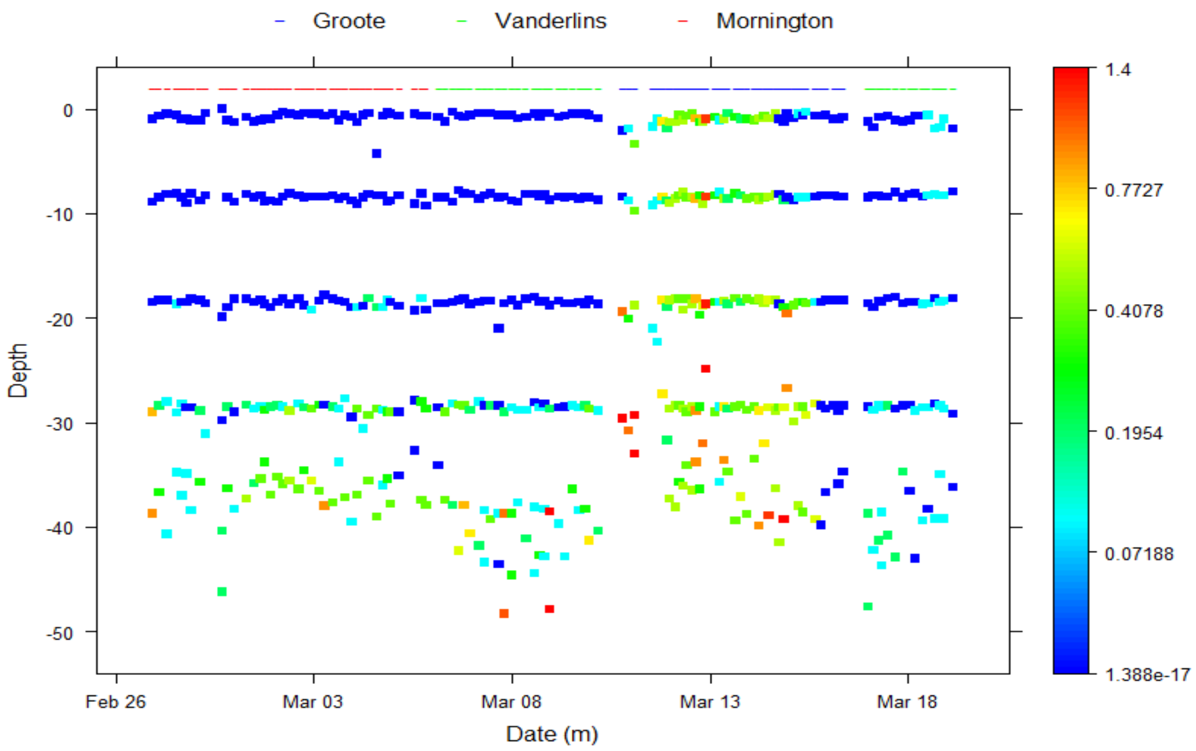

Figure 30. Nitrate concentrations (µM) at 10 m depth intervals at each sampling site. Blue region = Groote Eylandt, Green region = Vanderlins, Red region = Mornington Island.

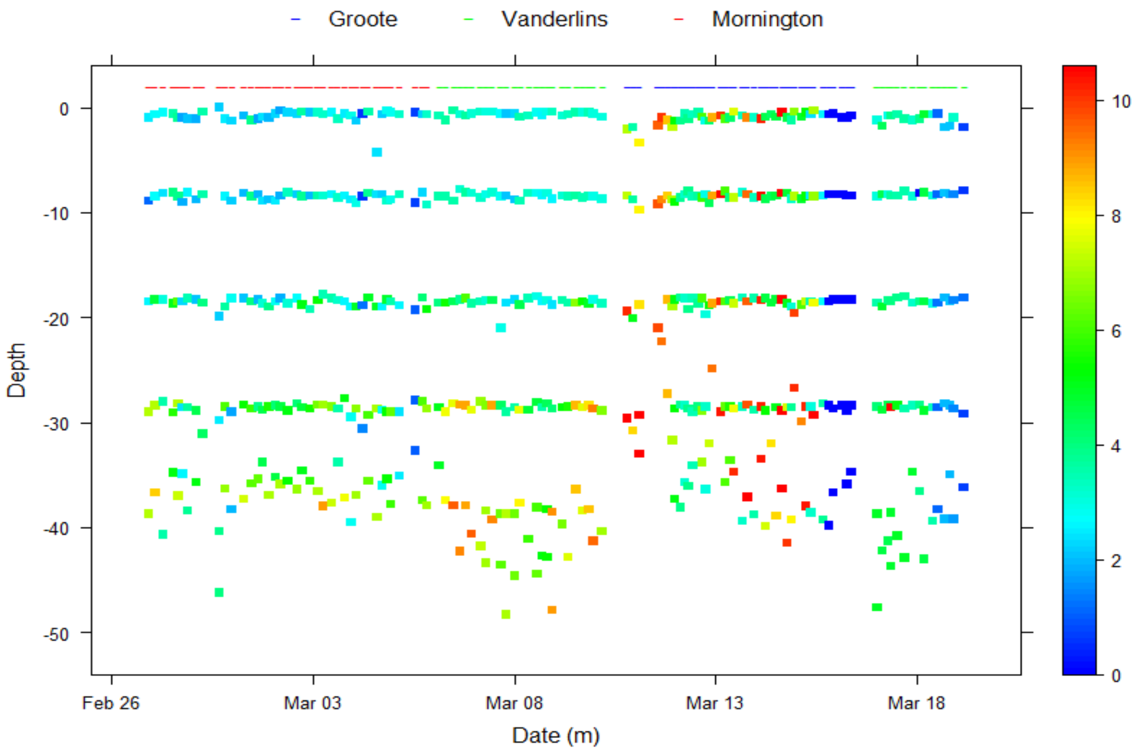

Figure 31. Ammonia concentrations (µM) at 10 m depth intervals.

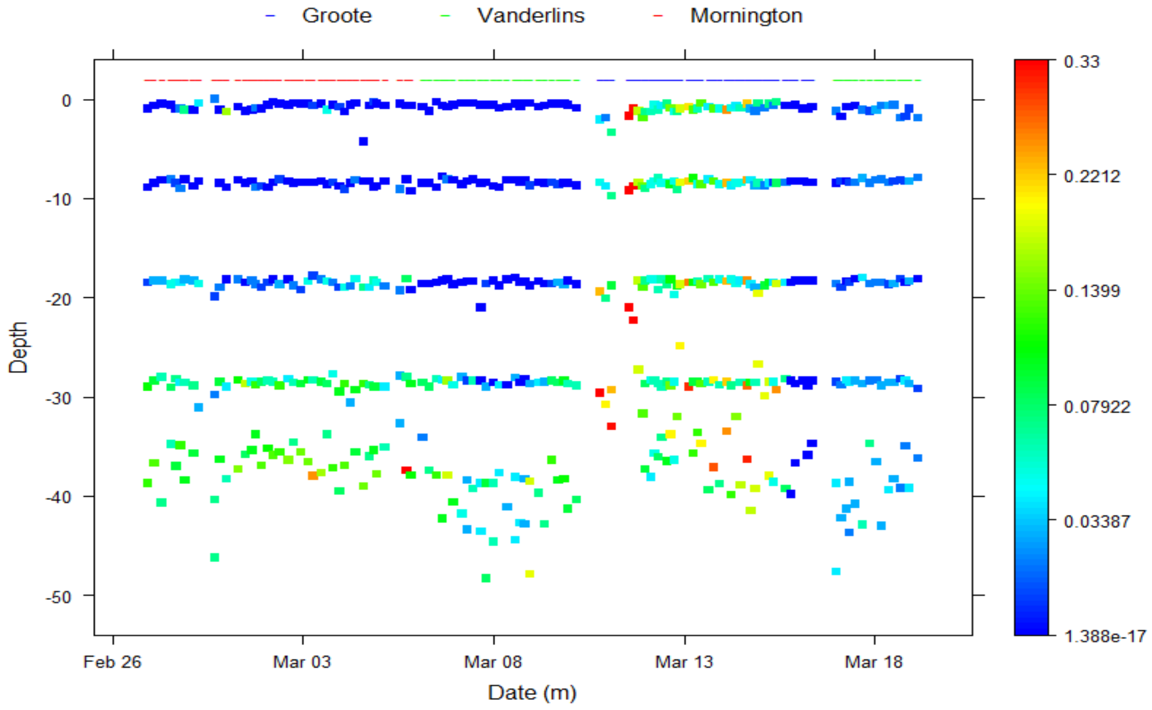

Figure 32. Phosphate concentrations (µM) at 10 m depth intervals.

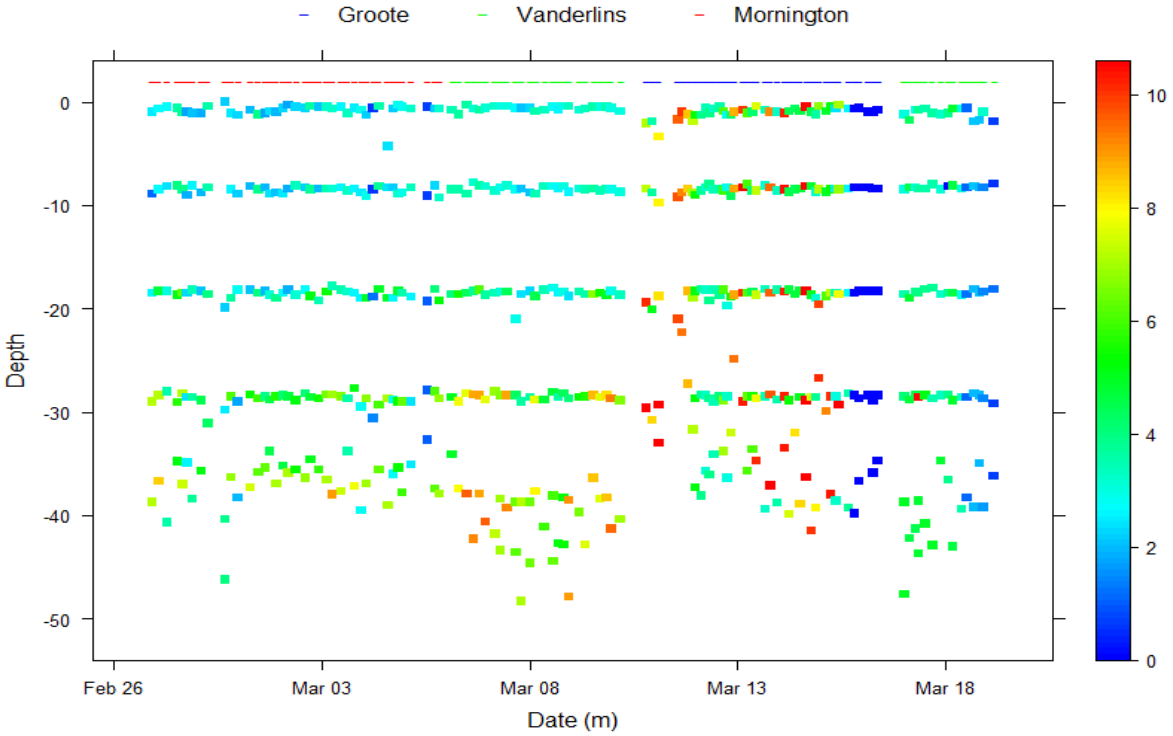

Figure 33. Silicate concentrations (µM) at 10 m depth intervals.

Chlorophyll *a* concentrations were measured at the surface and bottom at 9 sites throughout the Gulf (Table 25). The mean concentration was 0.80 ± 0.45 µg L^{-1}. Early in the voyage there were distinctly higher concentrations in bottom waters, but after cyclonic activity, concentrations were similar throughout the water column. The chlorophyll *a* concentrations during this voyage are similar to those measured on previous voyages to the Gulf (Rothlisberg et al. 1994, Burford et al. 1995, Burford & Rothlisberg 1999). Fluorescence profiles were not available due to problems with instrumentation.

Table 25. Chlorophyll a concentrations (Mean (SD) µg L^{-1}) in the surface and bottom waters at selected sites in Mornington, Vanderlins and Groote Eylandt.

Region	Surface/bottom	Chlorophyll *a* concentration	n
Mornington	Surface	0.52 (0.11)	3
	Bottom	1.47 (0.64)	3
Vanderlins	Surface	0.57 (0.43)	3
	Bottom	0.90 (0.27)	2
Groote	Surface	0.77 (0.16)	3
	Bottom	0.55 (0.17)	3

In addition to the chlorophyll *a* and fluorescence data, rates of photosynthetic yield by phytoplankton were measured using a PHYTOPAM. Yield measurements were typically quite low in surface waters, probably reflecting photoinhibition due to high light intensities (Figure 34). Bottom samples typically had higher yields indicative of a nutrient replete phytoplankton community capable of rapid growth. Interestingly, in the day after Cyclone Ingrid, surface samples had very low yield, suggesting that photosynthesis was being compromised, possibly due to vigorous physical mixing in the water column.

In contrast to the water column, the yield could not be measured on the sediment surface. This was because the fluorescence signal was very low, reflecting little benthic microalgal biomass. This is consistent with previous studies suggesting that there is a low biomass of benthic microalgae in the deeper (> 20m) waters of the Gulf and that the benthic production is driven by phytoplankton detritus, rather than benthic microalgae (Burford et al. 1994, 1995).

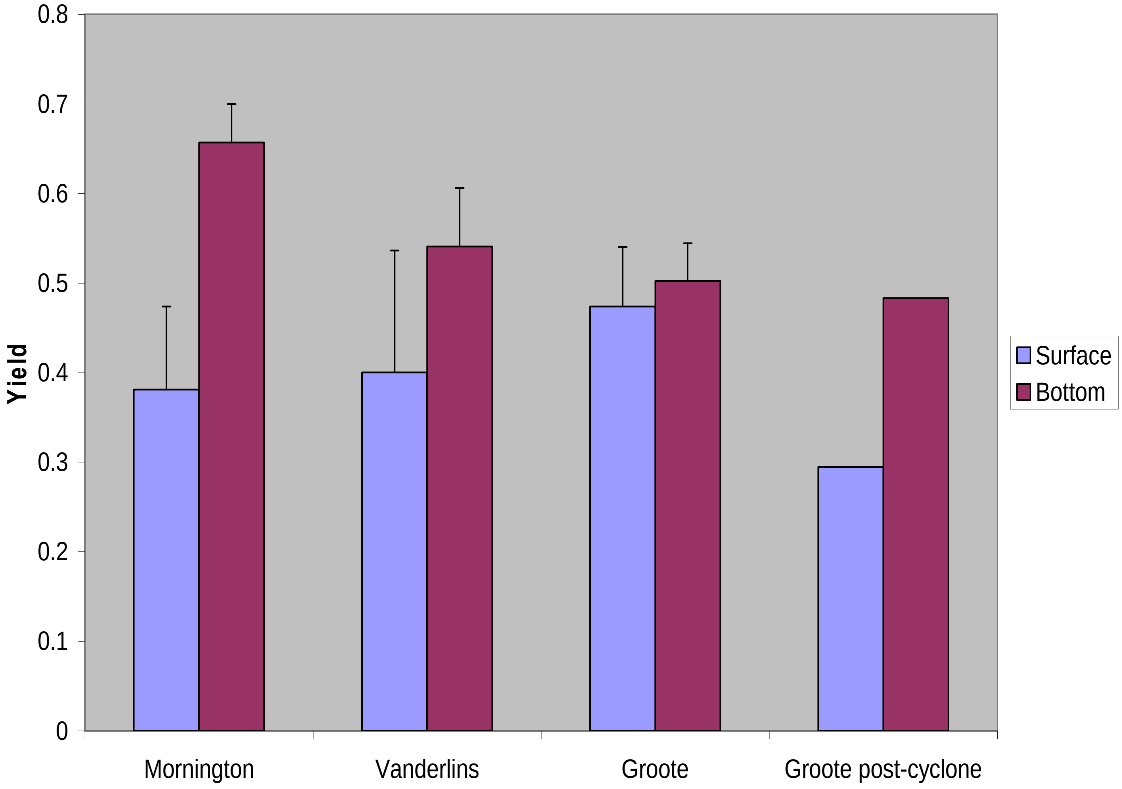

Figure 34. Mean (±SD) Photosynthetic yield for phytoplankton collected from the surface and bottom at three regions in the Gulf of Carpentaria

In summary, meteorological conditions appeared to have the greatest impact on physico-chemical parameters on the Gulf voyage. This means that it is not advisable to use the data for statistical comparisons between regions or trawling strata. Therefore, the data is mostly useful in a descriptive sense.

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4.6 SEDIMENT PROCESSES

4.6.1 AIMS

This component of the study was aimed at examining the effect of region (Mornington, Vanderlins, Groote) and trawling intensity (low, medium, high) on a range of sediment biogeochemical processes (sediment oxygen demand, denitrification) and parameters (organic carbon concentrations, stable isotope ratios).

4.7 METHODS

During the *Southern Surveyor* voyage, mini sediment cores were collected from the box cores at every sampling site. The cores were used for a number of analyses: denitrification rates; sediment oxygen demand rates; stable isotope signatures; and carbon and nitrogen content.

Triplicate sediment cores were collected at every sampling site. The top 2 cm of each undisturbed core was sliced off and frozen until returned to the laboratory. In the laboratory, sediments were screened through a wire mesh screen to remove rocks and rubble, dried at 60°C and ground. Half the sample had 1M HCl added, the sediment stirred, then allowed to effervesce in the fume hood to dissolve the carbonaceous material. Once effervescence stopped, the acidified sample was dried at 60°C for 24 h. Samples were then analysed in a mass spectrometer to determine δ^{13}C-carbon ratios and %carbon concentrations. The other half of the sample which was not acidified was analysed in a mass spectrometer to determine δ^{15}N-carbon ratios.

Sediment oxygen demand (SOD) incubations were done in the dark on-board in sealed Perspex chambers fitted with oxygen electrodes and pumps circulating water above the sediment. Changes in oxygen concentrations were recorded every 10 min for 24 h in duplicate chambers. The number of sites analysed were constrained by equipment and staff availability. As a result, only three to four sites for each trawl intensity, for each region, were completed.

Sediment cores were collected at 27 sites for denitrification measurements (i.e. nitrogen gas production from microbial activity). Within the trawling strata of high, medium and low, three sites were sampled and incubations conducted. Details of the incubations, analyses and calculations are given in the Appendix 5.

Infauna were also sorted from triplicate sediment box cores at 6 sites within each region for stable isotope analysis. Infauna were grouped into crustaceans, molluscs, worms and fish. Numbers of animals were typically low. Samples were dried at 60°C for 24 h. Samples were ground and half the sample was acidified as described above for δ^{13}C-carbon. The other half was analysed without acidification for δ^{15}N-nitrogen ratios. All samples were analysed in a mass spectrometer.

Statistical analyses, using SAS (Version 9.1) software were performed to examine differences in the organic carbon and δ^{13}C-carbon ratios in the sediment samples between regions and trawl strata. Both organic carbon and δ^{13}C-carbon ratios were square-root transformed before being analysed using PROC GLM.

4.8 RESULTS & DISCUSSION

Mean organic carbon concentrations in the sediment were not statistically different between trawling strata ($P>0.05$) with mean values ranging from 0.57 to 0.64%. However there were statistically higher concentrations in the Groote Eylandt region ($P<0.05$) compared with either Mornington or Vanderlins (Figure 35, Table 2). There was no interaction between region and trawl strata. These findings are consistent with previous studies in the Gulf that showed higher organic carbon concentrations in the northeast Gulf compared with the southern Gulf (Burford et al. 1994, Somers & Long 1994).

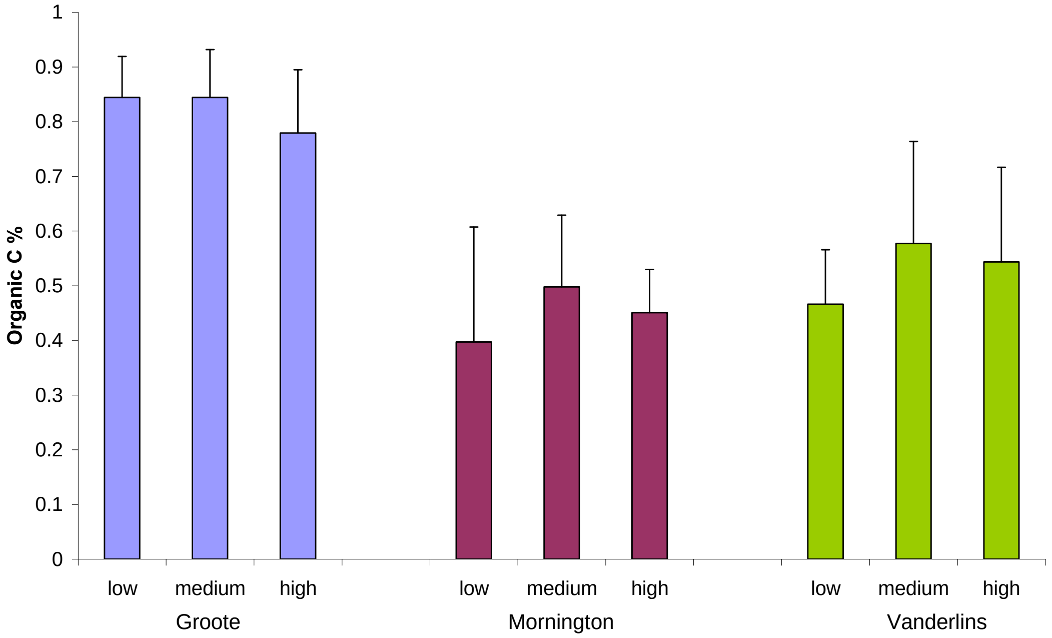

Figure 35. Mean (+ SD) organic carbon concentrations (%) in the surface sediments in the three trawling strata (high, low, medium) in the three regions in the Gulf.

Mean δ^{13}C stable isotope ratios in the sediment were highly variable and not statistically different (*P*>0.05*)* between strata or regions (Figure 36).

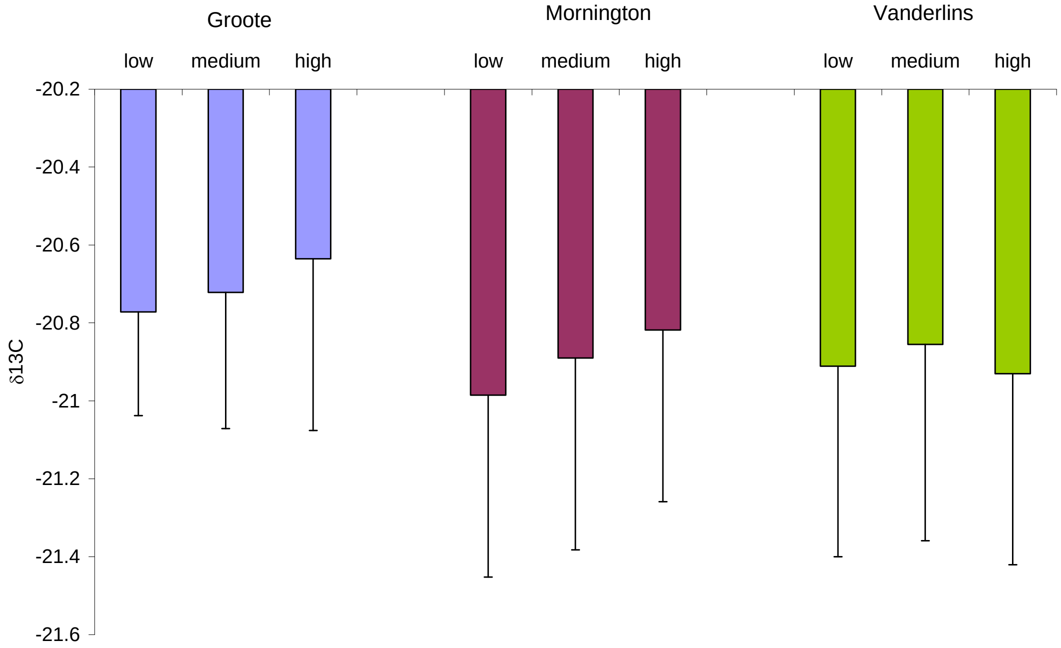

Figure 36. Mean (- SD) δ^{13}C stable isotope ratios in the surface sediments in the three trawling strata (high, low, medium) in the three regions in the Gulf.

Mean δ^{15}N stable isotope ratios were measured in the sediment at a subset of sites (Figure 36). Ratios were lowest in Vanderlins and highest in Groote Eylandt. Lower ratios typically reflect higher rates of nitrogen fixation. This may be the result of water column processes, i.e. nitrogen fixing *Trichodesmium* blooms which have died and settled as detritus and/or nitrogen fixation by bacteria within the sediment. A *Trichodesmium* bloom was observed during the sampling in the Mornington and Vanderlins regions when conditions were calm during the first part of the voyage (see Appendix 11). Sampling in Groote occurred during rough conditions when there was no evidence of a *Trichodesmium* bloom.

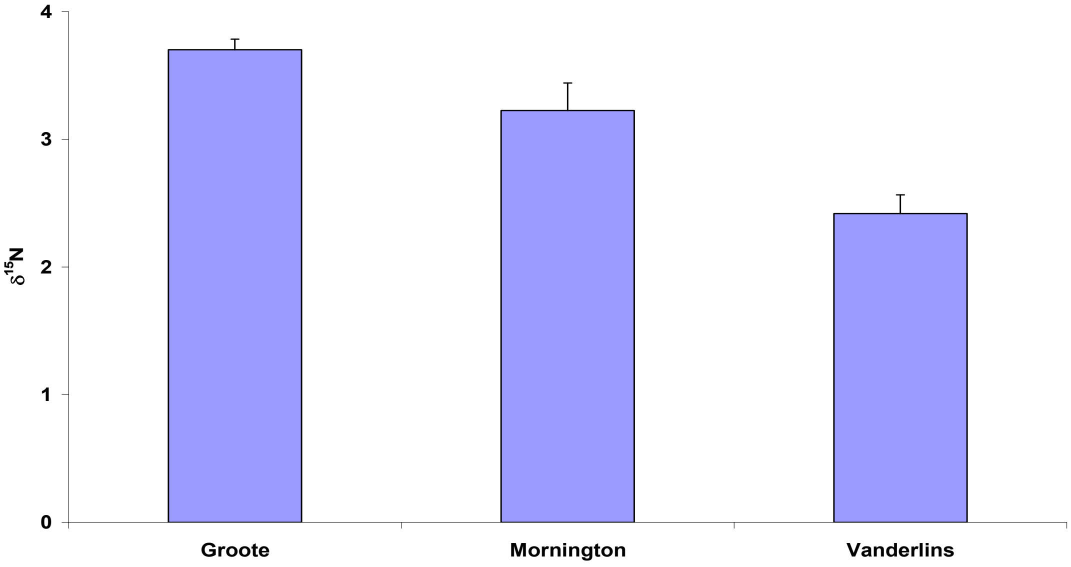

Figure 37. Mean (+SE) δ15N stable isotope ratios in the surface sediments in the three trawling strata (high, low, medium) in the three regions in the Gulf.

There was no obvious difference in SOD with changes in trawl intensity although SOD was less variable in the Mornington Island region than either Vanderlins or Groote Eylandt (Figure 38). This is consistent with a study of benthic trawl disturbance in the North Sea which measured no difference in sediment oxygen uptake between high and low trawling intensities (Trimmer et al. 2005).

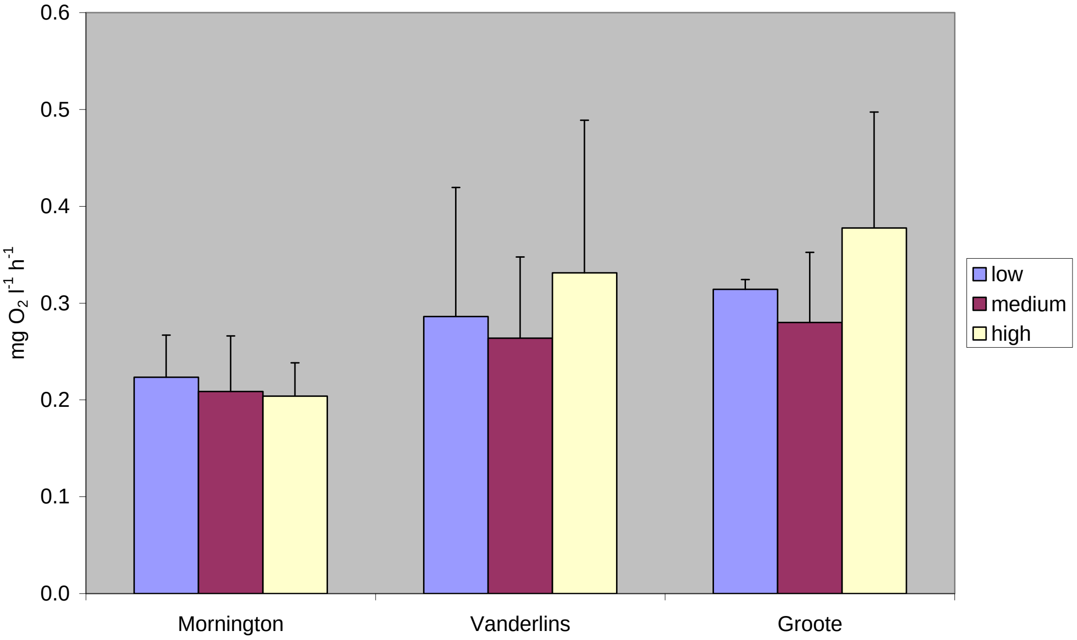

Figure 38. Mean sediment oxygen demand (± SD) (mg O_2 L^{-1} h^{-1}) at each trawling intensity in each region.

Denitrification rates were very low, i.e. mean of 0.27 ± 0.02 µmol m^{-2} h^{-1} throughout the Gulf and there was no statistical difference between strata or regions. This is also consistent with the finding of Trimmer et al. (2005). Their study found no difference in denitrification rates with trawl intensity. The low rates also suggest a low nitrogen environment with insufficient nitrate to fuel denitrification. Further discussion of the denitrification rates is given in Appendix 5.

There were no consistent differences in the $\delta^{13}C$ and $\delta^{15}N$ ratios of infauna between regions. Indeed there was a wide range of $\delta^{13}C$ and $\delta^{15}N$ ratios between taxonomic groups and between regions, (Figure 39) suggesting a range of food sources and trophic positions. The low numbers of individuals collected from each site has made more rigorous statistical analysis impossible.

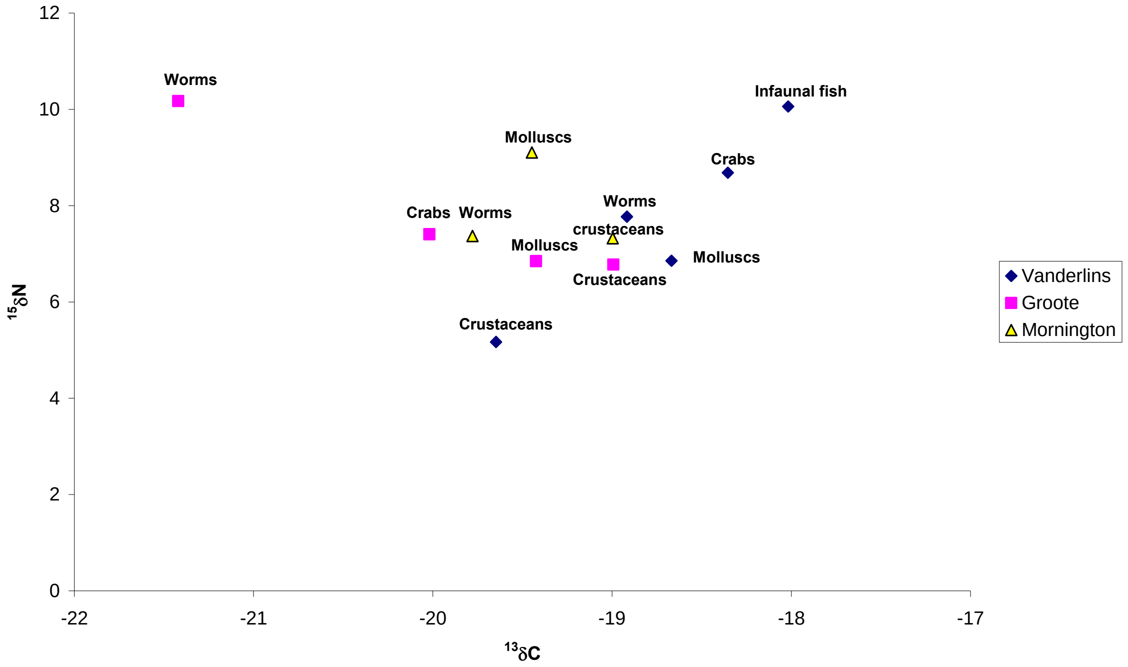

Figure 39. Mean $\delta^{13}C$ and $\delta^{15}N$ ratios for infaunal taxonomic groups in the three regions in the Gulf of Carpentaria

In summary, there was no evidence of differences in sediment processes between trawling strata or regions. The exception to this was organic carbon concentrations which were higher in the Groote Eylandt region than in the southern Gulf.

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APPENDIX 5 SOURCES OF NUTRIENTS DRIVING PRODUCTION IN THE GULF OF CARPENTARIA, AUSTRALIA – A SHALLOW TROPICAL SHELF SYSTEM

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Manuscript for submission to Estuarine, Coastal and Shelf Science

5.1 ABSTRACT

This study examined the sources and transformations of nitrogen, and the role it plays in fuelling primary productivity in a tropical shelf system, the Gulf of Carpentaria, Australia, using a combination of historical and new data. The Gulf appears to be a nitrogen limited system. This conclusion is based on: low dissolved inorganic nitrogen concentrations; low dissolved inorganic nitrogen: phosphate ratios; stimulation of primary productivity with nitrogen, but not phosphorus addition; prevalence of a nitrogen-fixing cyanobacterium *Trichodesmium*; and low rates of denitrification. The nitrogen budget estimates on a whole-of-Gulf basis suggest that nitrogen inputs from rivers are unlikely to be major contributors to primary productivity. However, in the coastal area (< 20 m deep) is separated from the deeper Gulf by a coastal boundary current. This constitutes 20% of the areal extent of the Gulf and thus annual river inputs are relatively more important. In the deeper waters of the Gulf, outside the coastal boundary current, the main source of nitrogen is estimated to be nitrogen fixation by cyanobacteria, principally *Trichodesmium.* This species is in highest abundance in summer months when stratification of the water column occurs. Our study demonstrated high rates of nitrogen fixation and depleted δ^{15}N-nitrogen ratios in the particulate matter in the water column, indicating nitrogen fixation, during a bloom. During periods of summer stratification, nitrogen concentrations increase in bottom waters, suggesting that benthic mineralization is occurring. δ^{15}N-nitrogen ratios were also depleted in the sediment at this time. It is therefore likely that detrital material from nitrogen-rich *Trichodesmium* is an important contributor to these benthic processes. During winter, wind-driven mixing results in nitrogen from bottom waters reaching the euphotic zone, and fuelling primary productivity. Therefore, *Trichodesmium* plays both an important direct and indirect role in contributing to primary productivity in the Gulf depending on the seasonal mixing regime.

5.2 INTRODUCTION

Tropical coastal waters are characterized by high water temperatures and large summer rainfall events resulting in significant inflows and associated inputs of nutrients from adjacent rivers. This contributes to high year-round primary production (Longhurst & Pauly 1987). Development of catchments for agriculture and other human activities has typically reduced or regulated river flows in many rivers, and increased nutrient loads to coastal waters (Carpenter et al. 1998, Beman et al. 2005). The increase in algal blooms worldwide, including harmful species, and development of dead zones in coastal waters has been attributed to increasing nutrient inputs from catchments (Ferber 2001, Anderson et al. 2002, Chen et al. 2007).

Further offshore many areas of the tropics have low dissolved inorganic nitrogen concentrations and low N:P ratios, resulting in nitrogen limitation. In these areas, atmospheric nitrogen inputs from cyanobacteria can be an important source of nitrogen. The most studied species is the filamentous cyanobacterium *Trichodesmium* which can form large surface rafts stretching hundreds of kilometres, and resulting in a high input of nitrogen to fuel productivity. More recently the importance of other symbiotic and small unicellular cyanobacteria in providing nitrogen to tropical food webs has also been established (Carpenter *et al*. 1999, Zehr *et al*. 2001, Montoya *et al* 2004, Capone *et al*. 2006).

The Gulf of Carpentaria in tropical Australia is a large coastal sea (330,000 km^2) surrounded by many large rivers seasonally providing 92,000 GL of water to the Gulf each year during the NW monsoonal wet season (January to March). The shallow depth of the Gulf (<70 m) combined with the high tidal energy means that tidal and wind mixing dominates over advective processes (Forbes 1984, Wolanski & Ridd 1990). There is also relatively little mixing of waters in the Gulf with the adjacent Arafura and Coral Seas (Forbes 1984). Within the Gulf there are two distinct regions: a turbid nearshore zone at a depth of 15m to 20 m; and deeper waters separated from the coastal zone by a boundary (buoyancy) current (Wolanski & Ridd 1990). River runoff is effectively trapped within the coastal zone.

Unlike many tropical areas around the world, this region has a low human population (~9,000) and limited human activities in the seas or catchments. However, there is increasing pressure for human development in the Gulf region, including water abstraction for irrigation, and regulation of water flow by damming. However, the role of river inputs in promoting coastal production, and the links with offshore production remain poorly understood. This study developed a nitrogen budget for the Gulf, based on measurements carried out in this study and on historical data, to establish whether changes in river inputs are likely to have a significant effect on the Gulf.

5.3 MATERIALS AND METHODS

Three separate sampling trips to the Gulf of Carpentaria in the summer wet season are described below. The first two trips focussed on sampling the water column and sediment particulate organic matter for stable isotope analyses. The first voyage was aboard the FV *Flinders Pearl* from 12 to 29 January 2004. Areas sampled were: the offshore waters adjacent to Albatross Bay; the southeastern Gulf near Karumba; near Mornington Is.; the Vanderlins; and Groote Eylandt (Figure 40). The second sampling episode was aboard the FV *Northern Pearl* on 8 February 2005 when Albatross Bay was sampled.

The final voyage was aboard the RV *Southern Surveyor* between 27 February and 19 March 2005. Three areas adjacent to Mornington Island, the Vanderlins and Groote Eylandt were sampled (Figure 40). A range of physico-chemical parameters were sampled in the water column. Samples of particulate organic matter were also taken in the water column and sediment for stable isotope analyses. Sediment box cores were also collected for denitrification measurements. There was a bloom of the cyanobacterium *Trichodesmium* during the voyage and nitrogen fixation measurements were also taken of the bloom.

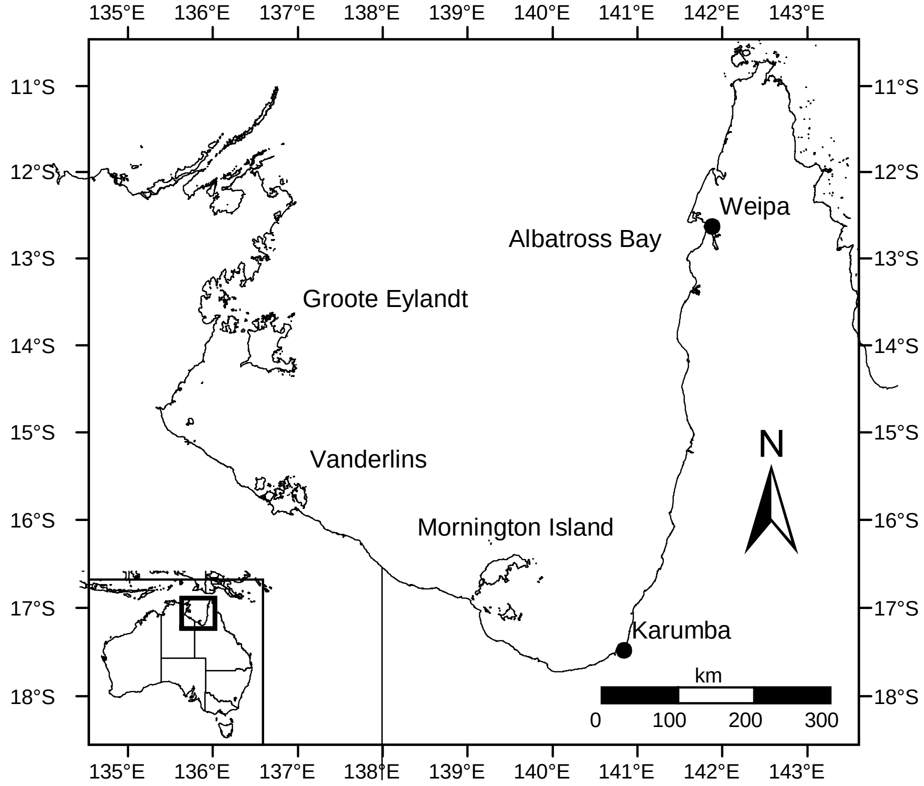

Figure 40. Map showing regions of the Gulf sampled on the three trips.

5.3.1 NUTRIENTS AND PHYSICO-CHEMICAL PARAMETERS

Water samples were collected every 10 m through the water column at each station on the RV *Southern Surveyor* voyage. A 12-bottle rosette of 10 L Niskin bottles was used to collect water. The samples were immediately put into glass and plastic bottles for nutrient and chlorophyll *a* analyses. Temperature and oxygen were measured continuously through the water column using a CTD.

Silicate, phosphate and nitrate plus nitrite concentrations were measured on board with an autoanalyzer during the Gulf transects (Airey & Sandars 1987). The nitrate method was modified with imidazole buffer and cleaned cadmium granules to convert nitrate to nitrite. Ammonia concentrations were measured on frozen samples on shore using the standard colorimetric methods (American Public Health Association, 1995).

For chlorophyll analysis, known volumes of water were filtered through glass fibre filters (Whatman GF/F) to trap the particulates. The filters were then frozen until returned to the laboratory. Filters were extracted by sonicating for 1 min in cold 100% acetone, and extracts were measured spectrophotometrically after adjusting the acetone concentration to 90%, when there was sufficient chlorophyll, and with a spectrofluorometer when chlorophyll values were low (Jeffrey & Welshmeyer, 1997).

5.3.2 NITROGEN FIXATION MEASUREMENTS

During the RV *Southern Surveyor* voyage, concentrated *Trichodesmium trichomes* was observed in the surface waters on 2 March 2005. A bucket was used to sample the surface water. Subsamples were poured into twelve 250 mL Schott bottles with sealed rubber ports in the lid. Acetylene gas, pre-bubbled through water to remove contaminants, was bubbled into the sample in the sealed bottles for 5 min to ensure saturation of the water with acetylene. Triplicate bottles were incubated at four light levels using shade bags: 100% surface light, 50% surface light, 8% surface light and 0% surface light. Samples were incubated on deck for 24 h in a plastic tub with flow-through seawater taken near the surface. Temperature was logged throughout the incubations in an additional bottle using a sensor (ETemperature, On Solutions). Subsamples (3 mL) were withdrawn from the bottles into Venoject vials containing 0.2 mL 20% trichloroacetic acid to halt nitrogen fixation at 0, 4 and 24 h. Subsamples were stored at 4ºC until analyzed in the laboratory. A gas chromatograph with a flame ionization detector was used to determine the ethylene content of the samples. Nitrogen fixation rates were calculated from the conversion of acetylene to ethylene (Knowles, 1990).

Subsamples of water were taken at time zero (t_o) for chlorophyll *a* concentrations and ^{15}N-nitrogen stable isotope ratios. For chlorophyll analysis, the method described above was used. For stable isotope analyses in particulate material, triplicate water samples were filtered through precombusted glass fibre filters (Whatman GF/F) to trap the particulate organic matter. Filters were then frozen until returned to the laboratory where they were dried at 60ºC in an oven then analysed for δ^{15}N-nitrogen ratios with a continuous-flow isotope-ratio mass spectrometer (Europa Tracermass and Roboprep, Crewe, England).

5.3.3 DENITRIFICATION

Rates of denitrification were measured in the sediment using the isotope pairing technique (Nielsen 1992) as described by Dalsgaard et al. (2000). Box cores were taken at 29 sites across the three regions on the FRV *Southern Surveyor* voyage described above. Four sub-cores (4.8 cm id × 30 cm) were taken from each such that there was *ca.* 8 cm of sediment (and would be *ca.* 17 cm of water column during incubation).

A teflon-coated stirrer bar was then suspended ~5 cm above the sediment, this was driven by an external rotating magnet rotating at 60–70 rpm. Typically, core tubes from three consecutive sites were collected and incubated for each batch. Incubation bins were filled with seawater such that the cores were immersed, and allowed to equilibrate for 6 h or more. After equilibration, the water level was reduced to below core tube height and experiments commenced with the addition of stock ^{15}N-nitrate to a concentration of 60 µM. The water in each core was stirred for 20 min and sub-samples taken for the analysis of nitrate in order to calculate the final ^{15}N enrichment. Cores were then capped with sealed Perspex lids and incubated in the dark in flowing seawater. Cores were sacrificed at 0, 1.4, 2.5, 4.5 and 6 h. At each time point, 1 ml of 50% zinc chloride was added to the water overlying the sediment before the sediment was gently slurried with the water column using a metal rod, coarser particles were allowed to settle for about a minute before a ~40 ml sample was taken using a gas tight syringe. The sample was then placed in a 12.5 ml Exetainer (Labco, High Wycombe, UK) to which 250 µl of 50% w/v zinc chloride had been added and stored at 4°C until returned to the laboratory.

A headspace of helium was introduced into the Exetainer within 2 weeks and the samples were then analysed within several months. Sample analysis for $^{28}N_2$-, $^{29}N_2$- and $^{30}N_2$-nitrogen gas was carried out using a isotope ratio mass spectrometer (Finnigan MAT delta S) in line with a gas chromatograph (5890 Hewlett Packard). A copper reduction column heated to 640°C was used to remove oxygen from the sample, carbon dioxide and water were removed using a liquid nitrogen cryotrap. Denitrification rates were calculated according to the isotope pairing equations in Dalsgaard et al. (2000).

5.3.4 STABLE ISOTOPE RATIOS

The stable isotope ratios were determined for particulate organic matter in the Gulf of Carpentaria on a voyage on FV *Flinders Pearl* in January 2004, aboard the FV *Northern Pearl in* February 2005, and in

February/March 2005 on the RV *Southern Surveyor*. The following regions were sampled: the southeastern Gulf adjacent to Karumba; Mornington Island; the Vanderlins; Groote Eylandt, Albatross Bay, and offshore from Albatross Bay. For stable isotope analyses in particulate material, triplicate water samples were filtered through precombusted glass fibre filters (Whatman GF/F) to trap the particulate organic matter. Filters were then frozen until returned to the laboratory where they were dried at 60ºC and analysed for δ^{15}N-nitrogen ratios with a continuous-flow isotope-ratio mass spectrometer (Europa Tracermass and Roboprep, Crewe, England).

For sediment stable isotope ratios, box cores were taken at 48 sites across three regions (Mornington Island, Vanderlins and Groote Eylandt) on the RV *Southern Surveyor* voyage described above. Three small cores (31 mm dia.) were taken from one box core at each site for measurement of ^{15}N-nitrogen stable isotope ratios. The sediment was extruded from the core to maintain the integrity of the surface and the top 10 mm sliced off the top. This was frozen until analyzed in the laboratory. Marine sediments are screened through a wire mesh screen to remove rocks and rubble, dried at 60°C and ground before being analysed for δ^{15}N-nitrogen ratios with a continuous-flow isotope-ratio mass spectrometer (Europa Tracermass and Roboprep, Crewe, England).

Grab sediment cores were taken at five sites near Karumba, Groote Eylandt and Weipa on the FV *Flinders Pearl* in January 2004. Mini-cores were inserted into the surface of the sediment and the sediment was extruded from the core to maintain the integrity of the surface and the top 1 cm sliced off the top. The sediment was processed and analyzed as above.

5.3.5 NITROGEN BUDGET

The results of this study were combined with historical data from a number of publications on the oceanography of the Gulf of Carpentaria to generate a nitrogen budget for the Gulf of Carpentaria.

The area of the Gulf was assumed to be 330,000 km^2, with an average depth of 43.4 m, resulting in a volume of 1.4 x 10^{13} m^3. The total pools of nitrate and ammonia in the water column of the Gulf were calculated based on concentrations determined by a number of studies (Rothlisberg *et a*l. 1989, Rothlisberg *et al*. 1994, Burford & Rothlisberg 1999, this study). Annual nitrate and ammonia uptake rates by phytoplankton were determined using the ^{15}N-nitrogen uptake data of Burford & Rothlisberg (1999). Annual areal nitrogen fixation by *Trichodesmium* was determined using the depth-integrated nitrogen fixation rates for *Trichodesmium* blooms measured in this study, and assuming a bloom depth of 1 m. The calculated rates were within the range determined by Capone *et al*. (2005) for the northern Atlantic Ocean. It was assumed that this species was fixing nitrogen at these rates 30% of the year, based on a previous study of the relative abundance of *Trichodesmium* in the Gulf (Burford *et al.* 1995).

Annual nitrogen fixation rates in the seagrass beds were determined by Moriarty *et al*. (1990) and assumed a spatial coverage of 52,000 ha (Roelofs *et al*. 2005). Annual denitrification rates were determined based on this study. River inflows were based on summing the annual river flow volumes (www.nrw.qld.gov.au/) for the 29 largest rivers in the Gulf (A. Brooks, unpubl. data) and multiplying by total nitrogen concentrations in the water column which were determined for the Embley, Flinders and Norman Rivers (M. Burford, unpubl. data).

5.4 RESULTS

In February 2005, mean ammonia, oxides of nitrogen, phosphate and silicate concentrations across three regions and throughout the water column were 0.12 ± 0.16, 0.20 ± 0.27, 0.06 ± 0.06 and 4.51 ± 2.42 µM, respectively (

Table 26). Chlorophyll *a* concentrations were 0.80 ± 0.45 µg L^{-1} and did not show any trends between regions. Denitrification rates measured during the study varied substantially, 0.27 ± 1.39 µmol m^{-2} h^{-1}.

Table 26. Mean (± SD) concentrations for physico-chemical parameters and nutrient rate measurements through the water column in February/March 2005 in the Gulf of Carpentaria. DIN = dissolved inorganic nitrogen.

Parameter	**Value (SD)**
Ammonia (μM)	0.12 (0.16)
Oxides of nitrogen (μM)	0.20 (0.27)
Phosphate (μM)	0.06 (0.06)
Molar DIN:phosphate	12 (16)
Silicate (μM)	4.51 (2.42)
Chlorophyll *a* (μg L^{-1})	0.80 (0.45)
Oxygen (mg L^{-1})	5.98 (0.37)
Water temperature (ºC)	30.61 (0.36)
Denitrification rate (μmol m^{-2} h^{-1})	0.27 (1.39)

The nitrogen fixation rate was determined during a *Trichodesmium* bloom (3,400 trichomes mL^{-1}) in the Mornington Island region in February 2005. Nitrogen fixation rates were highest when samples were incubated at 50% surface light (76 μmol N m^{-2} h^{-1}) compared with 8% (42 μmol N m^{-2} h^{-1}) and 100% (27 μmol N m^{-2} h^{-1}) light. The depth-integrated nitrogen fixation assuming a 1 m deep surface bloom was 578μmol N m^{-2} d^{-1}. The δ^{15}N ratios of *Trichodesmium* during the bloom, and particulate matter in the water column throughout the February/March 2005 voyage were close to 0 - 1 ‰ while the molar C:N ratios were between 3 and 5 (Figure 41). These values were compared with a previous Gulf voyage in January 2004 when no *Trichodesmium* bloom was apparent. The δ^{15}N ratio at a range of sites was substantially higher, 5 to 6.5 ‰, as was the C:N ratio (6.5 to 9).

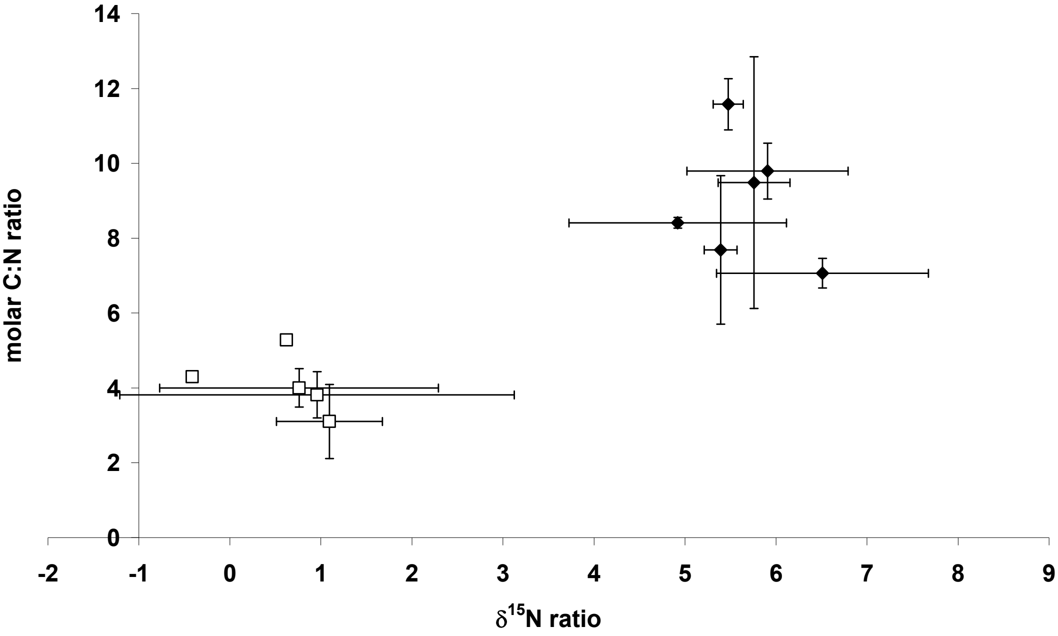

Figure 41. δ^{15}N-nitrogen ratios (± SD) in the particulate organic matter and *Trichodesmium* in the water column during a *Trichodesmium* bloom (open squares) in February/March 2005, and during a non-bloom period in January 2004 (closed diamonds) in the Gulf of Carpentaria

The δ^{15}N ratio was also measured in the sediment during the *Trichodesmium* bloom in February/March 2005 with values ranging from 2.4% to 3.7 ‰ (Figure 42). This compared with values ranging from 3.9 to 6.0 ‰ during two previous Gulf voyages in January 2004 and 2005 when no *Trichodesmium* blooms were apparent. Infaunal δ^{15}N ratios were compared with ratios in particulate organic matter in the water column in both a period of *Trichodesmium* bloom in the southern Gulf (February/March 2005) and a non-bloom period in the northeastern Gulf (January 2005) (Figure 43). There was a wide range of δ^{15}N ratios for the infauna but no obvious differences between the two sampling periods. In contrast, the $\delta^{15}N$ ratios for the particulate organic matter were substantially different between the northeast Gulf in January 2005, and the southern Gulf in February/March 2005.

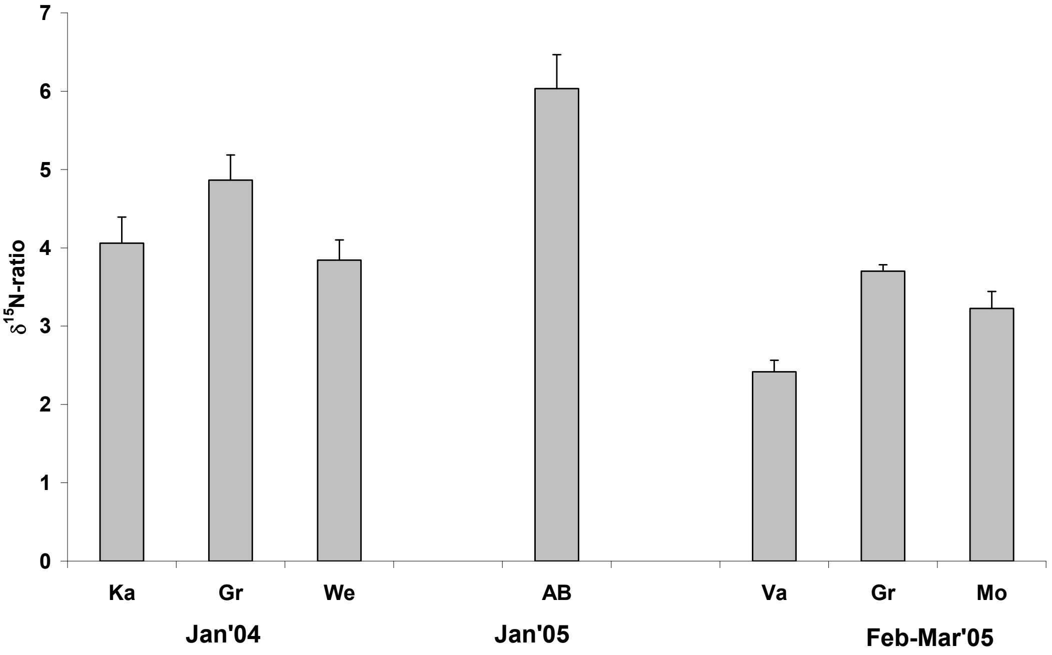

Figure 42. $\delta^{15}N$-nitrogen ratios (± SD) in the sediment on three separate sampling occasions in the Gulf of Carpentaria. Ka = Karumba region, Gr = Groote Eylandt region, We = Weipa offshore region, AB = Albatross Bay, Va = Vanderlins region, Mo = Mornington Island region.

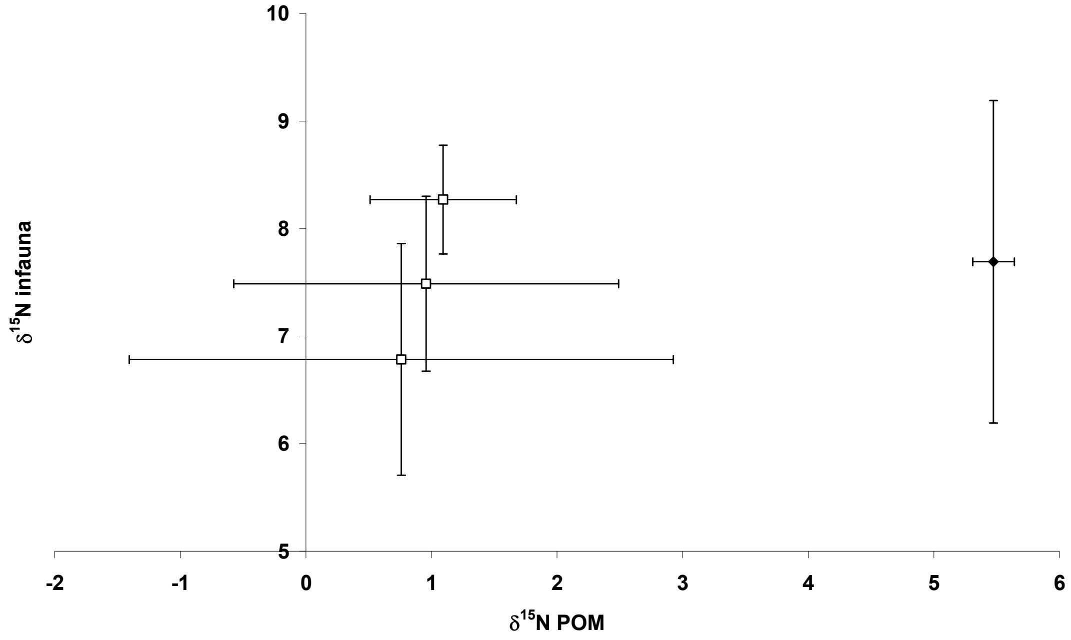

Figure 43. Comparison of the $\delta^{15}N$-nitrogen ratios (± SD) of particulate organic matter and infauna in January 2005 (closed diamonds) and February/March 2005 (open squares).

The nitrogen data from the February/March voyage in the Gulf were combined with data from previous studies in the Gulf to produce a Gulf-wide nitrogen budget (Table 27). The nitrate pool in the Gulf was higher than the ammonia pool, 75 Mt compared with 24 Mt, with nitrate taken up more rapidly by phytoplankton than ammonia (Table 27) (Rothlisberg *et al.* 1989, Rothlisberg *et al.* 1994, Burford & Rothlisberg 1999). However, much of the nitrate was present in bottom waters and therefore during periods of stratification would not be available to phytoplankton.

The importance of nitrogen fixation was determined using two methods: direct measures of nitrogen fixation during a bloom of *Trichodesmium,* and measurement of $\delta^{15}N$ stable isotope ratios in the water column and sediment. Nitrogen fixation from *Trichodesmium* was calculated to be a substantial source of nitrogen into the Gulf at a rate similar to nitrate and ammonia uptake (290 Mt compared with 390 and 260 Mt respectively) based on the acetylene reduction measurements and using a number of assumptions as outlined in the methods (Burford & Rothlisberg 1999). River input loads of nitrogen were determined based on total nitrogen concentrations of 36.17 ± 25.99 µM (Burford, unpubl data). Nitrogen fixation in the seagrass beds was based on previous studies in the Gulf (Moriarty & O'Donohue 1993, Pollard, unpubl. data). In contrast, denitrification rates measured in this study were low compared with N fixation rates.

Table 27. Mean (± SD) pools (Mt N) and fluxes (Mt N y^{-1}) of nitrogen in the Gulf of Carpentaria.

Parameter	**Pools and fluxes**	**Source**
Pools		
Nitrate	75	Rothlisberg *et al.* (1989), Rothlisberg *et al.* (1994), Burford & Rothlisberg (1999), Burford *et al.* (this study)
Ammonia	24	Burford *et al.* (this study)
Fluxes		
Nitrate uptake - phytoplankton	390	Burford & Rothlisberg (1999)
Ammonia uptake - phytoplankton	260	Burford & Rothlisberg (1999)
Nitrogen fixation - *Trichodesmium*	290	Burford *et al.* (this study)
Nitrogen fixation – seagrass beds	47	Moriarty and O'Donohue (1993), Pollard unpubl. data
Denitrification	11	Burford *et al.* (this study)
River inflows		
Total nitrogen	46	Brooks unpubl. Data, Burford unpubl. data

The $\delta^{15}N$ ratios in the particulate organic matter and sediment were used to infer the importance of nitrogen fixation as a nitrogen input. The $\delta^{15}N$ ratio of the particulate organic matter in the water column was depleted during a *Trichodesmium* bloom period (trichome densities up to 3,400 L^{-1} in surface waters) compared with a non-bloom period (Figure 41).

5.5 DISCUSSION

This study is a first attempt to determine a nitrogen budget for the Gulf of Carpentaria, Australia. Our study supports previous studies that suggest that the Gulf is nitrogen limited (Rothlisberg *et al.* 1989, Rothlisberg *et al.* 1994, Burford & Rothlisberg 1999). Dissolved inorganic nitrogen: phosphorus ratios and dissolved inorganic nitrogen concentrations were low. Previous studies in the Gulf have also shown a response of phytoplankton to nitrogen, but not phosphorus addition (Burford & Rothlisberg 1999), and the prevalence of the nitrogen-fixing cyanobacteria, *Trichodesmium,* which typically dominates in low nitrogen environments (Burford *et al.* 1995). Additionally, this study showed that denitrification rates were low compared with other studies in coastal waters, i.e. the tropical Great Barrier Reef, the eutrophic Gulf of Gdansk in Poland and the upwelling regions off Chile (Witek *et al*. 2003, Molina *et al*. 2004, Alongi *et al*. in press). These low rates may reflect low nitrate availability consistent with a nitrogen-limited system. Nitrogen limitation in the Gulf parallels findings in other marine waters around Australia (Condie & Dunn 2006).

River inputs are not major contributors of nitrogen on a whole-of-Gulf scale. However, coastal runoff is trapped within the coastal boundary layer to a water depth of approximately 20 m (Wolanski & Ridd 1990). This coastal area is approximately 20% of the area of the Gulf. In this context, river nitrogen inputs, particularly in the southern Gulf where most of the large river inputs occur, may be significant in fuelling coastal productivity. Within the coastal boundary layer there is a net movement of water from east to west in the Gulf, with water ultimately entering the western Arafura Sea, rather than mixing with the central Gulf. The coastal area of the Gulf is dominated by vast highly productive seagrass beds (Moriarty *et al*. 1990, Roelofs *et al*. 2005). These provide an important food source for biota, including commercially important shrimp species (Loneragan *et al*. 1997). Estimated river nitrogen inputs were comparable with nitrogen fixation in the seagrass beds (Moriarty & O'Donohue 1993) suggesting that these inputs may provide an important source to fuel seagrass and epiphyte production.

Although nitrogen from river sources may be important to coastal production, the presence of the coastal boundary current means that it is likely that little of this source of nitrogen is reaching the deeper waters of the Gulf. Although previous studies have not tracked riverine nitrogen offshore, studies of $\delta^{13}C$ ratios in particulate organic matter throughout the deeper waters of Gulf found no evidence of a terrestrial signal (Rothlisberg *et al*. 1994, Burford, unpubl. data). Additionally, a study of food sources for estuarine shrimp in the Gulf showed no evidence of mangrove and terrestrial carbon exported from estuaries contributing to offshore food webs (Loneragan *et al*. 1997).

This study has shown that nitrogen fixation can be a significant process in the Gulf, as demonstrated by the measurement of high rates of fixation and the depleted $\delta^{15}N$ ratios in the particulate matter in the water column during a *Trichodesmium* bloom. It has been suggested that depleted $\delta^{15}N$ ratios in marine waters can be the result of deep water nitrate however recent studies in other systems do not support this (Montoya *et al*. 2002). Previous studies in the Gulf have shown the net phytoplankton community is composed principally of the nitrogen-fixing cyanobacteria *Trichodesmium* and large diatoms (Hallegraeff & Jeffrey 1984, Rothlisberg *et al*. 1994, Burford *et al*. 1995), and that primary productivity rates are relatively high (Motoda *et al*. 1978, Rothlisberg *et al*. 1994, Burford *et al*. 1999). *Trichodesmium* is typically numerically most dominant in the summer months when stratification occurs (Burford *et al*. 1995). Conservative assumptions about the annual rates of nitrogen fixation from data in our study were made based on the information regarding seasonal abundance (Burford *et al*. 1995). Rates of nitrogen fixation measured in February/March 2005 during a *Trichodesmium* bloom (3,400 trichomes L^{-1} in surface waters) were comparable with those in other tropical waters (Capone *et al*. 2005). More recently, *Trichodesmium* has been shown to be more abundant at depth than previously thought, due to sampling artefacts (Davis & McGillicuddy 2006). Therefore it is likely that rates may be significantly higher than previously thought. Previous studies in Australian waters suggest that the main factor likely to be limiting *Trichodesmium* growth is iron availability (Sohm *et al*. 2006).

The rates of nitrogen fixation for the Gulf do not take into account the role of nitrogen-fixing symbiotic cyanobacteria typically associated with diatoms, and small unicellular cyanobacteria which have recently been shown to contribute significantly to global nitrogen fixation rates (Carpenter *et al*. 1999, Zehr *et al*. 2001, Montoya *et al*. 2004, Capone *et al*. 2006). Previous studies in the Gulf have shown that diatoms with symbionts are numerically dominant, e.g. *Rhizosolenia* with *Richelia* (Burford *et al*. 1995). Therefore, nitrogen inputs from fixation could be greater than the estimates calculated from this study.

Previous studies have shown that *Trichodesmium* is poorly grazed by zooplankton (Mulholland 2007). Studies in the Gulf have also shown that it is a poor nutritional source for shrimp larvae because the sheath surrounding the trichome is difficult to digest, resulting in the passing of intact trichomes through the gut. (Preston *et al*. 1998). However, *Trichodesmium* may contribute to nitrogen cycling via release of dissolved organic nitrogen and ammonia from cells (Glibert & Bronk 1994, Mulholland

2007) or viral lysis (Hewson *et al.* 2004). Alternatively the detrital material may be deposited on the sediment as it senesces. In our study, lower $\delta^{15}N$ ratios were measured in the sediment during a *Trichodesmium* bloom compared with non-bloom periods. Other studies have found isotopically light $\delta^{15}N$ ratios in sediment traps and sediment during *Trichodesmium* blooms (Capone *et al.* 1998, Knapp *et al.* 2005). Infaunal $\delta^{15}N$ ratios were not depleted during this time, suggesting that detrital material from the bloom was not substantially grazed by infauna and is therefore likely to be subject to microbial degradation.

During periods of water column stratification, particularly during summer months when wind mixing is low, a higher concentration of nitrate has been observed in bottom waters (Rothlisberg *et al.* 1989, Rothlisberg *et al.* 1994, this study). This suggests that remineralization and nitrification of nitrogen is occurring in or near the sediment. It is likely that deposition of nitrogen-rich cyanobacterial detritus generated from nitrogen fixation is an important source of nitrogen fuelling these bacterial processes. During winter months when the wind strength increases, the water column may become completely mixed and dissolved nitrogen from bottom waters becomes evenly distributed through the water column (Burford & Rothlisberg 1999). Therefore atmospheric nitrogen inputs via nitrogen-fixing cyanobacteria may create a coupling between water column and sediment processes, and hence also fuel primary productivity.

Net exchange of nutrients from the Arafura Sea is unlikely to be substantial given the similarity in the nutrient concentrations in the Gulf and the Arafura Sea (Condie & Dunn 2006). Indeed, the deeper waters of the Gulf and the Arafura Sea have been classified within the same biogeographical region based on their physico-chemical and primary productivity characteristics (Condie & Dunn 2006). Additionally, water circulates around a central gyre in the Gulf meaning that the water residence time in the deeper waters of the Gulf is in the order of 3 y (Wolanski & Ridd 1990, Condie & Dunn 2006). The relatively low exchange of water in the northern Gulf, lack of difference between nitrogen concentrations entering and leaving the Gulf (Condie & Dunn 2006) and low rates of denitrification mean that little nitrogen is likely to be lost from the system.

The Gulf is a relatively pristine ecosystem, with the main nutrient inputs in the catchment being natural erosion processes, and low intensity agriculture (principally cattle grazing). Future agricultural development is proposed for the Gulf catchments including irrigation, damming of rivers and other intensive forms of agriculture (e.g. ponded pastures and cotton production). The change in nutrient loads that may result from some of these activities could impact on coastal waters, although they are unlikely to impact on the deeper waters of the Gulf. For example, damming of rivers and regulation of flow may reduce nutrient loads hence ddecreasing coastal production and negatively impacting on coastal fisheries. Conversely intensive forms of agriculture are likely to increase nutrient loads, in turn augmenting estuarine and coastal production with the inherent problems of anoxia and harmful algal blooms.

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APPENDIX 6 ACCUMULATED EFFECTS OF TRAWLING ON THE TROPHIC ECOLOGY OF TROPICAL DEMERSAL FISH ASSEMBLAGES

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6.1 SUMMARY

Have 40 years of penaeid prawn (shrimp) trawling in the Gulf of Carpentaria, Northern Australia, had any large-scale ecological effects on benthic communities? As a first step to answering this question, we compared stomach contents of demersal fishes in three regions where sites had experienced ‘high’ and ‘low’ fishing intensity effort. The sites were identified from Vessel Monitoring System data. The diets of eight benthic predator fish species, across the regions and fishing intensities, were compared by analysing 3476 stomachs; of which 2036 contained content. Seven predator species had significantly different diets between the regions. Five species had significantly different diets between the high and low fishing intensities at various regions, while another species had an overall significant difference between fishing intensities, regardless of region. Analysis of predator diets, between areas of different fishing intensity, can serve as a useful tool that can be used in conjunction with other methods to identify trophic changes in the benthic community over a long period of exposure to intensive fishing pressure. The present study also provides valuable diet information to improve food-web modelling tools needed to support ecosystem based fisheries management.

Keywords: fishing intensity; benthic community; diet; stomach contents; fishing disturbance

6.2 INTRODUCTION

The potential for fishing to impact negatively on the structure and functionality of marine ecosystems has been the main impetus for the growing interest in ecosystem approaches to fisheries management in recent years (Hall 1998, Pauly et al. 1998, FAO 2005). Several studies have reported that not only target species in fisheries have declined, but so have other species caught incidentally as bycatch (Jennings & Kaiser 1998, Jackson et al. 2001). Trawl fisheries in particular may have the greatest potential to impact marine communities, for their gear is not selective and captures a large variety of mobile animals. Furthermore, demersal trawl gear such as those used in penaeid prawn fisheries, often make contact with the sea floor and remove sessile biota and alter structural heterogeneity, which many mobile animals use for feeding and shelter (Jennings et al. 2005). Several authors have demonstrated that the reduction of habitat spatial heterogeneity and physical removal of particular species by fishing can significantly alter the structure of benthic marine communities (Frid & Hall 1999, Bianchi et al. 2000, Jennings et al. 2002, Jennings et al. 2005).

In the Gulf of Thailand, 25 years of intensive multi-species trawl fishing reduced the average trophic level of the ecosystem from 3.35 to 3.15 (Christensen 1998). The practice of fishing higher trophic level species and progressively targeting species in lower trophic levels, or ‘fishing down the food web’, can result in a proliferation of species with high production rates, like penaeids, but also other taxa that often have low economic importance. This can result in substantial shifts in ecosystem structure, with flow-on negative economic consequences for fisheries (Carscadden et al. 2001, Jackson et al. 2001, Walters & Kitchell 2001, Daskalov 2002). For example, depletion of predatory sharks in the western Atlantic Ocean caused a dramatic trophic cascade throughout the system that brought about the collapse of the bay scallop fishery (Myers et al. 2007). Similarly, trophic cascades have been documented in the tropical Pacific Ocean where declines in apex pelagic predators caused an increase of pelagic stingrays (Ward & Myers 2005). Although fishing down ecosystems changes ecosystem structure by top-down pressure, impacts on intermediate and lower trophic levels can have cascading effects either upward or downward through the system. For example, Rothlisberg & Okey (2006), predicted that doubling the fishing mortality rate of tiger and banana prawns in Albatross Bay, Australia, over a 20 year period would lead to > 90% declines in the biomass of several groups of predatory sharks and teleosts. Trophic cascades can also occur in both directions in "wasp-waist" controlled systems, where a few highly productive species in the intermediate trophic levels comprise the bulk of the biomass in the system and act as both key prey and predators (Bakun 1996, Cury et al. 2000).

Changes in species composition is often the most obvious effect of fishing on ecosystems, but in some diverse tropical communities changes in species composition may be less pronounced or more difficult to detect, since many species caught in surveys are naturally rare (Heales et al. 2000, Stobutzki et al. 2001, Tonks et al. 2008, Dell et al. 2009). Changes in size spectra can also be an important indicator of the accumulated impacts of fishing on marine ecosystems (Rice & Gislason 1996, Bianchi et al. 2000, Jennings et al. 2002, Shin & Cury 2004). However, this indicator requires a long time-series of data to elucidate trends of fishing effects beyond the generation time of affected animals, which is often difficult or expensive to collect, especially for low-value species caught as bycatch (Griffiths et al. 2006, Zhou & Griffiths 2008). The use of benthic organisms, as an indicator of trophic changes in the benthic community, is a particularly useful approach applied in a number of scenarios (e.g. Kaiser 1996, Callaway et al. 2007). Stomach content analysis has been used to infer changes in prey availability, which may arise from fishing activities (Frid & Hall 1999, Serrano et al. 2003, Hinz et al. 2005, de Juan et al. 2007). For example, in an area of the north-western Mediterranean Sea exposed to commercial demersal trawling, de Juan et al. (2007) found that a starfish (*Astropecten irregularis*) and flat fish (*Citharus linguatula*) both had significantly different diets in fished and unfished regions. They also found that both animals had higher feeding rates in fished regions because trawl disturbance increased the availability of benthic prey.

The Northern Prawn Fishery (NPF) is a multi-species demersal trawl fishery that has operated continuously since the early 1970s. It is one of Australia's largest (~ 250 000 km^2) and most valuable fisheries (Dichmont et al. 2007). In the 17 years of records, the fishery peaked at AU $164.7 m in 2000/2001 and for 2007/2008 was worth AU $74.1 m (historic non-adjusted values; Brown et al. 2002, Pham & Peat 2009). This fishery has been the centre of controversy over its impacts on the benthos and large megafauna such as turtles. However, the NPF has been through proactive management, consistently addressing concerns over its sustainability and also unseen and indirect impacts on particular species and the broader ecosystem (Brewer et al. 2008). Some of the actions taken include a (i) compulsory use of turtle excluder devices and bycatch reduction devices since 2000 (Brewer et al. 2008, Milton et al. 2009); (ii) a reduction in fleet size and Statutory Fishing Rights from a peak of 286 vessels (Robins & Somers 1994) to 52 fishing vessels (Pham & Peat 2009); (iii) establishment of temporal and spatial closures (Kenyon et al. 2005) (iv) adoption of harvest strategies and biological and economic targets (DAFF 2007, Dichmont et al. 2008); and (v) establishment of a strong industry, science and management partnership; including the current trial of a co-management system (Dichmont et al. 2007, AFMA 2009).

Despite the large size of the fishing grounds, trawling takes place in less than 10% of the fishery's managed area (Zhou & Griffiths 2008). The continual reduction in fishing season and effort have generally forced fishers to fish 'hot spots' rather than areas where catches have been historically variable. Within these hot spots fishing is highly aggregated in small areas, in which vessels repeatedly trawl the same path. This method is known as "line fishing", which has cumulative effects on the structure of benthic communities (Haywood et al. 2005).

In the present study we explore the local and regional ecological effects of trawling in Australia's NPF. This is achieved by comparing the trophic ecology of selected benthic feeding species of fish in three trawled regions, each characterised by its recent history of fishing range (i.e. exposed to high and low fishing intensity). This study provides new data and will improve the knowledge of the direct and indirect effects of fishing on the diet of common benthic predatory teleosts that occur in trawl bycatch.

Our results will contribute to the understanding of the trophic pathways of the benthic food web connecting different components of this tropical benthic ecosystem.

6.3 MATERIALS AND METHODS

6.3.1 STUDY AREA AND SURVEY DESIGN

The present study was part of a scientific survey conducted onboard the research vessel RV Southern Surveyor from 28 February to 15 March 2005. The survey area was in tiger prawn (Penaeus semisulcatus and Penaeus esculentus) fishing grounds of the tropical Northern Prawn Fishery in the Gulf of Carpentaria. The study sites were situated within three of the main historical prawn fishing regions: (i) north of Groote Eylandt; (ii) northeast of Vanderlin Island; and (iii) north of Mornington Island (Figure 44). These regions were selected based on the relative within-region homogeneity of the physical benthic environment of trawlable areas; i.e. sediment, substrate, hydrology and bathymetry (Somers & Long 1994, Hill et al. 2002, Ridgway et al. 2002, Passlow et al. 2004, Webster & Petkovic 2005).

To explore the broad-scale direct and indirect effects of historical prawn trawling on the benthos, the survey design was a natural experiment; i.e. sampling sites were distributed among areas of high and low fishing effort (approximating control and impact sites), but there was no time component (samples before and after the impact). The design was repeated in each of the three regions described above. There were no true control areas because all comparable areas have been trawled in the past or recently. Untouched areas are usually either shallow (<15 m) inshore areas supporting very different communities or are interspersed with reefs and seagrasses (Blaber et al. 1994). Fishing intensity was quantified by using Vessel Monitoring System (VMS) data from 1999 to 2003 and applying the method of Haywood et al. (2005). This allowed fishing levels to be assigned to each one-minute cell of a grid covering the NPF, where the low fishing level was assigned to grid cells exposed to an average of less than one boat hour of trawling per year over the five years. The high fishing level was assigned to cells that were exposed to: (i) an average of at least 24 boat hours of trawling per year; and (ii) at least six boat hours of trawling in four of the five years, including the last year (2003). For each combination of region and fishing level, replicate sampling sites were randomly assigned to one-minute grid cells in that region and fishing intensity, and then manually edited to avoid undesirable and non-comparable habitats (mostly hard bottom) and to eliminate excessive random clumping.

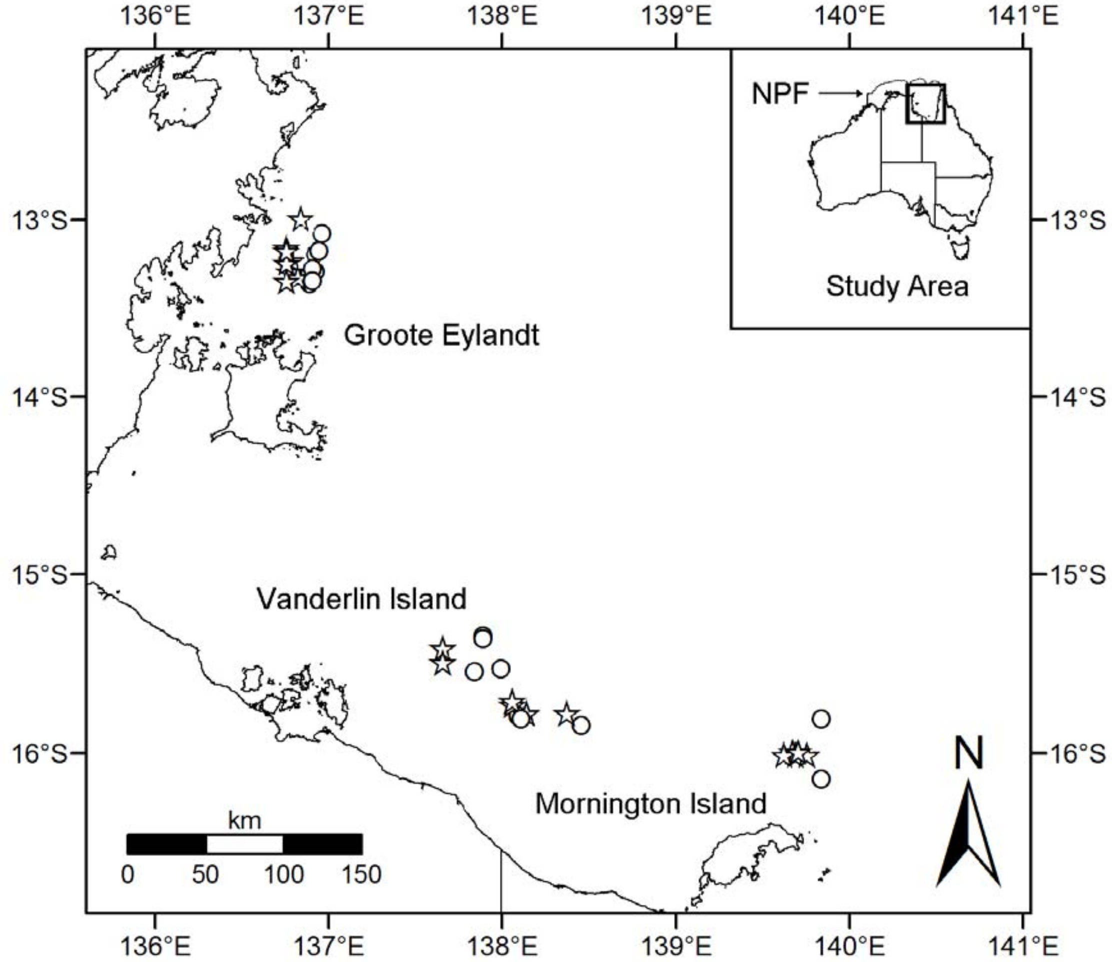

Figure 44. Target sites for the natural experiment component of the 2005 research voyage in the southern and south-western Gulf of Carpentaria, Australia. The star symbols indicate high fishing intensity sites; the circle symbols indicate low fishing intensity sites. The Northern Prawn Fishery managed area is shown in the inset as 'NPF'.

6.3.2 SAMPLE COLLECTION AND PROCESSING

Trawl samples were taken with a standard industry 'Florida Flyer' style NPF prawn net, supplied by one of the industry net-makers. The net had a headrope length of 21.2 m (12 fathoms), with a diamond net pattern of 55 mm (~2 inch) stretch-mesh size and Bison #9 trawl boards. The net was towed over the seabed at a median speed of 1.6 m/s for a median distance of 1064 m.

The predator fish species chosen for dietary analysis were those that were benthic feeders and representative of the demersal fish assemblage most likely to be affected by trawling (Harris & Poiner 1991, Pender et al. 1992, Blaber et al. 1994, Stobutzki et al. 2001). The fish samples were frozen at sea (–20˚C) and later freighted to CSIRO laboratories where they were thawed and their stomachs removed. Prey items were extracted from small teleost stomachs and identified under a binocular microscope to the lowest taxonomic level possible. The items termed 'Natantia' were Decapods that were prawn or shrimp like as per Grey et al. (1983). The prey were then weighed (wet weight, 0.000 g), counted and measured where possible (mm). The stomach contents of each predator species were examined in detail to provide a complete list of identified prey taxa, prey biomass and prey frequency of occurrence data. The numbers of stomachs with content for each predator species, per region and fishing intensity, are presented in (Table 28.)

Table 28. The eight predator fish species used for diet analysis, the number of stomachs analysed, and the percentage with content. The count of stomachs with content is presented for each region (Groote, Mornington and Vanderlin) and fishing intensity (high and low) in the Gulf of Carpentaria. The numbers in brackets indicate mean standard length.

					Groote		Mornington		Vanderlins	
Species	**Family**	**Standard Length Range (mm)**	**Stomachs analysed (n)**	**% With Content**	**High**	**Low**	**High**	**Low**	**High**	**Low**
Caranx bucculentus	Carangidae	94–520 (146)	255	66.7	38	28	49	38	—	—
Nemipterus hexodon	Nemipteridae	57–205 (118)	410	46.6	58	55	49	23	—	—
Pentaprion longimanus	Gerreidae	53–110 (83)	493	67.7	75	49	48	39	79	44
Priacanthus tayenus	Priacanthidae	89–223 (112)	315	93.0	50	52	46	55	38	52
Saurida macrolepis	Synodontidae	74–218 (145)	416	62.3	54	53	53	42	21	36
Selaroides leptolepis	Carangidae	67–170 (126)	602	41.0	52	41	56	22	49	27
Terapon theraps	Terapontidae	99–180 (127)	275	81.8	58	55	49	50	—	—
Upeneus sulphureus	Mullidae	64–134 (99)	710	44.6	63	62	53	52	40	47

A stomach fullness index was calculated for each predator in order to explore whether there was variation in the biomass of prey consumed in different regions and fishing intensities. This was calculated using the equation:

$$\text{Stomach fullness index} = \frac{\text{total wet weight of prey}}{(\text{predator wet body weight}) - (\text{total wet weight of prey})}$$

6.3.3 DATA ANALYSIS

Statistical comparisons of the diets of the selected predator fish species were made between regions and fishing intensities using multivariate analysis procedures in PRIMER (Version 6.1.10, Clarke & Warwick 2001). In preparation for the multivariate analyses, the stomach samples were randomly allocated to one of four replicates for each trawl region and fishing intensity per predator species. This was done to incorporate the natural variability in diet composition that often occurs between individual stomach samples. Prey items were represented in terms of biomass as percentage wet weight.

The data was square-root transformed to reduce the influence of highly weighted taxa and a Bray-Curtiss similarity index was constructed for the prey biomass (%) for each replicate. The resulting predator-prey matrices were then analysed using the non-metric multidimensional scaling (nMDS) and analysis of similarities (ANOSIM) to investigate differences in diet between species, between regions for each species, between fishing intensities at each region, and between fishing intensities over all regions. Similarity percentages (SIMPER) were used to identify the prey responsible for significant differences between *a priori* groups as determined by ANOSIM. This procedure identified the prey taxa that contributed to the dissimilarity between groups (high and low fishing intensity), and the similarity within each group (at each fishing intensity).

Apart from investigating differences in diet composition between regions and fishing intensity for each species, it was of interest to investigate whether the total amount (wet weight) of prey consumed by fish in high and low fishing intensity areas differed. As some species were not caught in sufficient numbers in some regions to facilitate a balanced full-factorial design, separate two-factor ANOVAs were performed for each species. This allowed testing for differences in the mean stomach fullness index between regions (Groote, Mornington and Vanderlin) and fishing intensities (high and low). Both factors were considered fixed. Cochran's and Shapiro-Wilk's tests were used to analyse homogeneity of variances and normality of the data, respectively (Zar 1984). Data for each species were $\log_{10}(x + 1)$ transformed before analysis in an attempt to stabilise heteroscedastic variances, which was not successful in all cases. Following the recommendations of Underwood (1981), analyses were undertaken on the transformed data, with alpha set to 0.01 to minimize the chances of incurring Type I errors. Tukey's Honestly Significant Difference (HSD) tests were used for *a posteriori* comparison of means.

6.4 RESULTS

6.4.1 AREAS SELECTED FOR FISHING INTENSITY COMPARISONS

The median of the average yearly trawling intensity from 1999 to 2003 at the sample locations was 0.2 and 43 h/n.mile^2 for samples at low and high fishing intensity sites respectively (interquartile ranges 0.1–0.7 and 33–66 h/n.mile^2 respectively).

Samples at high fishing intensity sites tended to be from depths close to or slightly shallower than those at low fishing sites (median 41 and 43 m respectively); the greatest difference was at the Groote region (median 35 for the high and 43 m for the low fishing intensity levels).

6.4.2 DIET COMPOSITION AND SPATIAL VARIATION

For the dietary analysis, eight predator fish species with prey in their stomachs were caught in sufficient numbers for comparison between the fishing intensities in at least two of the three regions (Table 28). A total of 3476 stomachs of these species were examined; 2036 (59%) contained prey. Table 32 contains all the percentage biomass (%B) and percentage frequency of occurrence (%F) of each prey taxon identified for each of the eight predator fish species; across all regions and fishing intensities in the Gulf of Carpentaria, Australia.

The first ANOSIM results were for comparisons to investigate differences in diet between the eight predators, regardless of regions or fishing intensities. For this analysis, there were significant differences in diet between the predator species (Global R 0.667; all p values for the comparisons <0.01), except for the comparison between *Nemipterus hexodon* and *Terapon theraps*.

The diets of each predator species were then compared to one another in each region (i.e. interspecific comparisons regardless of fishing intensity). The nMDS ordinations suggested that for most fish predator species, their diets differed from one another (Figure 45). This was confirmed in the ANOSIM results for comparisons of predator species for each region, across all fishing intensities. Except for the comparisons between *N. hexodon* and *T. theraps* diets at Mornington, all the predator species had significantly different diets in each region (Groote Global R 0.719, all $p < 0.01$; Mornington Global R 0.704, all $p < 0.05$; Vanderlin Global R 0.712, all $p < 0.05$). This was the same trend as for all the regions combined.

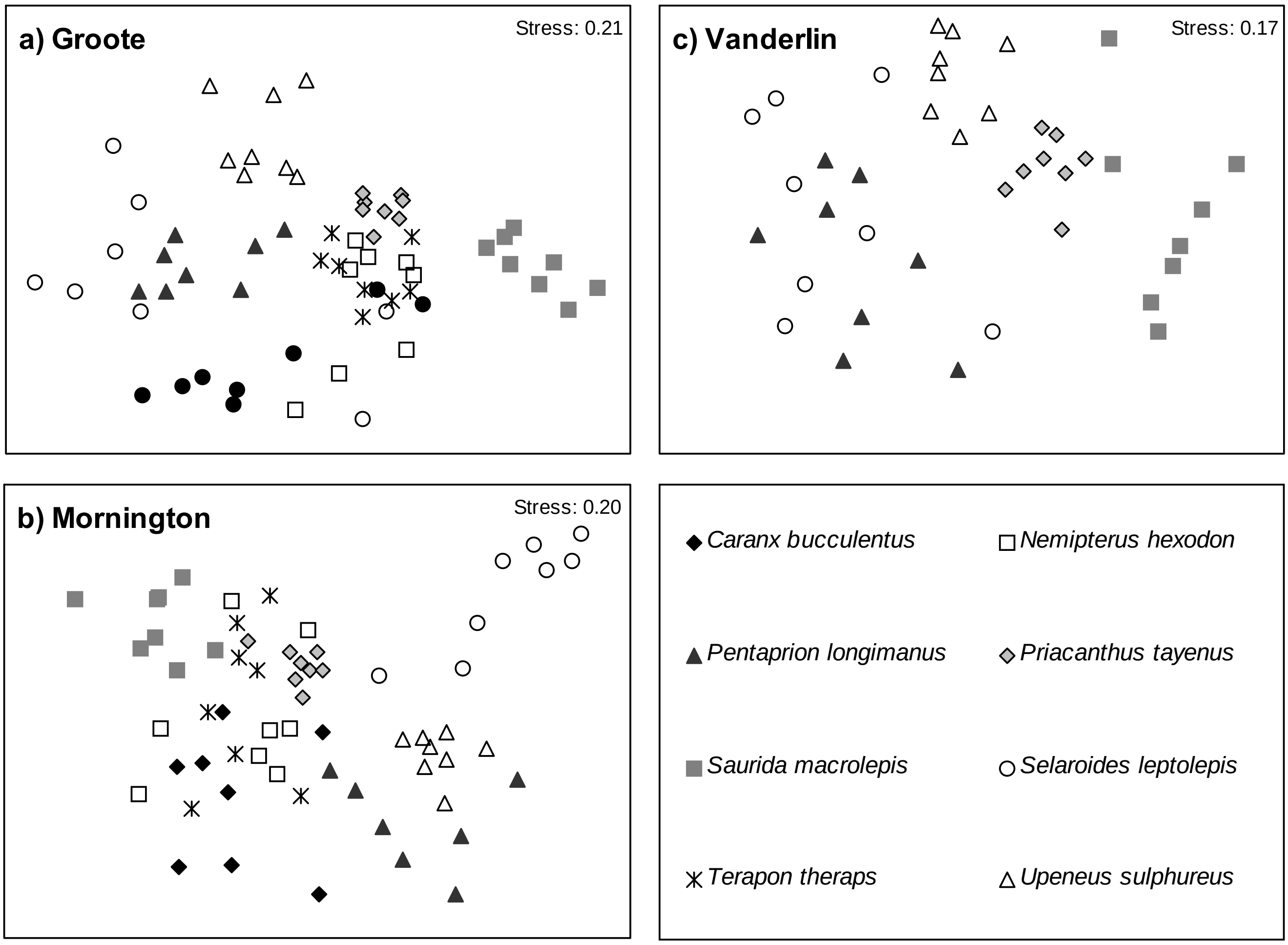

Figure 45. Non-metric multidimensional scaling ordination comparing diets of the eight predator fish species at three regions within the Gulf of Carpentaria.

When the diets of each predator species were compared between regions (i.e. intraspecific comparisons regardless of fishing intensity), every species except *N. hexodon* had a significantly different diet between regions (Table 29).

Table 29. Comparison of a predator's diet between regions. The Global *R* and Significance results are for the overall comparisons (difference between regions).

Predator Species	**Global R**	**Regions significantly different to one another**	**Significance**
Caranx bucculentus[#]	0.604	Gro, Mor	**
Nemipterus hexodon[#]	0.406	-------------------	NS
Pentaprion longimanus	0.657	Gro, Mor, Van	***
Priacanthus tayenus	0.405	Gro, Mor, Van	***
Saurida macrolepis	0.418	Gro, Mor, Van	***
Selaroides leptolepis	0.655	Gro, Mor, Van	***
Terapon theraps[#]	0.745	Gro, Mor	***
Upeneus sulphureus	0.416	Gro, Mor, Van	***

# indicates species for which a comparison with the Vanderlin region was not possible

NS Not Significant; * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$. Gro = Groote; Mor = Mornington; Van = Vanderlin

6.4.3 EFFECTS OF FISHING INTENSITY ON DIET COMPOSITION

Comparison of the predator diets at each region indicated that five species had significantly different diets between the high and low fishing intensities at various regions for each species (Table 30). *Nemipterus hexodon* had an overall difference between high and low, but not in any one region. The nMDS ordinations, of those five species with significant diet differences, showed separation between fishing intensities in each of the regions identified in Table 30. These predators were investigated further to find which prey taxa contributed to the differences (dissimilarity) between high and low fishing intensities for a predator at a region Figure 47. The prey taxa at a dissimilarity of 66% accounted for over 88% of the prey biomass for *Pentaprion longimanus*, *Terapon theraps* and *Upeneus sulphureus* at Groote; over 84% of the prey biomass for *P. longimanus*, *Selaroides leptolepis*, *T. theraps*, and *U. sulphureus* at Mornington; and over 76% of the prey biomass for *Saurida macrolepis* and *S. leptolepis* at Vanderlin Figure 47.

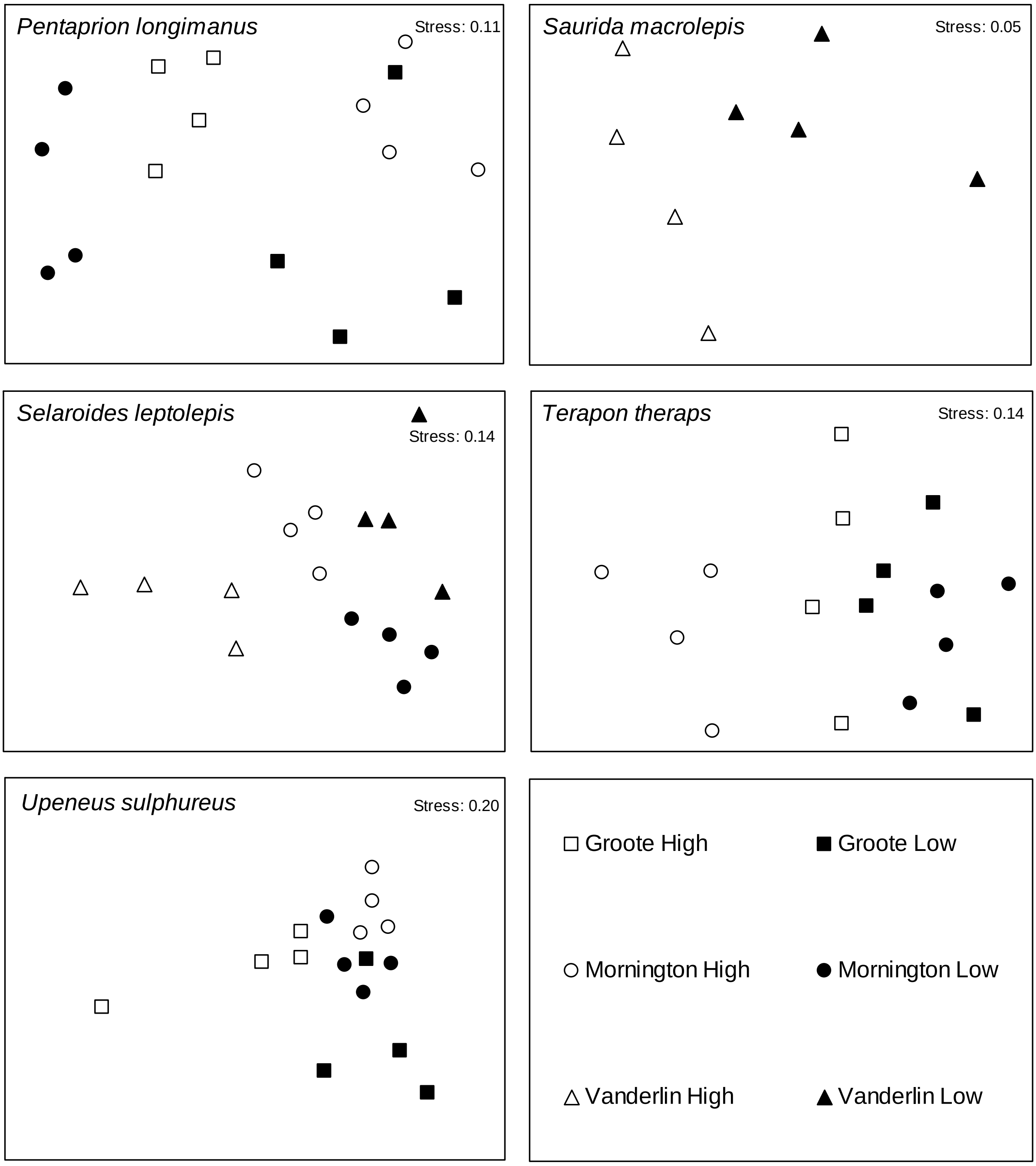

Figure 46. The non-metric multidimensional scaling ordinations by predator diet. Comparisons are between high and low trawling intensity for a particular region (for species and regions where significant differences were detected). Each symbol represents a replicate group of stomachs that contained prey.

Table 30. Regions where there were significant differences in a predator species' diet between high and low fishing intensities. Global *R* and Significance results are for the overall comparisons (difference in fishing intensity over all regions).

Predator Species	**Global R**	**Regions with differences between 'High' & 'Low'**	**Significance**
Caranx bucculentus	0.115	—	NS
Nemipterus hexodon +	0.234	—	*
Pentaprion longimanus	0.635	Gro, Mor	***
Priacanthus tayenus	0.101	—	NS
Saurida macrolepis	0.184	Van	*
Selaroides leptolepis	0.722	Mor, Van	***
Terapon theraps	0.698	Gro, Mor	**
Upeneus sulphureus	0.396	Gro, Mor	***

+ indicates no significant difference in any one region, but a significant difference across all regions

NS Not Significant; * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$. Gro = Groote; Mor = Mornington; Van = Vanderlin

For the diet differences in high and low fishing intensity areas, most of the total dissimilarity (range 81.6–100% in all regions and fishing intensities) was accounted for by benthic organisms (i.e. prey taxa excluding scyphozoa and teleosts). The exception was *S. macrolepis*, which had a greater component of teleost diet (44.1% of the total dissimilarity accounted for by benthic organisms). Of those prey items contributing toward 66% of the dissimilarity, a variety of Crustacea were present in all nine predator/region combinations, while at least one or more other benthic prey (species of Annelida, Echinodermata and Mollusca) were present in various combinations in all nine predator/region combinations (Figure 47). Teleosts contributed to the 66% dissimilarity for *P. longimanus*, *S. macrolepis* and *T. theraps*, but not for *S. leptolepis* and *U. sulphureus*. However, teleosts were contributors within 90% dissimilarity for *S. leptolepis* at Groote and Mornington, and for *U. sulphureus* at Groote.

For the predator *N. hexodon* there were no significant regional diet differences (and consequently it is not plotted as per the other predators). The fishing intensity comparisons, irrespective of region, indicated that this predator had a total of 93.6% of the total dissimilarity accounted for by benthic organisms. There were 13 prey taxa that contributed to 66% of the dissimilarity between high and low fishing intensities: Teleost, Polychaeta, Stomatopoda, Portunidae, Brachyura, Goneplacidae, Pilumnidae, Octopoda, Engraulidae, Natantia, Decapoda, Penaeidae and Holothuroidea. For *N. hexodon*, these 13 prey taxa collectively account for 89% of the prey biomass at high fishing intensity, and 80% of the prey biomass at low fishing intensity.

6.4.4 EFFECTS OF FISHING INTENSITY ON PREDATOR CONSUMPTION

The results of two-factor ANOVAs did not show significant region x fishing intensity interactions(Table 31), indicating that any variation in the mean stomach fullness index (SFI) could be attributed to the main effects alone (region or fishing intensity). Mean SFI differed significantly among regions for all species (). However, the regions where fish had the highest SFIs differed among species (Figure 48). For example, SFIs were significantly higher at Mornington than other regions for *Caranx bucculentus*, *N. hexodon*, *P. longimanus* and *S. leptolepis*, while SFIs were highest at Groote for *S. macrolepis*. Vanderlin generally had the lowest SFIs, except for *Priacanthus tayenus* (Table 31).

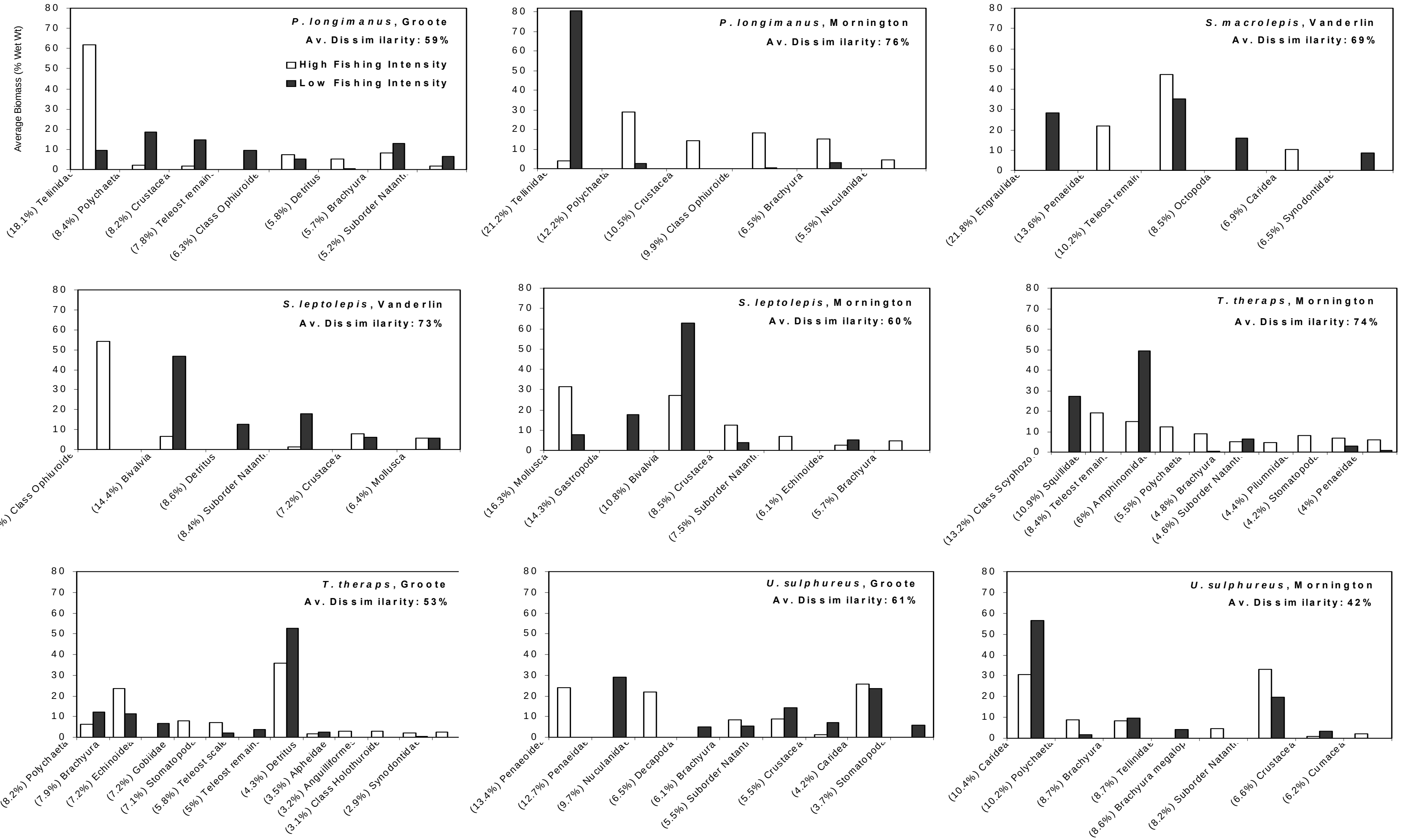

Figure 47. The dissimilarity and biomass of prey for each predator fish species that had significant differences in diet between high and low trawling intensities. The contribution to dissimilarity (%) is in parenthesis; prey taxa are listed in descending order of contribution to dissimilarity (left to right). The average (Av.) dissimilarities are indicated for each species in a region. At a dissimilarity of 66%, over 75% of the prey biomass for all these predator species is accounted for.

Table 31. Results of two-factor fixed ANOVAs for eight predator fish species; testing for differences in the mean stomach fullness between regions (Groote, Mornington and Vanderlin) and fishing intensities (high and low) in the Gulf of Carpentaria. Mean squares are shown. Degrees of freedom are shown in parentheses, although for species denoted by # degrees of freedom for region = 1 and region x fishing intensity = 1 (since they were caught in sufficient numbers only in two regions). Data for each species were log10 (x + 1) transformed before analysis, which did not remove heteroscedastic variances. Alpha was set to 0.01 to reduce the chances of incurring type II errors.

Species	**Region (2)**	**Fishing Intensity (1)**	**Region x Fishing Intensity (2)**	**Residuals**
Caranx bucculaentus #	**0.966****	0.019	0.354	0.597
Nemipterus hexodon #	**1.588****	0.178	0.579	0.631
Pentaprion longimanus	**1.310*****	0.059	0.784	0.828
Priacanthus tayenus	**0.933****	**2.271*****	0.51	1.181
Saurida macrolepis	**9.427*****	0.28	1.543	4.393
Selaroides leptolepis	**0.364*****	0.06	0.175	0.371
Terapon theraps #	**1.273***	0.247	1.043	0.818
Upeneus sulphureus	**0.599*****	0.748	1.931	1.107

Significant results are in shown bold and significance levels shown as: * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$

Mean SFI was not significantly different between high and low fishing intensity in any region for seven of the eight species analysed (Table 31). The exception was *P. tayenus*, which had significantly higher mean SFI at low fishing intensity, when data were combined across all regions.

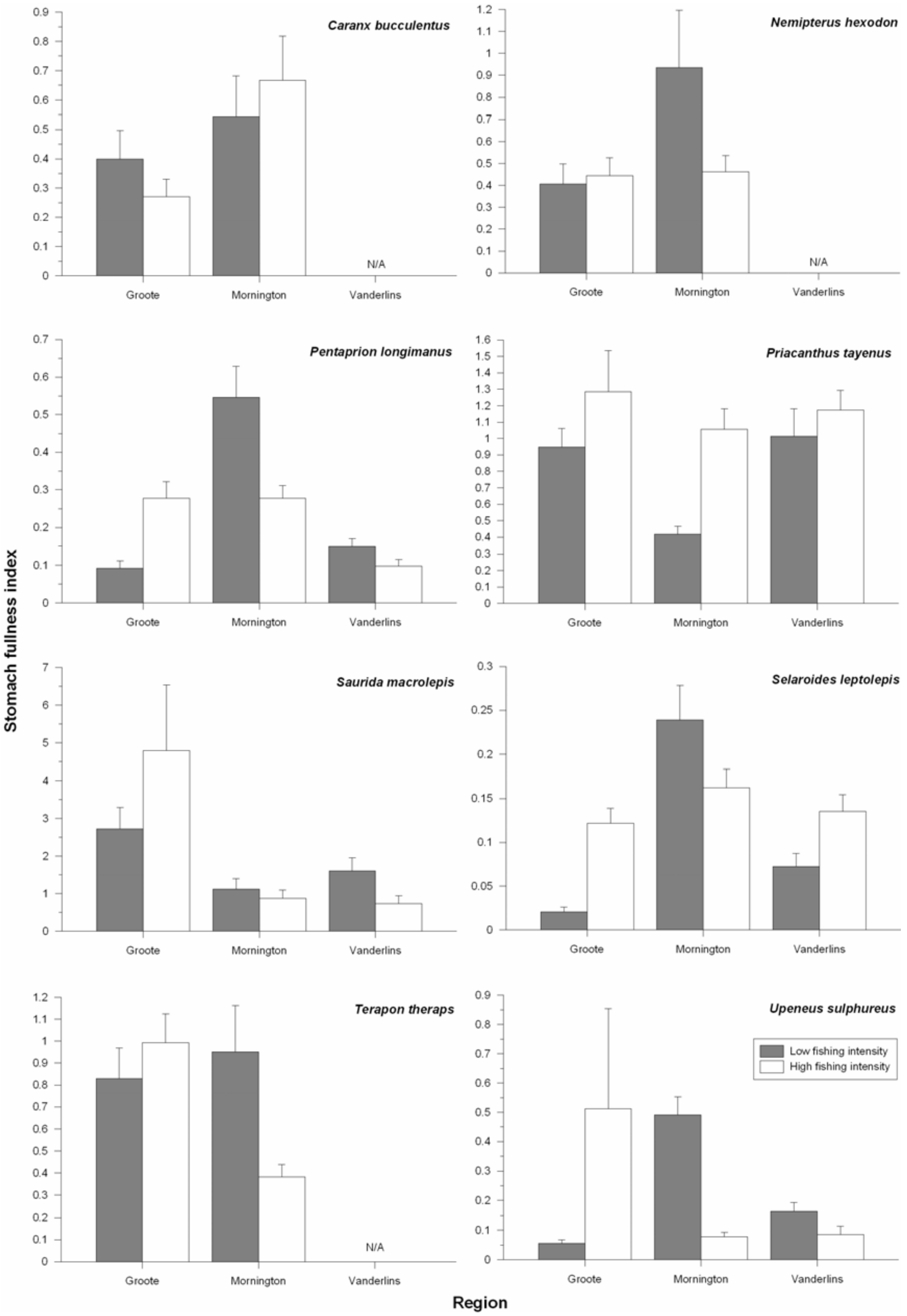

Figure 48. Mean (± s.e) stomach fullness index (SFI) of the predator fish species sampled in high and low fishing intensity areas for three regions in the Gulf of Carpentaria. N/A denotes regions where there were insufficient samples for analysis

6.5 DISCUSSION

This study aimed to determine whether local and regional ecological effects of trawling could be inferred from the trophic ecology of common demersal tropical fishes sampled in discrete regions of the Northern Prawn Fishery, in areas exposed to differing fishing intensities. For the intraspecific comparisons, the diets of seven of the eight predator species were significantly different between regions; while the diets of several predator fish species differed with trawling intensity at various regions. Benthic organisms such as gastropods, bivalves and decapods were the main prey items of these fishes and accounted for much of the dissimilarity in diet composition between fishing intensities (except for one predator with high teleost prey). With regard to the prey taxa, Long & Poiner (1994) found that the distribution patterns of benthic infauna species assemblages throughout the Gulf of Carpentaria mostly overlapped rather than being structured, discrete communities. In that study, trends in abundance and species richness of benthic organisms were correlated with gulf-wide trends in sediment textures. They also found that there were few taxa accounting for much of the infauna and there were many rare species present. Differences in fish diets between regions may occur for a host of reasons, such as physical site differences, substrate chemistry (Engel & Kvitek 1998), prey availability and predator selection (Hinz et al. 2005). The species distribution of benthic organisms recorded in Long and Poiner's (1994) study indicates a lack of uniformity across the greater Gulf of Carpentaria study area. Our results highlight that the influence of region must be considered for disentangling possible effects of trawl disturbance (as accounted for in this study). Kaiser (2003) similarly suggests that studying communities at only one scale may overlook the effects of fishing on the benthos; since detection is generally scale-dependant and at a larger spatial level, the relative measures of high and low fishing intensity may prevent detection of fishing disturbance.

An exception to the influence of region in the present study was the predator *N. hexodon*, whose diet was not significantly different between fishing intensities in the Groote or Mornington regions, but was significantly different overall between fishing intensities regardless of region. Russell (1990), reports that this fish species is broadly distributed throughout the Indo-West Pacific, is found on mud or sandy bottoms in depths of 20–50 m, and feeds on a variety of benthic animals. Consequently, significant differences in the diet of *N. hexodon* were most likely not detected between regions because of the common depth range, broad distribution and diet. While the results mainly support the supposition by Kaiser (2003), there may be advantages to generalist predators like *N. hexodon*. Because of their broader feeding strategy, these types of predator may prove useful when attempting to detect larger-scale differences in prey availability and diversity than would be possible with more specialised benthic feeders. In using predators as samplers of the benthos, the results highlight the need for careful selection of which species are appropriate for the task, considering the objectives and spatial scale involved.

Benthic predator diet has been used to infer changes in benthic communities on commercial fishing grounds as a result of fishing (e.g. Frid & Hall 1999, Link 2004, de Juan et al. 2007). These studies suggest that there were differences in the relative abundance, or availability, of the taxa eaten by predators in fished and unfished areas. In contrast, the present study had instances where the prey was found at one fishing intensity stratum, but not the other. These were prey items at the highest order of contribution to the dissimilarity for a predator at a region (e.g. *S. macrolepis* and *S. leptolepis* at Vanderlin; *T. theraps* at Mornington; and *U. sulphureus* at Groote; Figure 47. These results suggest that past fishing activity may have influenced the low fishing intensity areas—the lack of a true 'control' is a constraint discussed Engel & Kvitek (1998). However, in agreement with similar diet studies, our results also indicate that in most instances the prey items for a predator were present at both the high and low fishing intensity areas in a region, but with clear differences in average biomass. The results may reflect a low homogeneity in the benthic prey available at a particular region (in the areas that have characteristics making them suitable to trawling). Additionally, de Juan et al. (2007) note that trawling directly disturbs benthic organisms with consequences that depend on an organisms' vulnerability and a populations' resilience to fishing activities; while indirect effects on food availability also arise by modifying habitat. The results for our study may simply indicate an expected 'mix' of impact on the benthic community.

Despite the lack of a true control, significant differences in predator diet between fishing intensities were detected. Similarly to Engel & Kvitek (1998), a large fishing pressure gradient between low and high trawling intensity provided an adequate basis for comparison in this natural experiment. While some variation would be expected in benthic prey availablity in each region, there are large areas of fairly uniform sediment types throughout the Gulf of Carpentaria, such as sand or mud (Somers & Long 1994). These uniform sediment areas dictate the distribution and habitat association of benthic organisms (Long & Poiner 1994). The similarity of benthic habitat was one of the criteria used in the selection of each region for this study. Thus, our results support the method of detecting differences in fishing intensities, despite some possible variation in the distribution of benthic prey for some predators. Link (2004) reported that the use of fish stomachs as samplers of the benthos was a useful proxy, given the cost and effort required for other types of benthic community monitoring. While Link (2004) identified caveats for fish stomachs as sampling devices, he also noted their use in other studies and that their application should not be discounted. Our work further highlights the need for careful selection of the study area and predator species, as well as caution in the application of this method.

Nevertheless, the present study improves understanding of the trophic ecology of tropical demersal fish assemblages, allowing for informed decisions within the context of ecosystem based fishery management (EBFM). By providing valuable information on species with no or limited diet data, the study improves modelling processes, such as food web models (Ecopath), which is are used as tools to support EBFM. Methratta and Link (2006) investigated indicators of ecosystem health (in developing EBFM) and noted that multiple approaches are required to assess fisheries and ecosystems, recommending eight biotic indicators (including benthic predator biomass) for the northeast United States large marine ecosystem. Their study evolved from considerable knowledge gained through multidisciplinary exploration, surveys and research of that system; their indicators encompassing total ecosystem production, functional composition, and response to disturbance.

Likewise for the Northern Prawn Fishery, this work is part of a larger study to understand the spatial footprint of trawl disturbance, using Vessel Monitoring System and catch-effort at site data to investigate effort distribution and define areas of different fishing intensity (Kaiser 1998, e.g. Deng et al. 2005).This work also contributes real data to modelling spatial-explicit trophic processes to allow the evaluation of effects up-and-down the benthic food web. These are useful processes to evaluate the impact of fishing and can occur in tandem with other approaches to achieve fisheries sustainability targets (e.g. Dichmont et al. 2008).

In conclusion, the comparison of the trophic ecology of benthic predatory fish can be a useful tool for detecting differences in fishing intensity regimes when considered holistically with other approaches. These approaches can be a range of biotic indicators as per Methratta & Link (2006), who note that more that one indicator is required to suitably assess the status of fish communities and ecosystems. The present study builds on literature indicating the benefits of using demersal fish as samplers and indicators of the benthic community state and composition. Ecological studies such as this provide crucial information to derive metrics, parameters or indicators to be used in the management of tropical benthic fisheries and their ecosystems.

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Table 32. The percentage biomass (%B) and percentage frequency of occurrence (%F) of each prey taxon identified for each of the eight predator fish species, across all regions and fishing intensities in the Gulf of Carpentaria, Australia (values in bold indicate subtotals; miscellaneous prey items are excluded).

Prey Taxa	*Caranx bucculentus*		*Nemipterus hexodon*		*Pentaprion longimanus*		*Priacanthus tayenus*		*Saurida macrolepis*		*Selaroides leptolepis*		*Terapon theraps*		*Upeneus s*
	%B	%F	%B	%F	%B	%F	%B	%F	%B	%F	%B	%F	%B	%F	%B
Subphylum Crustacea Total	***59.0***		***40.8***		***8.8***		***38.8***		***6.2***		***19.8***		***29.5***		***86.3***
Unid. Crustacea	0.7	6.5	0.7	5.2	1.8	8.7	1.6	10.2	<0.1	0.4	5.3	10.1	0.8	6.7	1.5
Unid. Malacostraca			<0.1	1.0	0.1	0.6					0.4	0.8			<0.1
Order Decapoda Total	**51.3**		**30.2**		**4.8**		**30.9**		**5.8**		**9.3**		**20.5**		**80.2**
Unid. Decapod	1.4	4.7	1.5	4.2	0.4	2.7	0.3	2.7	0.2	0.4	2.3	2.4	0.4	1.8	0.4
Unid. Dendrobranchiata			<0.1	0.5	<0.1	0.3									0.2
Unid. Pleocyemata					<0.1	0.6							0.2	0.4	
Infraorder Brachyura Total	**36.7**		**21.2**		**3.4**		**2.7**				**3.9**		**15.9**		**6.3**
Brachyura megalopa							0.1	3.8			0.1	1.2			0.1
Calappidae	1.2	1.8	0.1	0.5											
Carpiliidae	1.5	1.2	1.0	1.6											
Dromiidae							0.1	0.3							
Goneplacidae	1.0	0.6	2.8	3.7									0.3	0.4	
Leucosiidae	1.5	5.9			0.2	0.6					2.8	1.2	0.3	0.9	0.1
Majidae	0.2	0.6	1.2	1.0											
Parthenopidae	0.8	0.6													
Pilumnidae	0.5	0.6	2.3	1.6									0.7	0.9	0.1
Porcellanidae							<0.1	0.7					0.1	0.9	
Portunidae	13.1	5.9	5.0	3.7	0.1	0.6	0.1	0.7					0.1	0.9	0.3
Xanthidae			0.1	0.5									0.1	0.4	
Unid. Brachyura	16.8	30.6	8.7	24.1	3.1	13.8	2.5	8.9			1.1	6.1	14.2	21.3	5.7
Infraorder Caridea Total	**0.2**		**2.3**				**21.4**		**0.6**		**<0.1**		**1.9**		**58.8**
Alpheidae			0.8	0.5			0.1	0.7					1.4	1.8	
Crangonidae							<0.1	0.3							
Unid. Caridea	0.2	1.8	1.5	4.7			21.2	28.3	0.6	2.7	<0.1	0.4	0.4	5.3	58.8
Infraorder Anomura Total	**0.4**		**1.2**		**<0.1**		**0.3**						**<0.1**		**0.2**
Superfamily paguroidea	0.4	1.8	1.2	1.6	<0.1	0.6	0.3	0.7					<0.1	0.4	0.2
Infraorder Astacidea Total	**0.2**		**<0.1**				**0.1**								**0.1**
Scyllaridae	0.2	1.2	<0.1	0.5			0.1	0.3							0.1
Suborder Natantia Total	**1.4**	**8.8**	**3.7**	**17.8**	**1.2**	**6.9**	**4.7**	**29.4**	**1.1**	**3.9**	**5.3**	**9.7**	**1.4**	**12.9**	**8.9**
Superfamily Penaeoidea Total	**12.5**		**1.7**		**0.2**		**1.7**		**4.1**		**0.1**		**1.4**		**5.9**
Penaeidae	11.3	11.8	1.7	2.6			0.7	3.4	4.0	5.4			1.0	3.1	4.4
Solenoceridae	0.3	0.6					0.1	0.3							
Unid. Penaeoidea	0.8	1.2			0.2	0.6	0.9	1.7	0.1	0.4	0.1	0.4	0.4	1.8	1.5
Superfamily Sergestoidea Total							**<0.1**								**<0.1**
Luciferidae															<0.1
Sergestidae							<0.1	0.7							
Class Ostracoda Total	**<0.1**	**1.2**	**<0.1**	**0.5**	**<0.1**	**1.2**	**0.2**	**18.1**			**2.1**	**30.0**	**0.2**	**10.7**	**0.1**
Infraorder Thalassinidea Total			**<0.1**				**0.2**						**0.3**		
Callianassidae							0.2	0.3							
Thalassinidae			<0.1	0.5			<0.1	1.0							
Upogebiidae							<0.1	0.7					0.3	0.4	
Order Copepoda Total	**<0.1**	**0.6**			**<0.1**	**0.3**	**<0.1**	**0.7**			**<0.1**	**2.0**			**<0.1**
Order Stomatopoda Total	**5.1**		**7.8**		**1.0**		**2.4**		**<0.1**		**<0.1**		**6.3**		**3.4**
Squillidae	2.8	2.9	1.6	1.6	0.2	0.3	<0.1	0.3					2.3	3.1	<0.1
Stomatopod larvae			<0.1	0.5			0.8	3.4					0.2	0.9	
Unid. Stomatopoda	2.3	5.3	6.1	14.1	0.7	1.5	1.5	8.2	<0.1	0.4	<0.1	0.4	3.8	8.0	3.3
Superorder Peracarida Total	**0.4**		**0.6**		**0.8**		**3.1**		**0.1**		**0.3**		**0.8**		**0.5**
Amphipoda	<0.1	0.6	0.4	2.6	0.1	3.6	0.2	7.2	<0.1	0.4	0.1	3.2	<0.1	2.7	0.3
Cumacea			<0.1	0.5	0.7	2.4	0.7	13.3			<0.1	1.6			0.1
Isopoda	0.4	1.8	0.2	2.1			2.0	15.7	0.0	1.5	0.2	0.4	0.7	4.4	<0.1
Mysidacea							0.1	1.4							
Tanaidacea					<0.1	0.3	0.1	1.7					<0.1	1.3	<0.1
Phylum Annelida Total			***3.9***		***5.2***		***3.9***				***1.7***		***7.1***		***0.6***
Amphinomidae Total													**2.2**	**2.2**	
Class Polychaeta Total			**3.9**		**5.2**		**3.9**				**1.7**		**4.9**		**0.6**
Opheliidae											<0.1	0.8			<0.1
Unid. Polychaeta			3.9	10.5	5.2	16.2	3.9	27.0			1.6	5.3	4.9	13.3	0.6
Phylum Chordata Total													***0.2***		
Class Ascidiacea Total													0.2	0.9	
Phylum Cnidaria Total			***<0.1***		***<0.1***		***0.2***				***<0.1***		***7.6***		
Class Hydrozoa											<0.1	0.4			
Class Scyphozoa							0.1	0.3					7.6	2.7	
Order Alcyonacea			<0.1												
Subclass Zoantharia					<0.1	0.3	0.1	0.3							

Phylum Echinodermata Total	***3.0***		***3.7***		***2.7***		***<0.1***				***18.7***		***3.0***		***0.3***
Class Asteroidea Total													**<0.1**	**0.4**	
Class Echinoidea Total	**0.4**		**0.5**		**0.1**		**<0.1**				**3.2**		**2.2**		
Clypeasteridae	<0.1	0.6					<0.1	0.3							
Diadematidae													0.4	0.4	
Unid. Echinoidea	0.3	2.9	0.5	2.1	0.1	0.9					3.2	5.7	1.8	6.2	
Class Holothuroidea Total			**3.2**	**1.6**									**0.7**	**0.9**	
Class Ophiuroidea Total	**2.6**	**12.4**			**2.6**	**9.0**					**15.5**	**11.3**	**0.1**	**0.9**	**0.3**
Phylum Mollusca Total	***0.3***		***4.7***		***73.7***		***2.2***		***2.5***		***29.4***		***0.6***		***10.5***
Unid. Mollusca	<0.1	0.6			0.3	1.8					9.1	8.1	0.1	1.3	
Class Bivalvia Total	**0.3**		**<0.1**		**72.7**		**<0.1**		**<0.1**		**17.9**		**0.3**		**10.5**
Carditidae											<0.1	0.8			
Glycymerididae															
Laternulidae					<0.1	0.3									
Nuculanidae					0.5	3.0					<0.1	0.8			8.8
Tellinidae					69.8	15.3					0.2	1.6			1.4
Veneridae					<0.1	0.3					0.1	0.8			
Unid. Bivalvia	0.3	7.1	0.1	3.1	2.3	11.7	<0.1	2.7	<0.1	0.4	17.6	30.4	0.3	2.2	0.3
Class Cephalopoda Total			**4.6**		**0.1**		**0.3**		**2.4**				**0.2**		
Loliginidae							0.2	0.3							
Sepiolidae					0.1	0.3	<0.1	0.3							
Unid. Cephalopoda			<0.1	0.5			0.1	1.7					0.2	0.9	
Unid. Octopoda			4.6	1.6			<0.1	0.3	2.4	0.8					
Class Gastropoda Total	**<0.1**		**<0.1**		**0.7**		**1.8**		**<0.1**		**2.5**		**0.1**		**<0.1**
Architectonidae													<0.1	0.4	
Atlantidae					<0.1	0.3					<0.1	2.4			
Buccinidae											<0.1	0.4			
Cancellariidae															
Cavoliniidae					0.1	5.7	1.5	3.4			<0.1	0.4			<0.1
Costellariidae					<0.1	0.3									
Dentaliidae															
Marginellidae					0.1	0.6					<0.1	2.0			
Naticidae					0.1	1.5					<0.1	1.6			
Pyramidellidae															
Trochaclididae															
Turritellidae					<0.1	0.6									
Volutidae			<0.1	0.5							<0.1	0.4			
Xenophoridae															
Unid. Gastropoda	<0.1	0.6			0.4	3.6	0.3	0.7	<0.1	0.4	2.4	16.2	<0.1	2.2	<0.1
Phylum Protozoa Total					***<0.1***		***<0.1***				***0.2***				
Order Foraminiferida Total					**<0.1**	**0.3**	**<0.1**	**0.7**			**0.2**	**2.0**			
Phylum Sipuncula Total			***<0.1***	***0.5***	***0.1***	***0.6***							***0.1***	***0.4***	***0.1***
Teleost Total	***37.7***		***41.7***		***1.6***		***53.2***		***90.4***		***2.4***		***43.3***		***0.7***
Anguilliformes			<0.1	0.5	0.4	0.3							0.9	0.4	
Apogonidae	11.8	0.6					0.8	0.3	8.3	0.8					
Bregmacerotidae			2.8	0.5			0.3	0.7	2.3	3.9					
Carangidae									1.1	0.4					
Centriscidae							0.1	0.3							
Clupeidae									2.5	0.4					
Cynoglossidae	7.0	0.6													
Engraulidae	1.9	0.6	8.7	1.6			1.2	1.0	20.7	13.1					
Gerreidae									4.0	0.4					
Gobiidae					0.2	0.3							2.3	0.9	
Leiognathidae									4.4	1.2					
Monacanthidae			0.1	0.5			0.4	1.4							
Mullidae									0.1	0.4					
Muraenesocidae									0.4	0.4					
Muraenidae	1.7	0.6													
Platycephalidae		0.6							<0.1						
Plotosidae			0.4							0.4					
Sphyraenidae				0.5					0.3						
Synodontidae									1.3	0.4			0.7		
Triacanthidae							<0.1			1.5				0.4	
Order Pleuronectiformes	0.8					0.3				2.7		0.4			
Teleost larvae		0.6		1.6	<0.1	0.6			<0.1	0.8	<0.1	2.4		5.3	<0.1
Teleost scale	<0.1		0.2		<0.1			1.0	<0.1		0.1		2.7		<0.1
Unid. Teleost	14.5	15.3	29.4	26.2	1.0	4.5	50.4	61.4	44.9	61.0	2.4	3.6	36.7	40.0	0.6
Detritus Total	***0.1***	***0.6***	***1.5***	***4.7***	***4.8***	***14.4***	***0.4***	***5.5***	***0.6***	***2.3***	***15.0***	***15.4***	***2.9***	***8.0***	***1.1***

APPENDIX 7 INFLUENCE OF PREDATOR MORPHOLOGY ON FOOD RESOURCE PARTITIONING IN SYMPATRIC DEMERSAL FISHES

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7.1 SUMMARY

Differences in morphology and food selection can explain how sympatric fishes are able to partition food resources and therefore coexist successfully. The mouth gape of six abundant and widely distributed sympatric demersal species found in northern Australia was examined: *Elates ransonetti Suggrundus macracanthus, Kumococius rodericensis, Inegocia japonica* (Platycephalidae)*, Upeneus sulphureus* (Mullidae)*, Saurida macrolepis* (Synodontidae). Significant interspecific variation was detected in mouth gape relative to fish size and rate at which gape increased through ontogeny. Dietary overlap was significantly higher for the morphologically similar species compared to morphologically dissimilar species. Furthermore, morphological differences and food selection were used to explain significant interspecific and intraspecific variation in diet. We suggest that different feeding strategies are utilized by these morphologically similar and contrasting species to partition food resources by prey type and size. New species-specific dietary information is provided, particularly for two flathead species which have been described for the first time globally. New trophic information, including penaeid prawn predation, is useful in fine-tuning ecosystem models used to support ecosystem-based fisheries management in fisheries such as Australia's Northern Prawn Fishery.

Key words: Gape, Ontogenetic, Platycephalidae; Mullidae; Synodontidae; diet; predator-prey

7.2 INTRODUCTION

Fish species that are closely related need to partition available resources in order to reduce competition and successfully exist together (Ross 1986; Blaber and Bulman 1987; Fujita et al. 1995; Salini et al. 1998). Spatial and temporal partitioning are the most obvious means by which competition can be reduced, especially by utilising different habitats. However, in closely-related species with similar morphology and habitat requirements, partitioning of available prey becomes an important means by which fish species can reduce competition (Ross 1986; Labropoulou and Eleftheriou 1997; Platell et al. 1998). Variations in prey selectivity are often size–related allowing for partitioning of food resources at different life cycle stages (Werner and Gilliam 1984; Platell et al. 1998). Over time, sympatric species sometimes evolve different morphological features that result in different prey selectivity and feeding strategies that assist in food resource partitioning (Motta 1988; Wainright and Richard 1995; Scharf et al. 2000).

For example, sympatric pelagic fishes such as yellowfin tuna and bigeye tuna appear to reduce competition for resources by occupying different depth strata where they target different prey (Poter et al. 2004). Yellowfin tuna often feed in superficial waters targeting small crustaceans and fishes. In contrast, bigeye tuna feed at depths below the thermo-cline, targeting mesopelagic cephalopods and fishes. Bigeye tuna have evolved specialised adaptations for this type of feeding including retia that heat the brain and eyes. These allow them to spend more time foraging in deeper cooler waters than yellowfin tuna, which have a lower thermal inertia (Brill et al. 2005). In contrast, benthic fishes are generally restricted to horizontal stratification of habitats. For example, two morphologically similar sympatric goatfish were found to feed slightly differently and/or in different benthic microhabitats which helped to facilitate their co-existence (Platell et al. 1998). Others studies have shown that some demersal fish have reduced interspecific and ontogenetic competition mostly by varying their prey size selectivity (Scharf et al. 2000).

Understanding the trophic relationships among sympatric demersal fishes provides important information for ecosystem models such as Ecopath (Okey 2006; Pascoe et al. 2008) and Atlantis (Smith et al. 2007). These models are being used to support ecosystem-based fisheries management (EBFM) of the Northern Prawn Fishery (NPF), one of Australia's largest and most valuable fisheries. The bycatch of this fishery is large, consisting of numerous small demersal and benthic fishes (Stobutzki et al. 2001, Tonks 2008). There is a limited understanding of their trophic ecology, particularly in relation to their consumption of penaeid prawns. In contrast, there is a better understanding of the trophic ecology of some of the common larger predatory fish species found on trawl grounds (Salini et al. 1994) and in estuaries (Brewer et al. 1991; Brewer 1995; Haywood 1998; Salini et al. 1998) in northern Australia such as elasmobranches, emperors, sweetlips, trevallies, cods, catfish, barracudas, mackerels and queenfish. The contribution of prawns to the diets of some species examined in these studies was as high as 46% (dry weight). Fishes of the Platycephalidae (flathead), Mullidae (goatfish) and Synodontidae (saury) families are some of the most common and abundant small fishes caught as bycatch in the NPF, with 12 platycephalid, 9 mullid and 7 synodontid species recorded (Stobutzki et al. 2001). Many of these species are also abundant and widely distributed outside of traditional trawl grounds (CSIRO unpublished data); as such their trophic impact is likely to be significant. Although not the primary aim of this study, gathering dietary information for some of these species is relevant because it will not only provide the first description of diets for several species, it will also improve the resolution of ecosystem models that require trophic information, and will extend our understanding of the role of predation on prawns in the NPF.

For this study, abundant and broadly distributed demersal species were chosen that had similar and contrasting mouth structures, gape and body morphology. We aimed to i) examine interspecific and ontogenetic variation in mouth gape ii) assess the interspecific, ontogenetic and geographic variation in their diet iii) examine the role of ontogenetic variation in diet in reducing intraspecific competition in three of these species iv) provide new species-specific diet information that will enhance ecosystem models, and extend our understanding of prawn predation in the Northern Prawn Fishery.

7.3 MATERIALS AND METHODS

7.3.1 STUDY AREA

The study area was located on tiger prawn (*Penaeus semisulcatus* and *P. esculentus*) fishing grounds, determined from Vessel Monitoring System (VMS) data, in the southern and western regions of the Gulf of Carpentaria (Figure 49). The fish samples were collected in depths ranging from 35 to 51 m.

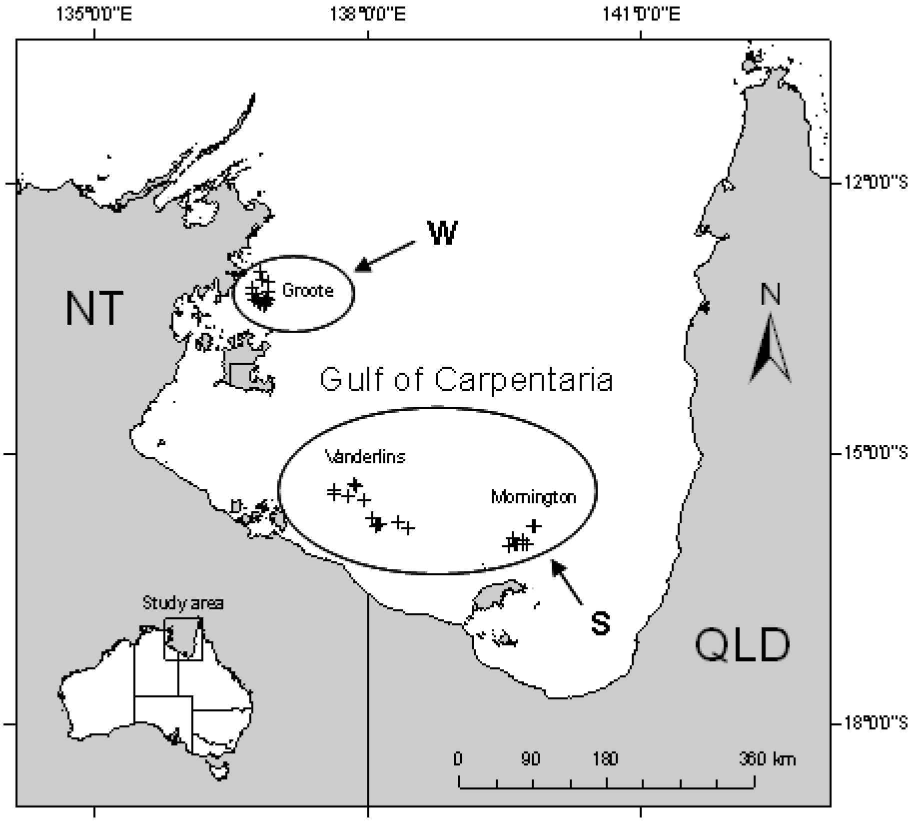

Figure 49. Map of the Gulf of Carpentaria, Northern Australia. '+' symbol represent locations where demersal species were collected. '**W**' represents western gulf, '**S**' represents southern gulf.

7.3.2 SAMPLE COLLECTIONS

Prawn trawls were made with the *FRV Southern Surveyor* towing a single standard industry (NPF) banana prawn net (12 fathom, 2 inch stretch mesh, Bison # 9 boards) to collect the following fish species: *Elates ransonetti Suggrundus macracanthus, Kumococius rodericensis, Inegocia japonica* (Platycephalidae)*, Upenueus sulphureus* (Mullidae)*, Saurida macrolepis* (Synodontidae). A total of 39 trawls of ten minute duration (bottom time) were used to collect 1828 fish, of the selected fish species, from the three regions between the 28th February and the 19th March 2005 (Table 33). Fish were frozen (-20°C) and returned to the laboratory for later analyses.

Table 33. Number of fish dissected and number of fish with stomach contents (in parenthesis) for each region sampled.

Species (common name)	Region			Total	Length range (mm)
	Western gulf	Southern gulf			
	Groote	Mornington	Vanderlins	*n*	
Elates ransonetti (Dwarf Flathead)	386 (122)	33 (26)	2 (1)	421 (149)	87–172, av =143
Suggrundus macracanthus (Bigspine Flathead)	99 (78)	70 (48)	9 (7)	178 (133)	65–190, av = 128
Kumococius rodericensis (Whitefin Flathead)	54 (41)	1 (0)	2 (2)	57 (43)	70–180, av =128
Inegocia japonica (Rusty Flathead)	9 (6)	34 (22)	3 (3)	46 (31)	87–225, av =161
Saurida macrolepis (Lizardfish)	158 (107)	168 (94)	90 (57)	416 (258)	74–218, av =145
Upeneus sulphureus (Sunrise goatfish)	338 (125)	121 (105)	251 (87)	710 (317)	64–134, av =99

7.3.3 LABORATORY METHODS

Each fish was thawed and its standard length (± 1 mm) and weight (± 0.1 g) measured. Fish showing signs of barotrauma were excluded from the analysis. For all other fish, stomachs were removed and their contents placed into a petri dish, sorted and identified under magnification. Prey were categorised to the lowest taxonomic group possible. The prey items defined as 'natantia' were those that were prawn or shrimp–like (Grey et al. 1983). Each individual prey item (or taxonomic prey group) was counted, weighed (wet weight, nearest 0.001 g) and measured. Total length was measured for teleosts, prawns, pericarida, stomatopods and polychaetes wherever possible. Carapace width was measured for crabs. The combined wet weight of prey items for a stomach provided an estimate of the mass of the stomach content. The wet weight of individual prey taxa was then converted to a percentage of total prey weight for each fish.

The mouth width (maximum width of gape with mouth fully open) and mouth height (maximum height of gape with mouth fully open) of each fish with prey contents were each measured ± 0.1 mm with callipers. When the mouths were fully extended each species showed a similar elliptical gape. Consequently mouth gape circumference G was then calculated using the equation derived from Hodgman (1954):

$$G = 2 * \pi * \text{sqrt}\ ((a^2+b^2)/2) \qquad (1)$$

Where a is the radius of the longest diameter (mouth height) and b is the radius of the shortest diameter (mouth width)

7.3.4 DATA ANALYSIS

Interspecific variability in mouth gape through ontogeny was examined by linear regression. T-tests were then used to determine if the slopes and y intercepts of the regressions varied among each species. Prey sizes to predator size scatter plots were generated and a linear regression was fitted to estimate the relationship between mean prey size and predator size. Patterns of relative prey size among predators were examined by generating frequency histograms of prey size to predator size ratios.

To investigate the extent of dietary overlap between species, diet biomass was used to calculate Horn's overlap index (*Ro*) (Horn, 1966):

$$R_O = \frac{\sum (p_{ij} + p_{ik}) \log (p_{ij} + p_{ik}) - \sum p_{ij} \log p_{ij} - \sum p_{ik} \log p_{ik}}{2 \log 2} \qquad (2)$$

Where p_{ij} = the proportion of prey type *i* of the total number of prey consumed by species *j,* and p_{ik}= the proportion of prey type *i* of the total number of prey consumed by species *k*. Horn's index ranges from 0 (no overlap) to 1 (complete overlap). Dietary overlap values ≥0.6 are considered "biologically significant" (Pianka 1976).

We used a multivariate approach to examine similarities in diet (% wet weight) among all six fish species; among four fish species between two regions (western/southern gulf); and between two fish size groups (immature and mature) for three species. Size at maturity of each species was determined from Fishbase (Froese and Pauly 2009) and defined as ≥ 160 mm SL for *S. macrolepis*, ≥ 145 mm SL for *S. macracanthus* and ≥ 100 mm SL for *U. sulphureus*. Data was square-root transformed to reduce the influence of highly weighted prey taxa, and a similarity matrix was constructed using Bray-Curtis similarity coefficients. Prey composition was compared among species by multidimensional scaling (MDS). Analysis of similarities (ANOSIM) was used to test whether prey composition in a priori groups differed statistically (Clarke 1993). Similarity percentages (SIMPER) were then used to determine which prey was responsible for differences between groups defined by ANOSIM as being statistically different. All multivariate analyses were carried out with PRIMER V6.

7.4 RESULTS

7.4.1 INTERSPECIFIC VARIATION IN MOUTH GAPE

A linear relationship exists between fish size and mouth gape for all species examined in this study (Figure 50). The regression slopes were similar for most species, however the gape of *S. macrolepis* increases with fish growth more so than the other species, and the gape of *E. ransonetti* increases least (Figure 50). The slope for *S. macrolepis* was significantly different from all other species (P=<0.01), except for *U. sulphureus*. The slope for *E. ransonetti* was also significantly different from *K. rodericensis* (P=<0.01).

The gape between the six species showed some differences with *S. macrolepis* having the largest gape relative to size, which was significantly different from all other species (P=<0.05). Three platycephalids; *K. rodericensis*, *S. macracanthus* and *I. japonica* did not have significantly different gapes (P=>0.05), while the remaining playcephalid *E. ransonetti* had the smallest gape of all species and was significantly different from all other species (P=<0.05). The goatfish *U. sulphureus* also had a relatively small gape that was intermediate between *E. ransonetti* and the three other flathead species (Figure 50), which again was significantly different from all other species (P=<0.05).

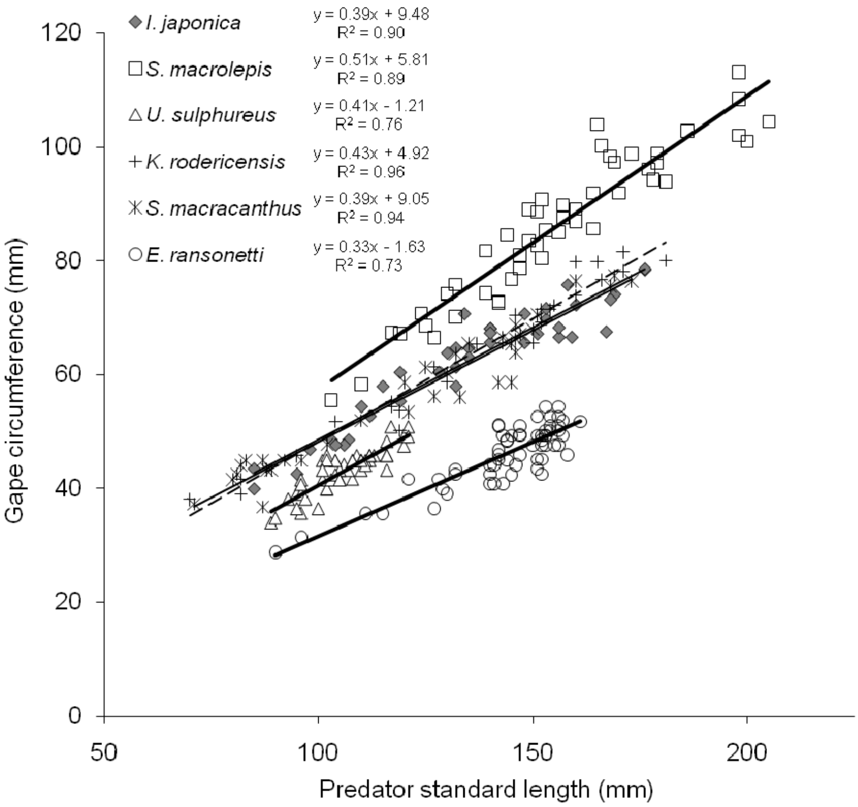

Figure 50. Regressions illustrating the relationship between fish size and gape for each predator species. The correlation coefficients are also shown for each linear regression.

7.4.2 ONTOGENETIC VARIATION IN PREY SIZE

Prey sizes were plotted against mouth gape for the six predator species (Figure 51). Only the lizardfish, *S. macrolepis* showed a clear positive relationship between mean prey size and increasing mouth gape. The four flathead species, *E. ransonetti*, *S. macracanthus*, *K. rodericensis* and *I. japonica* showed a small increase in mean prey size with increasing gape while the goatfish, *U. sulphureus* did not show an increase in mean prey size with increasing gape.

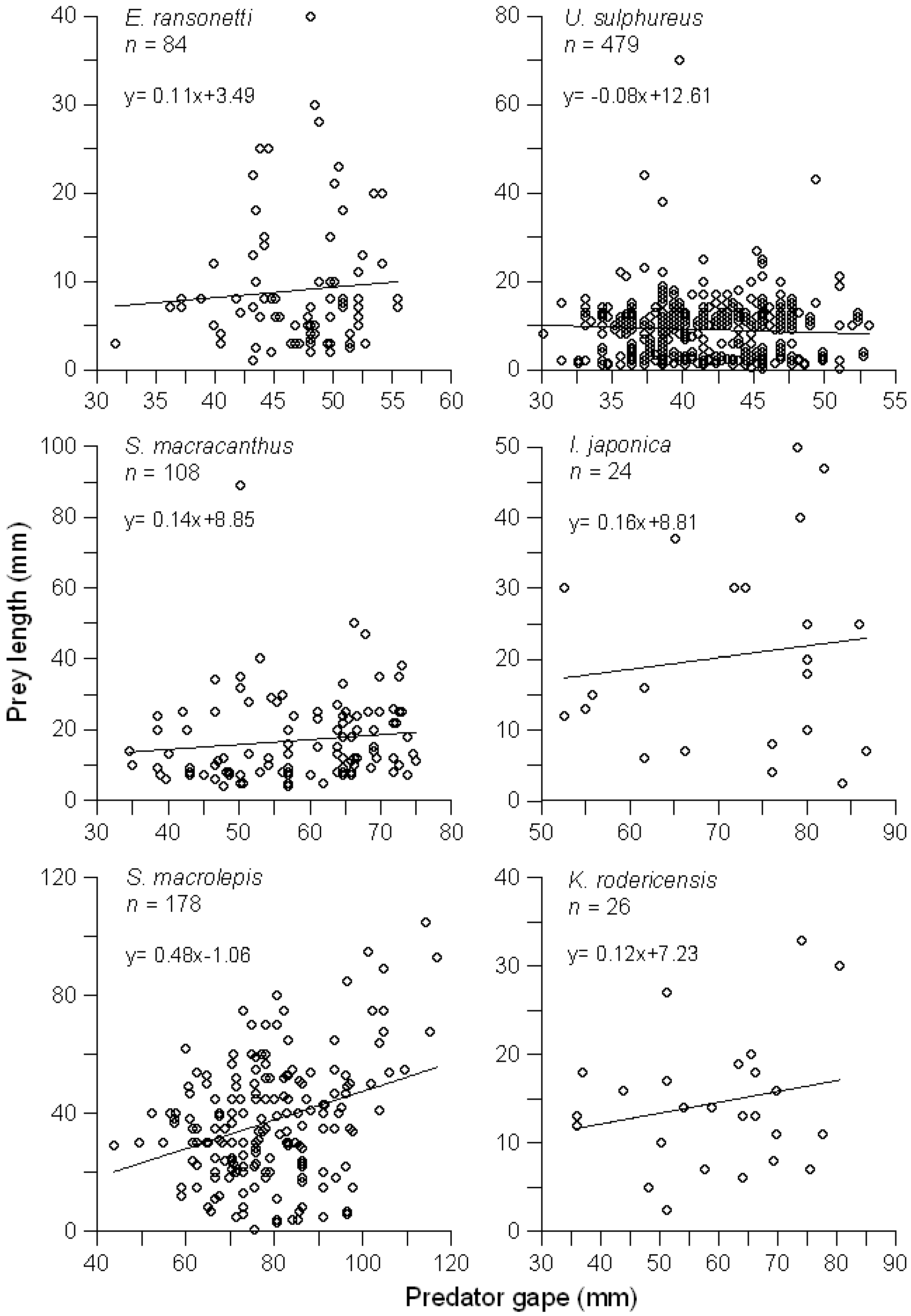

Figure 51. The relationship between mouth gape (circumference) and prey size for each predator species. Prey measurements–fish, prawns, stomatopods, pericarida, polychaetes (TL); crabs (CW).

7.4.3 PREY SIZE / PREDATOR SIZE RATIOS

The relative importance of prey size varied among the fish species (Figure 52). A majority of the prey consumed by five fish species were of a small size relative to predator body size (>75% of prey items <21% of body size): *K. rodericensis, E. ransonetti*, *S. macracanthus, I. japonica* and *U. sulphureus*. Both *E.ransonetti* and *U. sulphureus* had a very high proportions (>97%) of prey <21% of body size. *Elates ransonetti* in particular ate mostly small prey (<11% of body size) with the maximum prey size recorded at 26% of body size. The other three flathead species, *K. rodericensis, S. macracanthus and I. japonica* also had high percentages of small prey in their diet (77–88% of prey items <21% of body size). Maximum prey sizes in relation to fish size for these species were: *K. rodericensis* (25%)*, S. macracanthus* (85%) *and I. japonica* (28%). In contrast to these species, the lizardfish, *S. macrolepis*, had a much smaller proportion of their diet consisting of small prey (<21% of its body size). Instead, about 40% of prey was more than 30% of its body size. The largest prey item recorded for this species was 59% of body size.

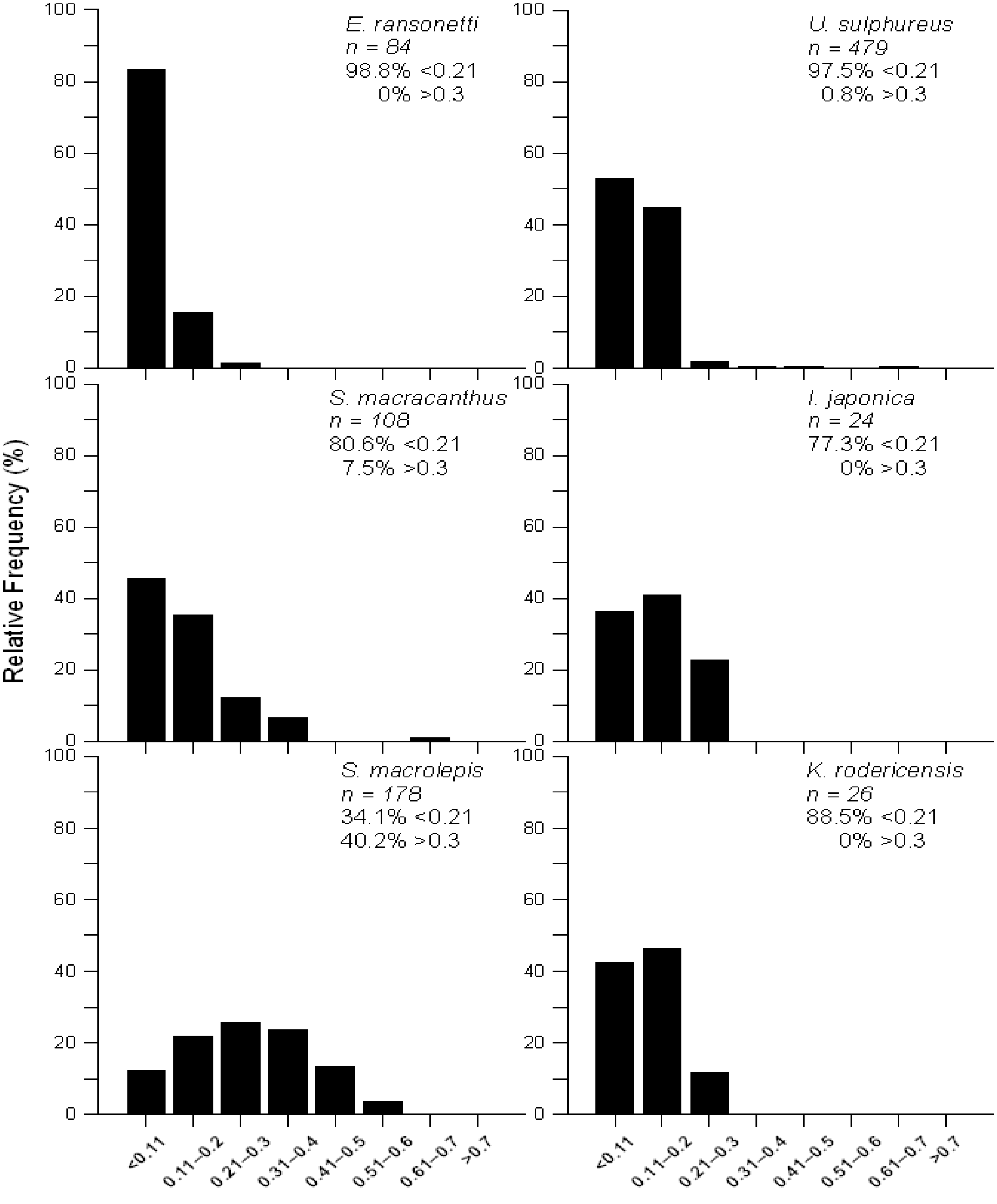

Figure 52. Relative frequency distributions of prey size/predator size ratios for the six fish species.

7.4.4 OVERALL DIET COMPOSITION

Between 31 and 317 stomachs of each of the six teleost species contained food, which represented 35% of all stomachs examined. Six broad prey groups were identified in the diets: crustacea, teleost, mollusca, echinoderm, polycheata and sipuncula (Table 34). The goatfish, *U. sulphureus*, had the broadest diet, feeding on all six prey taxa groups. The most specialised was the flathead, *I. japonica*, with prey from only two of the groups (Table 34). *Upenaeu. sulphureus* was the only species found to consume echinoderms (ophiuroid) and sipuncula.

Five of the six species: *E. ransonetti*, *S. macracanthus, K. rodericensis, I. japonica* and *U. sulphureus* ate mostly crustaceans. The contribution of crustaceans varied between species and ranged from 63–95% by wet weight. Of these species, *E. ransonetti* had a much greater teleost component (35%) compared to the others (<12%). The remaining species, *S. macrolepis* differed in that teleost was the dominant prey group (90%) with crustacean contributing only 6% by wet weight.

Differences within the crustacean component of the diets were also observed (Table 34). *Kumococius rodericensis* had a higher contribution of stomatopods and brachyura (>20% ww for both prey) compared with *E. ransonetti*, *S. macracanthus, I. japonica* and *U. sulphureus* (<12% for both). *Elate. ransonetti* had a higher contribution of pericarida (2.5% ww) compared with *S. macracanthus, K. roderisensis, I. japonica* and *Upeneus sulphureus* (≤0.4%). *U. sulphureus* was the only species that consumed ostracods which were found in 10% of stomachs (with food contents). Carids contributed 58% ww to the diets of *S. macracanthus* and *U. sulphureus,* which was considerably higher than the other species. In contrast, penaeoidea made up nearly 62% ww of the diet of *I. japonica* of which almost 52% consisted of Penaiedae. This penaied component of the diet was considerably higher than the other species.

Table 34. Diet composition of the six predator species, by percentage wet weight (%ww) and percentage of fish (with stomach contents) the prey taxa occurred in (%O).

	E. ransonetti		***S. macracanthus***		***K. roderisensis***		***I. japonica***		***S. macrolepis***		***U. sulphureus***	
	n = 149		*n* = 133		*n* = 43		*n* = 31		*n* = 258		*n* = 317	
	%ww	%O	%ww	%O	%ww	%O	%ww	%O	%ww	%O	%ww	%O
Subphylum Crustacea	**63.1**		**87.4**		**85.3**		**95.8**		**6.2**		**86.3**	
Crustacea - unid	2.4	10.1	0.1	3.8	-	-	0.6	3.2	0.1	0.4	1.5	10.1
Order Copepoda	<0.1	0.7	-	-	-	-	-	-	-	-	<0.1	1.9
Class Ostracoda	-	-	-	-	-	-	-	-	-	-	0.1	10.1
Order Stomatopoda- total	**7.8**	**-**	**12**	**-**	**20.7**	**-**	**7.2**	-	**<0.1**	-	**3.4**	-
Stomatopoda - unid	7.8	4	4.3	9.8	10.6	7	<0.1	3.2	<0.1	0.4	3.3	4.4
Squillidae	-	-	7.7	11.3	10	2.3	7.2	9.7	-	-	<0.1	0.6
Order Decapoda- total	**46.5**	**-**	**75**	**-**	**64.3**	**-**	**87.9**	-	**5.8**	-	**80.2**	-
Decapoda - unid	1.5	4.7	0.4	3	0.3	2.3	-	-	0.2	0.4	0.4	2.2
Order Pericarida- total	**2.5**	**-**	**<0.1**	**-**	**<0.1**	**-**	**0.1**	**-**	**0.1**	**-**	**0.5**	**-**
Mysidacea	<0.1	0.7	-	-	-	-	-	-	-	-	-	-
Cumacea	-	-	-	-	-	-	<0.1	3.2	-	-	0.1	4.4
Tanaidacea	1.4	3.4	-	-	-	-	-	-	-	-	<0.1	2.8
Isopoda	0.9	4	-	-	-	-	-	-	<0.1	1.6	<0.1	-
Amphipoda	0.2	4	-	-	<0.1	2.3	0.1	3.2	<0.1	0.4	0.3	11.4
Infraorder Brachyura- total	**8.3**	**-**	**7**	**-**	**24.1**	**-**	**9.2**	**-**	**-**	**-**	**6.3**	**-**
Brachyura - unid	3.2	8.7	1.1	6.8	7.3	7	1.4	6.5	-	-	5.7	14.8
Brachyura megalopa	-	-	-	-	-	-	<0.1	3.2	-	-	0.1	1.3
Pilumnidae	-	-	-	-	-	-	-	-	-	-	<0.1	0.3
Portunidae	3	3.4	4.9	8.3	16.8	7	7.8	6.5	-	-	0.3	1.6
Xanthidae	0.6	0.7	0.1	1.5	-	-	-	-	-	-	-	-
Leucosiidae	1.5	2	-	-	-	-	-	-	-	-	<0.1	0.3
Goneplacidae	-	--	0.9	0.8	-	-	-	-	-	-	-	-
Infraorder Astacidea- total	-	-	**0.5**	-	-	-	-	-	-	-	**<0.1**	-

Scyllaridae	-	-	0.5	1.5	-	-	-	-	-	-	<0.1	0.3
Infraorder Thalassinidea- total	**2.3**	-	**<0.1**	-	-	-	-	-	-	-	-	
Upogebiidae	2.3	1.3	<0.1	0.8	-	-	-	-	-	-	-	-
Infraorder Caridea - total	**5.9**	**12.8**	**58.3**	**-**	**22.3**	**-**	**7.6**	-	**0.6**	-	**58.8**	-
Caridea - unid	4.9	12.1	57.6	17.3	21.2	25.6	7.6	9.7	0.6	2.7	58.8	41.6
Alpheidae	1	0.7	-	-	1.1	2.3	-	-	-	-	-	-
Crangonidae	-	-	0.7	1.5	-	-	-	-	-	-	-	-
Superfamily Sergestoidea- total	-	-	-	-	-	-	-	-	-	-	**<0.1**	-
Luciferidae	-	-	-	-	-	-	-	-	-	-	<0.1	0.3
Infraorder Anomura - total	-	-	-	-	-	-	-	-	-	-	**0.2**	**0.6**
Superfamily Paguroidea	-	-	-	-	-	-	-	-	-	-	0.2	0.6
Superfamily Penaeoidea - total	**3.8**	**-**	**7.4**	**-**	**7.1**	**-**	**61.7**	-	**4.1**	-	**5.9**	-
Penaeoidea - unid	0.6	3.4	1.6	6.8	6.4	9.3	7	6.5	0.1	0.4	1.5	4.1
Penaeidae	3.2	3.4	5.8	13.5	0.7	2.3	51.5	19.4	4	5.4	4.4	1.6
Solenoceridae	-		-	-	-	-	3.2	3.2	-	-	-	-
Suborder Natantia	**28.5**	**46.3**	**1.8**	**21.1**	**10.8**	**-**	**9.5**	32.3	**1.1**	3.9	**8.9**	35
Teleost - total	**35.5**		**11.9**		**10.9**		**3.4**		**90.4**		**0.7**	
Teleost - unid	26.8	11.4	4	14.3	10.8	14	3.4	6.5	44.9	65.5	0.6	2.8
Teleost scale	-	-	-	-	-	-	-	-	-	-	<0.1	-
Apogonidae	-	-	-	-	-	-	-	-	8.3	0.8	-	-
Bregmacerotidae	-	-	-	-	-	-	-	-	2.3	3.9	-	-
Clupeidae	-	-	-	-	-	-	-	-	2.5	0.4	-	--
Carangidae	-	-	-	-	-	-	-	-	1.1	0.4	-	-
Engraulidae	-	-	-	-	-	-	-	-	20.7	13.2	-	-
Gerreidae	-	-	-	-	-	-	-	-	4	0.4	-	-
Gobiidae	8.7	0.7	7.9	1.5	-	-	-	-	-	-	-	-
Leiognathidae	-	-	-	-	-	-	-	-	4.4	1.2	-	-
Mullidae	-	-	-	-	-	-	-	-	0.1	0.4	-	-
Muraenesocidae	-	-	-	-	-	-	-	-	0.4	0.4	-	-
Platycephalidae	-	-	-	-	-	-	-	-	<0.1	0.4	-	-

Sphyraenidae	-	-	-	-	-	-	-	-	0.3	0.4	-	-
Synodontidae	-	-	-	-	-	-	-	-	1.3	1.6	-	-
Phylum Mollusca	**0.2**		**0.3**		**1.3**			**-**	**2.5**		**10.5**	
Mollusca - unid	<0.1	0.7	-	-	-	-	-	-	-	-	-	-
Class Cephalopoda - total	**-**	**-**	**-**	**-**	**1.2**	**-**	**-**	**-**	**2.4**	**-**	**-**	**-**
Cephalopoda - unid	-	-	-	-	1.2	2.3	-	-	-	-	-	-
Octopoda - unid	-	-	-	-	-	-	-	-	2.4	0.8	-	-
Class Bivalvia - total	-	-	**0.2**	-	-	-	-	-	-	-	**10.5**	-
Bivalvia - unid	-	-	0.2	2.3	-	-	-	-	-	-	0.3	6
Nuculanidae	-	-	-	-	-	-	-	-	-	-	8.8	1.6
Tellinidae	-	-	-	-	-	-	-	-	-	-	1.4	6
Class Gastropoda - total	**0.1**		**<0.1**	**-**	**0.1**	**-**	**-**	**-**	**<0.1**	**-**	**<0.1**	**-**
Gastropoda - unid	-	-	-	-	0.1	4.7	-	-	<0.1	0.39	<0.1	1.3
Cavoliniidae	-	-	-	-	-	-	-	-	-	-	<0.1	0.3
Marginellidae	0.1	1.3	<0.1	0.8	-	-	-	-	-	-	-	-
Class Polychaeta	**0.1**		**0.2**						**-**		**0.6**	
Polychaeta - unid	0.1	0.7	0.2	2.3	-	-	-	-	-	-	0.6	5
Opheliidae	-	-	-	-	-	-	-	-	-	-	<0.1	0.6
Phylum Echinodermata											**0.3**	**-**
Class Ophiuroidea	-	-	-	-	-	-	-	-	-	-	0.3	4.4
Phylum Sipuncula	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**0.1**	**0.3**

7.4.5 DIETARY OVERLAP

Prey composition, in terms of taxa eaten, varied among the six predator species. Significant dietary overlap was detected between four species comparisons: *K. rodericensis* and *E. ransonetti*, *K. rodericensis* and *I. japonica*, *S. macracanthus* and *K. rodericensis*, *U. sulphureus* and *K. rodericensis* (Table 35).

Table 35. Horn's index of dietary overlap for diet biomass between the six demersal species. * denotes significant dietary overlap

	E. ransonetti	*I. japonica*	*K. roderisensis*	*S. macracanthus*	*S. macrolepis*	*U. sulpureus*
E. ransonetti	**1.00**					
I. japonica	0.47	**1.00**				
K. roderisensis	*0.68	*0.60	**1.00**			
S. macracanthus	0.52	0.56	*0.74	**1.00**		
S. macrolepis	0.44	0.25	0.28	0.20	**1.00**	
U. sulphureus	0.48	0.47	*0.64	0.08	0.14	**1.00**

7.4.6 INTERSPECIFIC COMPARISONS OF DIET

The MDS ordination plot shows some clear differences in the diets (% ww) between the six species (Figure 53). These differences in diet are supported by the ANOSIM analyses (R = 0.69, P < 0.001). Pairwise comparisons between species identified a number of significantly different diets (Table 36). However, the diet of K. rodericensis did not differ from those of I. japonica and S. macracanthus (P>0.05). SIMPER analysis identified that the diet of S. macrolepis was consistently the most different from the remaining five species due largely to consumption of more teleosts and less crustaceans than other species. For four of the species comparisons with S. macrolepis, 'unidentified teleost' was the prey taxa that explained most dissimilarity among diets (Table 36, A-D). The diet of S. macrolepis was most dissimilar to that of U. sulphureus with two discrete groups represented in the ordination plot (Figure 53)Six prey taxa contributed to 49% of the dissimilarity (Table 36, A). The teleost components (unidentified. teleost and Engraulidae) were greater in the diet of *S. macrolepis* while the crustaceans were more important for *U. sulphureus*. The diet of *E. ransonetti* was the most similar to *S. macrolepis* with seven prey taxa contributing to almost 52% of the dissimilarity (Table 36, E). Both species ate more teleosts than the four other species.

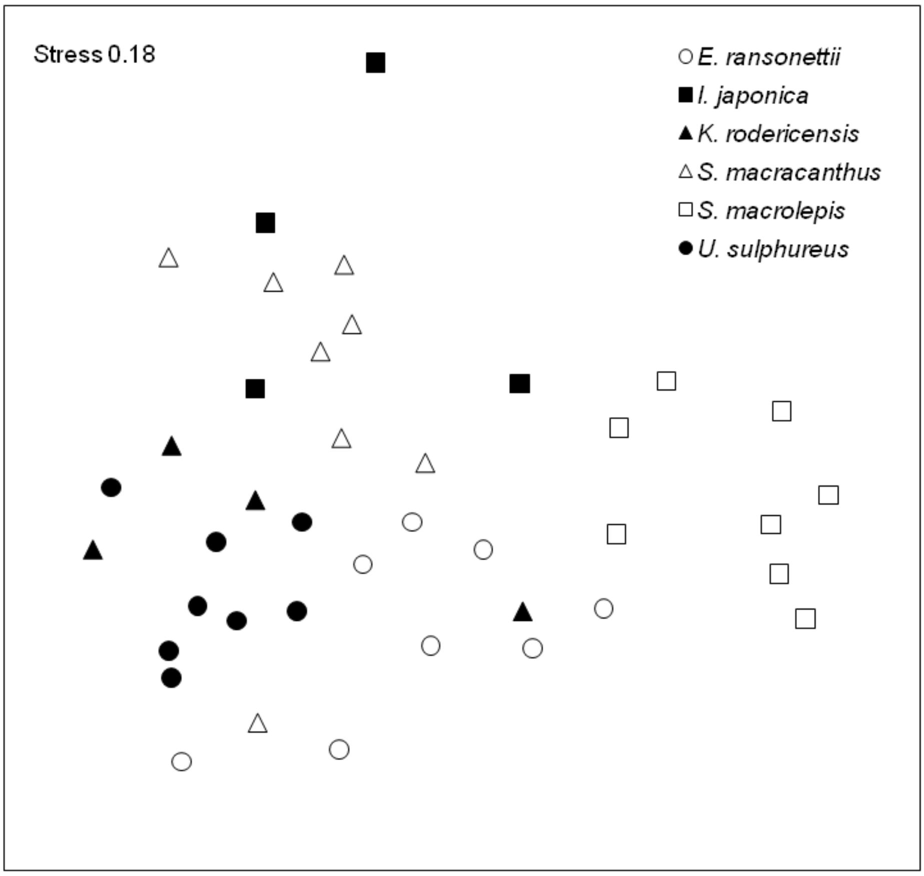

Figure 53. Non-metric MDS ordination plot showing species comparisons of diet composition (% wet weight). Each point represents the mean biomass of prey from a minimum of 5 individuals.

Table 36. Simper results for prey taxa contributions to the dissimilarity between the diets of the fish species. The Global R statistic and P values are shown with average dissimilarity. The prey with the greater contribution to mean % ww is indicated as A: the first listed species or S: the comparative species.

A. *S. macrolepis* vs *U. sulphureus* (R: 0.725, P<0.001, average dissimilarity 84%)

Prey taxa	Greatest contribution	Contribution %	Cumulative %
Unid. Teleost	A	13.2	13.2
Unid. Caridea	S	11.3	24.5
Engraulidae	A	8.2	32.7
Natantia	S	6.1	38.8
Brachyura	S	5.9	44.7
Penaeidae	S	4.3	49
B. *S. macrolepis* vs *I.japonica* (R: 0.895, P<0.01, average dissimilarity 80%)			
Unid. Teleost	A	17.2	17.2
Penaeidae	S	16.7	33.9
Engraulidae	A	11.2	45.1
Portunidae	S	6.5	51.6
C. *S. macrolepis* vs *K.rodericensis* (R: 0.849, P<0.01, average dissimilarity 80%)			
Unid. Teleost	A	13.1	13.1
Engraulidae	A	10.2	23.3
Unid. Caridea	S	9.5	32.8
Portunidae	S	7.1	39.9
Natantia	S	6.6	46.5
Unid. Penaeiodea	S	6.5	53
D. *S. macrolepis* vs *S.macracanthus* (R: 0.94, P<0.001, average dissimilarity 79%)			
Unid. Teleost	A	11.1	11.1
Engraulidae	A	9.8	20.9
Squillidae	S	9.4	30.3
Portunidae	S	7.8	38.1
Unid. Stomatopoda	S	6.6	44.7
Unid. Caridea	S	6.4	51.1
E. *S. macrolepis* vs *E.ransonetti* (R: 0.65, P<0.001, average dissimilarity 69%)			
Natantia	S	12.2	12.2
Engraulidae	A	11.6	23.8
Unid. Teleost	A	8	31.8
Unid. Caridea	S	5.5	37.3
Penaeidae	S	5.4	42.7
Brachyura	S	5	47.7
Unid. Stomatopoda	S	4.1	51.8
F. *E. ransonetti* vs *I. japonica* (R: 0.711, P<0.01, average dissimilarity 75%)			
Penaeidae	S	19.6	19.6
Unid. Teleost	A	13.8	33.4
Natantia	A	9	42.4
Portunidae	S	6.7	49.1
G. *E. ransonetti* vs *S.macracanthus* (R: 0.579, P<0.001, average dissimilarity 68%)			
Natantia	A	10.5	10.5
Squillidae	S	10.5	21
Unid. Teleost	A	10.2	31.2
Portunidae	S	7.7	38.9
Penaeidae	S	7.3	46.2
Unid. Caridea	S	7.1	53.3
H. *E. ransonetti* vs *K.rodericensis* (R: 0.303, P<0.05, average dissimilarity 62%)			
Unid. Teleost	A	14	14
Portunidae	S	8.9	22.9
Natantia	A	8.5	31.4
Unid. Stomatopoda	S	8.1	39.5
Unid. Caridea	S	7.4	46.9

Unid. Penaeiodea	S	7.4	54.3

I. *E. ransonetti* vs *U.sulphureus* (R: 0.551, P<0.001, average dissimilarity 61%)

Unid. Teleost	A	13.2	13.2
Unid. Caridea	S	10.6	23.8
Natantia	A	6.2	30
Unid. Stomatopoda	S	5.5	35.5
Penaeidae	S	5.4	40.9
Brachyura	S	5	45.9
Unid. Penaeiodea	S	4.9	50.8

J. *U. sulphureus* vs *I. japonica* (R: 0.897, P<0.01, average dissimilarity 73%)

Penaeidae	S	14.3	14.3
Unid. Caridea	A	12.6	26.9
Unid. Penaeiodea	S	6.4	33.3
Brachyura	A	6.3	39.6
Portunidae	S	5.8	45.4
Natantia	A	5.1	50.5

K. *U. sulphureus* vs *S. macracanthus* (R: 0.725, P<0.001, average dissimilarity 66%)

Unid. Caridea	A	10.9	10.9
Squillidae	S	9	19.9
Portunidae	S	6.5	26.4
Natantia	A	6.3	32.7
Penaeidae	S	6.3	39
Unid. Teleost	S	5.3	44.3
Unid. Stomatopoda	S	4.7	49

L. *U. sulphureus* vs *K.rodericensis* (R: 0.568, P<0.01, average dissimilarity 58%)

Portunidae	S	8.2	8.2
Unid. Teleost	S	8	16.2
Unid. Stomatopoda	S	6.8	23
Unid. Caridea	A	6.7	29.7
Brachyura	A	6.4	36.1
Penaeidae	A	5.8	41.9
Unid. Penaeiodea	S	5.6	47.5
Squillidae	S	5.6	53.1

M. *S. macracanthus* vs *I.japonica* (R: 0.369, P<0.05, average dissimilarity 63%)

Penaeidae	S	16.3	16.3
Squillidae	A	9.4	25.7
Unid. Teleost	A	9.1	34.8
Unid. Stomatopoda	A	8.6	43.4
Unid. Caridea	A	8.4	51.8

Dietary comparisons with *E. ransonetti* identified that fish prey were again important in explaining the differences in diet. The diet of *E. ransonetti* was most dissimilar to that of *I. japonica* due to differences in the contribution of penaeid prawns and fish to their diets (Table 36, F). Even for the species to which *E. ransonetti* was most similar (*U. sulphureus, K. rodericensis)*, the differences were due to the relative importance of teleosts in their diets (Table 36, H&I). The diet of *U. sulphureus* was similar to *K. rodericensis* (Table 36, L). Higher contributions of brachyuran crabs, fish and stomatopods in the diet of *K. rodericensis* and a higher contribution of carid prawns to the diet of *U. sulphureus* explained most dissimilarity.

7.4.7 GEOGRAPHIC VARIATION IN DIETS

Only four species had sufficient numbers for dietary comparisons between the western and southern Gulf of Carpentaria regions–*S. macrolepis, S. macracanthus, E. ransonetti* and *U. sulphureus*. ANOSIM found that only two species, *S. macrolepis* (R=0.57, P<0.05) and *S. macracanthus* (R = 0.27, P < 0.05), had significantly different diets between the two regions, while the other two did not (P > 0.05). SIMPER revealed that for S. macrolepis six prey taxa contributed 52% of the dissimilarity. Engraulids, apogonids and bregmacerotids were more prevalent in the diets from the western region, while penaied prawns, natantia and leiognathids contributed more to the diets from the southern region. For *S. macracanthus*, SIMPER showed that six prey taxa contributed to 54% of the dissimilarity. Carids, gobiids and unidentified teleost contributed more to their diets in the western gulf compared to penaid prawns, squillids and portunids in diets from the southern gulf.

7.4.8 ONTOGENETIC CHANGES IN PREY COMPOSITION

Only *S. macrolepis*, *S. macracanthus* and *U. sulphureus* had sufficient numbers of immature and mature fish to statistically compare diet composition. There was no difference in the prey composition between immature and mature *S. macracanthus* (P > 0.05). on the contrary the ordination plot for *S. macrolepis* showed a clear difference in the prey composition (% ww) between immature and mature fish (R = 0.2, P < 0.01) (Figure 54a). The analyses showed that 14 prey taxa contributed 90% to the dissimilarity of diets: Engraulidae (15.5%), unidentified teleost (10.8%), Bregamaceridae (8.8%), Penaeidae (8.1%), Leiognathidae (7.8%), Apogonidae (6.9%), Gerridae (5.6%), Synodontidae (4.8%), octopoda (4.7%) natantia (4.6%), Clupeidae (3.8%) caridea (3.6%), Carangidae (3.1%) and Sphyraenidae (2%). Eight prey taxa–Engraulidae, unidentified teleost, Bregamacerotidae, Penaeidae, Synodontidae, natantia, caridea and Sphyraenidae–had higher contributions to the diet of immature *S. macrolepis* while the remaining prey taxa had higher contributions to the diet of mature fish.
The ordination plot for *U. sulphureus* was less clear with a relatively high stress value (Figure 54b). Despite this, a significant difference in diets was detected between immature and mature fish for this species (R = 0.12, P <0.01). Eight prey taxa contributed 56% to the dissimilarity of diets: caridea (9.8%), brachyura (9%), Penaeidae (7.7%), natantia (7.5%), unidentified crustacea (6.6%), stomatopoda (5.9%), penaeoidea (5.6%) and polychaetes (4.4%). Three prey taxa–caridea, natantia and polychaeta–had higher contributions to the diets of immature *U. sulphureus* while the remaining five prey taxa, had higher contributions to the diet of mature fish.

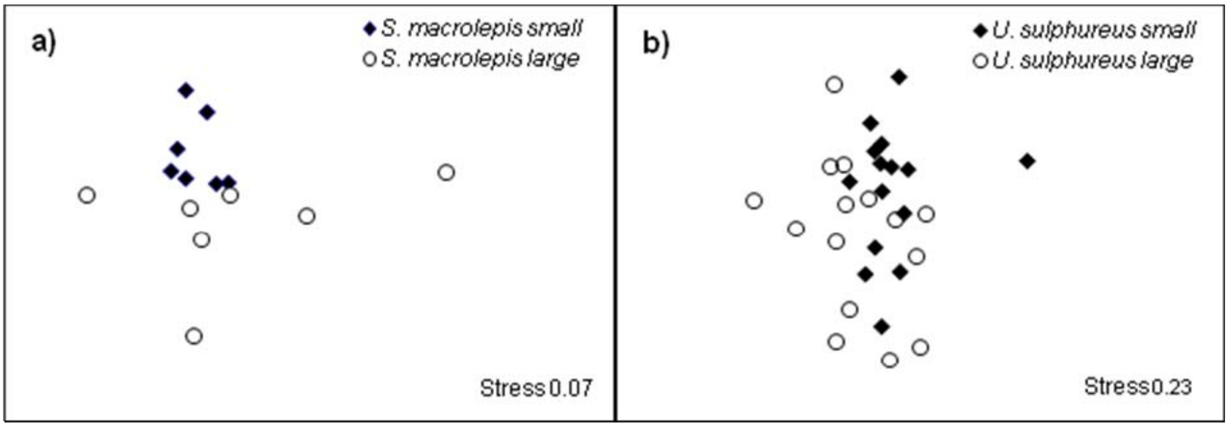

Figure 54. Non-metric MDS ordination plot showing fish maturity comparisons of diet composition (% wet weight) for **a)** *S. macrolepis*; and **b)** *U. sulphureus*. Each point represents the mean biomass of prey from a minimum of 8 individuals.

7.5 DISCUSSION

Strong morphological–dietary relationships were found in this study supporting the eco–morphological belief that fish morphology influences food selection (Wainright and Richard 1995).

Mouth gapes among these demersal species varied significantly, suggesting the existence of trophic niche segregation of these coexisting species. Dietary analysis supported this theory, with prey type and prey size varying significantly among and within some species. Further to this, the dietary overlap was low for those species having different morphological characteristics. In contrast, the species that were morphologically similar (Platycephalids) generally had significant dietary overlap. However there were differences in their main prey types–indicating perhaps a level of prey selectivity which may represent the 'ghost of competition past' (Crowder 1986; Kido 1997).

Differences in dietary compositions of fishes have been related to differences in their overall morphology. Variations in feeding structures (mouth orientation, gape, gill rakers, sensory apparatus) and body shapes can determine feeding ability and prey capture (Werner 1977; Winemiller 1991; Wainwright and Richard 1995; Labropoulou and Eleftheriou 1997; Scharf et al. 2000; Platell and Potter 2001). The predatory fishes examined in this study vary in their mouth structures. Predators that have large gapes, and rapidly increase mouth gape as they grow, tend to eat more fish than invertebrates (Hobson 1979; Scharf et al. 2000). For example, Scharf et al. (2000) found this to be the case for sea raven, *Hemitripterus americanus,* and goosefish, *Lophius americanus,* in waters over the continental shelf of the northeast US. Our study supports this, with the highly piscivorous lizardfish, *S. macrolepis,* also having a large gape which increases fairly rapidly with fish growth. In contrast, the mouth gape of the other predator species was significantly smaller and increased at a significantly slower rate. This was reflected in their diet which consisted primarily of invertebrates (crustaceans).

The widespread existence of asymmetric predator size-prey size distributions in aquatic ecosystems is well known (Juanes and Conover 1995; Mittelbach and Persson 1998). The theory implies that the range of prey sizes eaten expands with increasing predatory body size (which correlates linearly with gape size in this study). *Saurida macrolepis* provides the best example of this in our study (Figure 51), while the other species, except for U. sulphureus, showed a similar albeit less obvious trend. Upenaeus sulphureus does not appear to show an increase in prey size usage for the predator sizes examined (Figure 51). Perhaps the prey size available to this species is more restricted because not only do they have a small gape; they are specialist zoobenthivores that forage mainly over soft sediments using their tactile and chemosensory barbels to detect prey (Platell et al. 1998; Lukoscheck and McCormick 2001). Gape size seemed to correlate well with the differences observed in the distribution of prey sizes eaten by each predator. For example, S. macrolepis had the largest gape relative to body size (Figure 50) and this corresponded to the highest proportion of large prey (>30% of body size, (Figure 52) which included fish such as Engraulids, Leiognathids and Apogonids . In contrast, the predators with the smallest gapes E. ransonetti and U. sulphureus had the highest proportion of small prey (Figure 52).

Of the four platycephalids, three species had significantly similar mouth gapes: *K. rodericensis*, *S. macracanthus* and *I. japonica* while *E. ransonetti* had a much smaller gape relative to size (Figure 50). When comparing the diets of these platycephalids, no significant differences were detected between the diet of *K. rodericensis* and those of *I. japonica* and *S. macracanthus*. This is not surprising considering that these species are morphologically similar and therefore would likely feed on similar prey (Winemiller 1990). However, significant dietary differences were detected between *I. japonica* and *S. macracanthus*, two flathead species which were found to have almost identical gapes. The former had significantly more penaeid prawns in their diet compared to more stomatopods, fish and carid prawns in the later. These differences may be explained by a degree of preferential prey selectivity. However, we do not have available data on the abundance of prey taxa in the environment and therefore it is difficult to conclusively say whether the fish species in this study selected their prey preferentially or because of density dependence.

Prey availability associated with different habitat type may also explain why dietary differences were detected for *S. macrolepis* and *S. macracanthus* between the western and southern Gulf of Carpentaria regions. Long and Poiner (1994) found differences associated with sediment texture and its infauna. Higher infaunal abundance, wet weight biomass and species density was associated with the finer sediments of south-eastern regions of the Gulf of Carpentaria compared to the western regions. Interestingly, they also found crustacean biomass to be highest in the southern regions and this coincides with a higher contribution of crustaceans to the diets of *S. macrolepis* and *S. macracanthus* in the southern gulf found in our study.

Ontogenetic shifts in diet have been attributed to the changing feeding ability of fish. According to Werner (1977) and Hoyle and Keast (1987) as fish grow they become better equipped at handling larger prey which helps maximise their net energy return. This increase in prey size will usually lead to changes in the taxonomic composition of diets. Examples of this were demonstrated in this study with respect to intraspecific changes in diet for *S. macrolepis* and *U. sulphureus*. There are no published studies on ontogenetic shifts in diet for species of the Synodontidae family. This study provides the first evidence that there is perhaps a change in diet composition between immature and mature fish for the synodontid, *S. macrolepis*. Piscivores are generally known to reorient their prey after catching it, to allow the head to be swallowed first and lying on its side (Hoyle and Keast 1987). Therefore prey body depth is likely to be an important factor. While both immature and mature fish were primarily piscivores, a notable difference in the morphology of their fish prey was observed. Our results indicated that immature *S. macrolepis* consumed fish that were fusiform in body shape (Engraulidae, Bregamaceridae, Synodontidae, Sphyraenidae). In contrast, mature fish fed almost exclusively on deeper bodied fish from families such as Leiognathidae, Apogonidae, Gerridae, Clupeidae (*Pellona ditchella*) and Carangidae. Although prey body depth was not measured, it was noted that the prey species, consumed by the smaller predators, have body depths considerably smaller than those consumed by the larger fish for the prey lengths observed. This suggests that larger *S. macrolepis* are able to select these preys due to an increased gape size or perhaps a greater ability to move higher into the water column. In contrast, the deeper bodied prey species may not be exploited by the smaller predator group due to gape-limitation or perhaps the requirement of greater prey handling time which can be detrimental in terms of energy expenditure, risk of losing prey or increasing their exposure to predation themselves. Evidence of prey selectivity based on body morphology has been recognised in several piscivorous freshwater fish predators, with several laboratory studies showing that deeper-bodied prey species are less preferred than more fusiform prey (Wahl and Stein 1988; Hambright 1991; Nilsson and Bronmark 2000).

The goatfish *U. sulphureus* also showed evidence of an ontogenetic shift in prey. Platell et al. (1998) identified significant prey differences between small and large fish in two other species of goatfish (Mullidae) found in south–western Australia. Labropoulou and Eleftheriou (1997) found size related differences in the diet of another goatfish species on the Cretan shelf (north-eastern Mediterranean). In our study, immature *U. sulphureus* ate more carid prawns, natantia and polychaetes, while larger mature fish ate more brachyuran crabs, penaeid prawns and stomatopods. This might suggest a generalised shift from small, thinner shelled, softer bodied prey to larger, thicker–shelled prey, as has been found in many fish species (Norton 1995; Hyndes et al. 1997; Platell et al. 1998).

Most fish predators utilize one or two basic prey encounter tactics, either lying in wait to ambush prey or cruising continually to locate prey (Greene 1986). In addition to mouth gape, which was discussed earlier, other features such as body and fin shape are important factors in influencing locomotion and therefore prey selectivity. For example, *S. macrolepis* are fusiform in body shape and have a deeply forked tail which suggests that they are capable of sustained swimming (Helfman et al. 1997). They also possess long teeth for holding prey and an upturned mouth which, according to Winemiller (1990) and Motta et al. (1995), are features that coincide with greater piscivory. Because of their large gape and mobility this species is able to be selective in its prey size usage and therefore selecting larger prey will maximize their energy return. This may be particularly relevant for *S. macrolepis* with a gape circumference greater than 100mm, where the inclusion of smaller prey in their diet was not detected (Figure 51). Platycephalids are typically known as short-burst ambush predators that ‘lie in wait’ in the sediment until prey come into range (Helfman et al. 1997). These species utilize a range of prey sizes because they are less mobile and therefore less selective of prey size. However, there is evidence to suggest that the platycephalid, *E. ransonetti*, is not the typical ambush predator. Its small gape, narrow eel-shaped body and emarginate caudal fin (Froese and Pauly 2009) suggest it has greater mobility than the more robust platycephalids examined in this study. Due to the constraints of gape limitation it is likely to have a more mobile and selective feeding strategy that increases the potential interaction with appropriately sized prey. The composition of its diet supports this theory with more mobile teleost prey contributing significantly to its diet which contrasts with the diets of the other platycephalids examined. Greater evidence of piscivory for this species was demonstrated by Yamashita et al. (1987) in the Gulf of Thailand where 63% of its diet was fish. Also the significant occurrence of small benthic macro-invertebrates in their diet, relative to the other platycephalid species, such as tanaeids, isopods and amphipods, suggest perhaps an element of mobile foraging behaviour rather than a ‘lie in wait’ ambush technique.

7.5.1 BROAD DIETARY COMPARISON WITH OTHER STUDIES

The composition of diets for all species examined in this study supports the general findings of other studies, in that Synodontids are piscivores and Platycephalids and Mullids feed primarily on benthic crustaceans. Synodontids are known to be highly piscivorous (Salini et al. 1994; Kulbicki et al. 2005). According to Froese and Pauly (2009), the diet of *S. macrolepis* is unknown. However a study, by Salini et al. (1994) in northern Australia, described the diet of *Saurida* sp. *2* (identified from Sainsbury et al. 1985) which according to a revision by Inoue and Nakabo (2006) is probably *S. macrolepis*. Salini et al. (1994) found that the diet of three synodontid species in the Gulf of Carpentaria, Australia, each contained over 70% fish (dry weight). The diet of the goatfish, *U. sulphureus,* is only described from the Red Sea (Boraey and Soliman 1987) where polychaetes were dominant (>80%), followed by benthic crustaceans (14%). In contrast, we found that polychaetes made up less than 1% of their prey biomass and benthic crustaceans around 86%. Several other studies have described diets of numerous goatfish species (Labropoulou and Eleftheriou 1997; Platell et al. 1998; Kaya et al. 1999; Unluoglu et al. 2002; Krajewski et al. 2006). These studies show that goatfish have a broad diet including various benthic crustaceans, polychaetes, molluscs, echinoderms, sipuncula and teleosts. Variations in the main prey types between these studies could be a function of prey availability however some studies have determined that co-existing goatfish do feed differently. For example, Labropoulou and Eleftheriou (1997) reported a difference in morphology (intestine length, number of gill rakers) and foraging behaviour of two sympatric goatfish. These morphological differences coincided with a segregation of their feeding niche, where similar sized prey were utilised however the main prey types were contrasting–one species fed mostly on polychaetes while the other on decapods. In our study, decapod crustaceans were the main prey item for the goatfish, *U. sulphureus*; particularly carid prawns (59% of prey biomass). Intestinal length and number of gill rakers were not measured in our study but would be an interesting comparison with other goatfish observed to feed mostly on decapods. Platycephalid fishes (flathead) throughout the world generally feed mostly on benthic crustaceans

(Yamashita et al. 1987; Platell and Potter 2001; Froese and Pauly 2009). In contrast, Bulman et al. (2001) found that in south-eastern Australia, *Platycephalus bassensis*, fed primarily on fish (91% by wet weight) as did, *Platycephalus arenarius,* in northern Australia (Salini et al. 1998). These flathead species grow to a large size (>40cm SL) (Froese and Pauly 2009) and the diets described were from fish with mean size of >25cm SL, compared to a maximum mean of approximately 16cm SL in our study. Therefore greater piscivory could be related to larger gape sizes and perhaps and extended strike range associated with these larger species. Of the platycephalid species examined in our study, only the diets of *E. ransonetti* and *I. japonica* have been previously described (Yamashita et al. 1987). In contrast to our study, they found that similar sized *E. ransonetti* were mostly piscivorous in the Gulf of Thailand with fish contributing to over 60% of its diet, compared to 35% here. With respect to *I. japonica*, both studies indicate that benthic crustaceans are the main prey items; however crabs contributed four times more to their diet in the Gulf of Thailand. Our study provides the first dietary descriptions for *S. macracanthus* and *K. roderiensis* (Table 34).

7.5.2 PENAEID PRAWN PREDATION

The Northern Prawn Fishery (NPF) is a multi-species demersal trawl fishery that is one of Australia's largest (~ 250 000 km^2) and most valuable fisheries (Dichmont et al. 2007). Understanding the predation on penaeid prawns is therefore important in this region and has already been investigated in the NPF (Brewer et al. 1991; Salini et al. 1994; Brewer et al. 1995; Haywood et al. 1998). However, this study provides additional information on penaeid prawn predation as five of the six species investigated here, were not investigated previously. Predation on penaeid prawns was evident for all species examined in this study, particularly *I. japonica* which had 51% of its overall prey biomass consisting of penaeids and were found in almost 20% of their stomachs with prey content. The remaining species consumed much smaller amounts and ranged between 0.7 and 5.8% of their diet. It is unlikely that significant consumption of commercial sized prawns occur on prawn trawl grounds by these species due to gape limitation. However, these predator species are abundant in shallow inshore waters at depths <10m (CSIRO unpublished data), where post larval prawns and sub adult prawns migrate to and from nursery grounds, particularly during spring and summer (September to Febraury) (Crocos and van der Velde 1995; Vance and Pendrey 2008). At these smaller life cycle stages, commercial prawns are likely to be consumed by these smaller gaped predators.

In conclusion, this study indicates that differences in the morphology, in particular the gape of these sympatric demersal species, appear to correlate closely with food selection and trophic separation therefore assisting in their coexistence. Previously unknown species-specific dietary information is also provided. Understanding the trophic relationships among these fishes provides important information for ecosystem models developed to support ecosystem-based fisheries management.

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APPENDIX 8 METHODOLOGY BEHIND THE SPATIAL DISTRIBUTION MODEL

8.1 CLASSIFICATION AND REGRESSION BY RANDOM FOREST

We adapted and used a regression tree[2] method for the the biophysical and spatial prediction of GoC benthic biota. The technique, commonly known as *random forests*, was first proposed by Breiman (2001) as a powerful non-parametric estimation method. In the field of ecology, random forests have been considered by Prasad et al. (2006) and were applied by Garzon et al. (2006), Garzon et al. (2007) for estimating species distribution and habitat suitability, as well as by Schwartz et al. (2006) for predicting future species distributions.

Random forests may be described as bagging regression trees. That is, many bootstrap samples are selected from the data and a binary decision tree is fit to each. When splits are made at each node in each tree, a subset of predictors is chosen at random for evaluation. The predictor and the split point is chosen so as to minimize the sum of squared residuals.

Trees constituting the random forest are not pruned using cross-validation, but are instead fully grown (i.e. over-fit). These measures act to increase the random variability of the constituent trees within the forest so as to reduce correlation in the errors of the ensemble of estimators. Given:

- sufficient independence between estimators,
- some ability of the trees to approximate the underlying function, and
- a reasonably large number of trees,

then the final estimate produced by averaging estimates over the ensemble can be expected to have good bias/variance properties (Boinee, 2005). As an estimator, they are based on sound bootstrapping principles (Davison and Hinkley, 2003), and a growing body of empirical results suggests random forests are an effective applied estimation tool. By averaging estimates produced by a large set of trees, they avoid the instability, model uncertainty, and much of the bias associated with any single regression tree. A significant attraction of random forests is the manner in which they satisfy the competing goals of high model flexibility and strong generalization ability.

Much of the power of random forests is derived from the bootstrap aggregation procedure, in which random samples (of the same size as the original dataset) are drawn with replacement. Due to the replacement property, a bootstrap sample will incorporate ~ 2/3 (‘in-bag’ (INB)) unique samples from the original data. Thus, ~ 1/3 of the CART will not have ‘seen’ any given datum. Utilizing only that subset of CART in calculating the average random forest estimate yields an ‘out-of-bag’ (OOB) estimate, which is similar to a having an independent ‘validation set’ for un-inflated estimates of model performance.

As mentioned above, the CART constituting the random forest estimator is based on optimizing sums of squares or sums of absolute deviations. From standard regression theory, it is known that departures from normality decrease the efficiency of such estimators (Box and Cox 1964). In realistic ecological surveys, outliers often exist, and the variation of the response may be related to the mean estimates. Non-constant variation gives greater weight to data with higher variation, and therefore, as with other forms of regression, it is often desirable to transform the response variable when applying CARTs (De'Ath 2002).

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[2] Trees are a standard tool in statistical pattern recognition and will not be described in detail here. Readers may refer to any text in pattern recognition or machine learning, e.g. (Duda et al. 2001) for an introduction.

8.2 TRANSFORMATION IN NON-PARAMETRIC ESTIMATION

One way to improve the efficiency of sum of squares–type regressors is to consider monotone transformations of the response. A power transformation of the estimand is one approach to better satisfy the assumption of normal and constant variance, to 'tame' large outliers, or more generally, to find the optimal scale in which the fixed effects influence the response (Rubin 1984). A given power transformation and its associated model may be compared with other possible transformations by some measure of model efficacy (e.g. heteroscedacity of model residuals). We propose that the efficiency of modern non-parametric regression estimators can be improved by implementing this well-established technique. In particular, we expect that random forest estimation of ecological distributions can be more effective using empirical transformation of the biological response.

In classical statistics, the issue of an inhomogenous error distribution has been approached by transformation (Box and Cox 1964) or weighting of the response (Han 1987). Given an arbitrary response vector $\boldsymbol{y} = y_u = (y_1, y_2, ...y_n) \geq 0$, the power transform may be defined as:

$$y_u^{(\phi)} = \begin{cases} \dfrac{y_u^{\phi} - 1}{\phi} \phi \neq 0 \\ \log y_u \quad \phi \neq 0 \end{cases}$$

We are concerned with ecological count data, for which outliers and over-dispersion are the primary issues, and which by definition does not contain records in the region $^{(0,1)}$. In this context, it is sufficient to simply consider the transformations:

$$y_u^{(\phi)} = y_u^{\phi} \phi \in (0,1].$$

It was also shown that given a linear model of the transformed response:

$$y_u^{(\phi)} = x_u \beta + e_u$$

where the response is a linear combination of the covariates x_u, and e_u is iid and drawn from a known distribution, then ϕ may be optimized simultaneously with the parameters β so as to maximize the likelihood in relation to the untransformed observations (Box and Cox 1964).

This method of optimizing ϕ cannot be applied to random forests for a number of reasons, first among them being the fact that a likelihood criterion does not exist. We can write a more general model:

$$y_u^{(\phi)} = g\left(x_u\right) + e_u$$

where $g\left(x_u\right)$ is a random forest estimate (or more generally, the output of some arbitrary non-parametric non-linear estimator), and where the error term is iid, but has an unknown distribution. In place of a likelihood, we consider applying Kendall's rank-correlation to the response and the model estimates

$$\tau\left(y_u^{(\phi)}, g\left(x_u\right)\right) = \frac{4k}{n\left(n-1\right)} - 1,$$

where n is the number of observations, and k is the sum, over all observations, of observations ranked after the given observation, by both rankings (Kendall 1938). Considering the situation where $g(.)$ is linear with respect to finite parameters, (Han 1987) shows that as the true e_u departs from normality, maximizing Kendall's τ criterion leads to an estimator with greater efficiency than the corresponding likelihood criterion. Further it is shown that ϕ uniquely maximizes τ in large samples. We do not have corresponding theory showing that these results hold when $g(.)$ is a non-linear stochastic estimator. However, given that $g(.)$ is optimized so as to minimize the residual (square) deviations, we have a strong expectation that similar properties should apply. The approach of maximizing Kendall's τ criterion for a random forest estimator will be tested empirically in this study. Although we are lacking methods to estimate ϕ and $g(.)$ simultaneously, the speed of random forest estimation, combined with modern computing resources, entails that iterative estimation of ϕ and $g(.)$ is relatively undemanding.

8.3 BIOLOGICAL AND PHYSICAL DATA

Survey and historical data from northern Australia that covered the NPF were used for random forests with Box-Cox power transformation model –i.e. species distribution model. Most of these data are explained in detail by Hill et al. (2002), Rochester et al. (2007) and in the Appendix 8. Physical environmental data used as correlates to the biological data were obtained from each sampling station from a compilation of seabed characteristics and ocean climatology from Rochester et al. (2007), resulting in a total of 25 variables. Physical data were interpolated, resampled and mapped to a 1 minute cells or 0.0167° resolution (i.e. ~11 km) across the GoC region between 10° - 18° S and 135° - 142° E, resulting in 230, 400 grid cells (480 x 480 cells)..

APPENDIX 9 SIMULATED ECOLOGICAL EFFECTS OF DEMERSAL TRAWLING ON THE GULF OF CARPENTARIA ECOSYSTEM

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9.1 ABSTRACT

Australia’s legislation requires all export fisheries to demonstrate ecological sustainability. Australia’s Northern Prawn Fishery (NPF) is a large multi-species demersal trawl fishery that targets tiger and banana prawns, primarily within the Gulf of Carpentaria, in northern Australia. Due to the indiscriminate nature of demersal trawling, the fishery has the potential to disrupt the functionality of the supporting ecosystem by catching large biomass of hundreds of species of fish, reptiles and invertebrates, and well as habitat-forming species, such as gorgonians, sof-corals and sponges. An Ecopath with Ecosim model (EwE6) was constructed to explore the ecological effects of demersal trawling on the Gulf of Carpentaria ecosystem from 1970 to 2010, and explore to potential effects of the recent changed effort regimes from 2005-2010. Prior to undertaking modelling using Ecosim, the model was calibrated using time-series data of biomass, fishing mortality and catch for tiger prawns to ensure that known historical changes in biomass and catch could be reproduced, before attempting to forecast ecological impacts. The modelled the food web described well the dynamic of prawn catches with some poor fitting in the early years, probably due to the uncertainty in these historic values predicted for the fishing effort. The model positions the main prawn species as relative low abundance intermediate consumers in which tiger prawns could presumably play a keystone function in the ecosystem –i.e. they have a disproportionate + and - trophic effects despite their relative low biomass. The model also describe well the historical dynamics of trawling over the past 40 years showing clear biodiversity impacts by lowering the mean trophic levels of the catches (TL). The main impacts occurred in the expansion 70s to 80s periods when the TL was the lowest. However, when fishing effort was reduced (in the mid to late 80s) the TL increased steadily to values in 2010 close to the ones estimated for the mid 70s. The simulated historical effects of trawling showed to be positive and negative impacts on relative biomass of biodiversity. The small sharks, banana prawns, mud crabs large gastropods (conchs) and echinoids (urchins) were the most negatively affected functional groups with reductions up to 50% (small sharks). Conversely and as expected, discarded bycatch increased substantially with fishing > 250%), but also did increase tiger prawns (150%), sand crabs and the large shark groups. The evaluation of the 2005-2010 reductions of fishing showed to have very small effects in the overall biomass of all functional groups, with biomass variation of <20%, with similar groups responding positively and negatively. In all simulations, tiger prawns showed that are in a stock-rebuilding trajectory, largely due to the reduction and adjustment of fishing effort. Our results suggest that indeed the ecosystem in the GoC is largely influenced by trawling, but due to the drastic reduction of fishing (from 286 vessels in 1981 to 52 vessels in 2009) these impacts on biomass removal and trophic levels have been reduced. These rapid responses suggested that the modelled GoC ecosystem was resilient to fishing, but this does not means that fishing does not have impacts. Our simulations showed clearly the individual groups and overall biodiversity are affected but their recover rapidly when trawling is suppressed or reduced. Despite all the limitations of the mass-balance models such as EwE6, we believe that offers probably one of the most robust modelling approaches to model the complex tropical ecosystem under fishing. Or results showed that the model simulated well the trophic interaction affected by fishing fisheries
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9.2 INTRODUCTION

The management paradigm of many fisheries worldwide has undergone a significant transformation in the past decade, shifting from a single species (i.e. target species) focus to considering fishery impacts on entire ecosystems (Hall and Mainprize, 2004). This management approach - widely referred to as Ecosystem-Based Fisheries Management (EBFM) - has arisen in response to an increasingly body of literature that demonstrate the negative impacts of fishing on structure and functionality of marine ecosystems and the subsequent implementation of stringent worldwide environmental and fisheries policies and legislation that demand fisheries take greater responsibility for managing the direct and indirect impacts on the supporting ecosystem (NMFS, 1999; NRC, 1999; Garcia et al., 2003; Pikitch et al. 2004; Scandol et al., 2005).

In the Gulf of Thailand for example, Christensen (1998) showed how intensive fishing was responsible for a decline in the average trophic level of the ecosystem from 3.35 to 3.15 over a 25 years period. This progressive shift to targeting species in lower trophic levels once larger species no longer become viable targets – known as 'fishing down the food web' - can result in ecosystems becoming increasingly dominated by fast-growing 'weed' species, often having lower economic importance than previous target species. Ecosystem regimes shifts may result from these fishing-induced impacts, which may have negative economic consequences for fisheries and society (Carscadden et al. 2001, Jackson et al. 2001, Walters & Kitchell 2001, Daskalov 2002).

Similar scenarios have occurred in the tropical Pacific Ocean where (Ward & Myers 2005) demonstrated that an increase in pelagic stingrays was a result of released predation pressure by apex pelagic predators that had undergone population declines due to decades of intense fishing. In contrast to these top-down effects of fishing on marine ecosystems, impacts on intermediate and lower trophic levels can variable effects that can flow effects either upward, downward, or in both directions through the system. A number of studies have shown how biomass changes in primary productivity propagate through the system causing parallel changes in the biomass of higher trophic levels (Aebischer et al. 1990, Chavez et al. 2003). In regions where coastal upwelling is common, trophic cascades can occur in both directions if there is a change in the biomass of a few highly abundant species in intermediate trophic levels that act as both key prey and predators (Cury et al. 2000).

The Northern Prawn Fishery (NPF) is one of Australia's largest (~ 1 million km2) and most valuable fisheries (AU $78 m in 2007) that have operated continuously since the early 1970s. The fishery had been the centre of controversy for many years over its impacts on the benthos and large marine animals such as turtles (Brewer et al. 2008). Such tropical demersal trawl fisheries are one of the least selective marine fisheries that capture an enormous number of mobile and sessile organisms ranging from small crustaceans to large sharks (Stobutzki et al. 2001, 2003). Their demersal trawl gear such as those used in penaeid fisheries, often come into contact with the sea floor and alter the structural heterogeneity of habitats used by many animals for feeding and shelter (Jennings et al. 2005). Consequently, such fisheries have the potential to significantly alter the structure of marine communities if not managed carefully.

Unfortunately, the enormous complexity of this tropical marine ecosystem and the general paucity of quantitative data available for most species — particularly for rare or low-value bycatch species that can comprise a significant component of both catches and the ecosystem (Stobutzki et al. 2001)— means that demonstrating the sustainability of ecosystems supporting fisheries can be a difficult prospect. However, recent developments in ecosystem modelling software, such as the widely-used Ecopath package (www.ecopath.org), can provide a useful theoretical framework and tools with which the ecological effects of fishing and other perturbations, such as climate change, can be explored in isolation or in unison. This can assist in prioritising future research or management of particular components of the ecosystem.

The objective of the present study was to develop and use an ecosystem model to simulate the ecological effects of trawling on the benthos to: i) describe the spatial distribution of habitats and assemblages and evaluate the historical footprint of the NPF, and (ii) to evaluate the ecological consequences of reduced fishing effort in the NPF since 2005.

9.3 MATERIALS AND METHODS

We constructed an ecosystem model of the GoC using Ecopath with Ecosim software, which employs a trophic mass-balance model. Because this model was used to primarily investigate fishing effects on demersal and benthic organisms, we disaggregated lower trophic level functional groups and aggregated higher trophic levels (e.g. sharks, pelagic fishes) in order to focus on the ecological effects of prawn trawling in the GoC using dynamic simulations addressing the stated objectives. The model was constructed to distil data on the biomass of different functional groups, their production rates, diets and consumption rates, fisheries and the environmental characteristics, and other information to describe the trophic flows and non-trophic attributes of the ecosystem.

9.3.1 THE ECOPATH WITH ECOSIM APPROACH

Ecopath trophic models describe the state of energy flows in a food web. They are designed to include all biotic components of an ecosystem, and biomass wet-weight (used here) is usually the ‘currency’.

The first Ecopath master equation (Equation 1) expresses the law of conservation of mass or energy and it indicates the basic input parameters. This equation balances a group’s net production (terms to the left of the equal sign) with all sources of mortality, migration, or change for that group (terms to the right). More specifically, it says that the net production of a functional group equals the sum of (1) the total mass (or energy) removed by predators and fisheries, (2) the net biomass accumulation of the group, (3) the net migration of the group’s biomass, and (4) the mass flowing to detritus.

$B_i \cdot (P/B)_i \cdot EE_i = Y_i + \Sigma\,[B_j \cdot (Q/B)_j \cdot DC_{ji}] + BA_i + NM_i$ **(1)**

B_i and B_j are biomasses of prey ($_i$) and predators ($_j$) respectively;

P/B_i is the production/biomass ratio, equivalent to total mortality (Z) in most circumstances (Allen, 1971);

EE_i is the ecotrophic efficiency; the fraction of the total production of a group utilized in the system;

Y_i is the fisheries catch per unit area and time (i.e., $Y = F*B$);

Q/B_j is the food consumption per unit biomass of j;

DC_{ji} is the contribution of i to the diet of j;

BA_i is the biomass accumulation of i (positive or negative); and

NM_i is the net migration of i (emigration less immigration).

The Ecopath model is a system with as many such linear equations as there are functional groups. The energy balance (conservation of matter) within each functional group is ensured with the second master equation (Equation 2):

Consumption = production + respiration + unassimilated food **(2)**

The implied thermodynamic constraints of this equation underscore the power of *Ecopath* models as a focal point for refinement of ecosystem information. The need to reconcile energy production and demand among components of the food web narrows the possible ranges of parameter estimates for particular groups. Inclusion of a biomass accumulation factor and migration factor in the general *Ecopath* equation distinguishes *Ecopath* modelling as an 'energy continuity' approach rather than a strictly 'steady state' approach. Conservation of energy (*continuity*) is assumed for every identified component of the ecosystem, and the whole system. This basic constraint enables representation of changes in populations (i.e., functional groups) when expressed in dynamic form.

Ecopath was refined considerably with the dynamic simulation routines *Ecosim* and *Ecospace* (Walters et al., 1997; Walters et al., 1999; Pauly et al., 2000; Walters et al., 2000). In *Ecosim*, information in the static *Ecopath* file is re-expressed in a dynamic formulation (*Equation 3*).

$$\frac{dB_i}{dt} = g_i \sum_j Q_{ji} - \sum_j Q_{ji} + I_i - \left(M0_i + F_i + e_i\right) \cdot B_i \tag{3}$$

dB_i/dt is the change in the biomass of group i (B_i) over time, g_i is the efficiency of the conversion of food into growth, Q_{ji} is the rate of consumption by predators j of prey group i, I_i is the immigration rate, $M0_i$ is the natural mortality rate, F_i is the fishing mortality rate, and e_i is the emigration rate.

The dynamics and sensitivity of Ecosim models is largely controlled by the consumption rates (Q_{ji}) (Equation 3), which are limited by the proportion of a given predator group's prey that exist in a vulnerable state. Prey vulnerability is controlled within the expression of consumption rate by a user-specified (or calculated) transfer rate of prey movement between vulnerable and invulnerable pools (v_{ij} and v'_{ij}), thus representing the universal community stabilizer of prey refugia. The consumption rate (Q_{ji}) expressed in Equation 4 includes the prey vulnerability parameters.

$$Q_{ji} = \frac{v_{ij} \cdot a_{ij} \cdot B_i \cdot B_j \cdot T_i \cdot T_j \cdot S_{ij} \cdot M_{ij} / D_j}{v_{ij} + v'_{ij} \cdot T_i \cdot M_{ij} + a_{ij} \cdot M_{ij} \cdot B_j \cdot S_{ij} \cdot T_j / D_j} \tag{4}$$

a_{ij} is the rate of effective search for prey i by predator j, T_i and T_j are the relative feeding times of prey i and predator j, S_{ij} is the user-defined seasonal and long-term environmental forcing effects, M_{ij} is the non-trophic mediation forcing effects, and D_j represents the effects of prey handling time by predators, which further limits consumption rates. See Walters et al. (1997), Christensen and Walters (2004) and the Ecopath with Ecosim user's guide for more information. The free software can be downloaded from www.ecopath.org.

Parameters v_{ij} and v'_{ij} represent prey vulnerabilities, or the rate of exchange of biomass between two prey behavioural states: a state in which all predators have full access to prey and a state in which prey have full refuge from predators. Prey use refugia in real ecosystems. Thus, not all prey biomass is vulnerable to predation at any given time, and predator-prey relationships are limited by behavioural and physical mechanisms. Ecosim is designed so that the user can specify the type of trophic control (Lotka-Volterra type vs. donor control) that mediates any interaction in the food web. Maximum consumption rates are hypothesized, and thus the rate of exchange of biomass (v_{ij}) that a predator normally exerts. For high prey vulnerability (v_{ij}) the functional relationship approximates a mass-action flow, or Lotka-Volterra type interaction implying a strong 'top-down' effect. For low prey vulnerabilities the functional relationship approaches a donor-controlled (bottom-up) flow rate so v_{ij} is the maximum possible instantaneous mortality rate that j can cause on i (Walters et al., 1997).

Prey vulnerabilities can be specified by adjusting the proportion of prey in vulnerable and invulnerable states (pools) via adjustment of the v values, which are scaled such that pure Lotka-Volterra (top down) type control and pure donor control. In the real world, this mixture of trophic control is mediated by temporal or spatial refugia, or by the relative primacy of physical and biotic forces in regulating communities, i.e., predator-prey interactions. Derivation of v values for each functional group is discussed in detail under the section fitting Ecosim to time series data.

9.3.2 MODEL CONSTRUCTION PROCEDURE

The following steps were followed when constructing the *Ecopath* model:

1. **Define the ecosystem in space and time** – The spatial extent of the system and the represented time period must be clearly defined. Parameter estimates are expressed in annual units.
2. **Define functional groups** – Myriad species comprise interaction webs, but these species must be aggregated into related groupings that make sense in terms of ecological function, and the questions of interest.
3. **Estimate basic parameters** for each functional group.
4. **Estimate fisheries information** – Landings, discards and discard fates is derived and entered for each fishery gear type. Effort and catch time series should also be specified such that the catches and discards in the initial modelled period is expressed properly over time.
5. **Estimate additional *Ecopath* parameters** – Detritus fates, assimilation rates, multi-year trends, temporal distributions, and habitat associations.
6. **Enter parameters** into the windows-based input interfaces (see www.ecopath.org).
7. **Balance the model** according to thermodynamic constraints.
8. **Calibrate Ecosim model** – the Ecosim model is fit to time series data in order to confidently make future predictions
9. **Define the habitats and parameters for Ecospace** – define the types and extent of ecosystem habitats and enter and estimate fisheries and ecological spatial explicit parameters
10. **Run Ecosim and Ecospace scenarios** – "what if" scenarios are run by varying fisheries, biological or environmental parameters.

9.3.3 SPATIAL AND TEMPORAL EXTENT OF THE MODEL

The Gulf of Carpentaria (hereafter referred to as the GoC) is a large (370,200 km^2), shallow (<70 m) tropical marine embayment located along Australia's northern coastline between Cape York Peninsula to the east and the Wessel Islands and Arnhem Land to the west. The Arafura Sea and New Guinea lie to the east. The colored area of Figure 55is the area included in the present Ecopath model of the GoC.

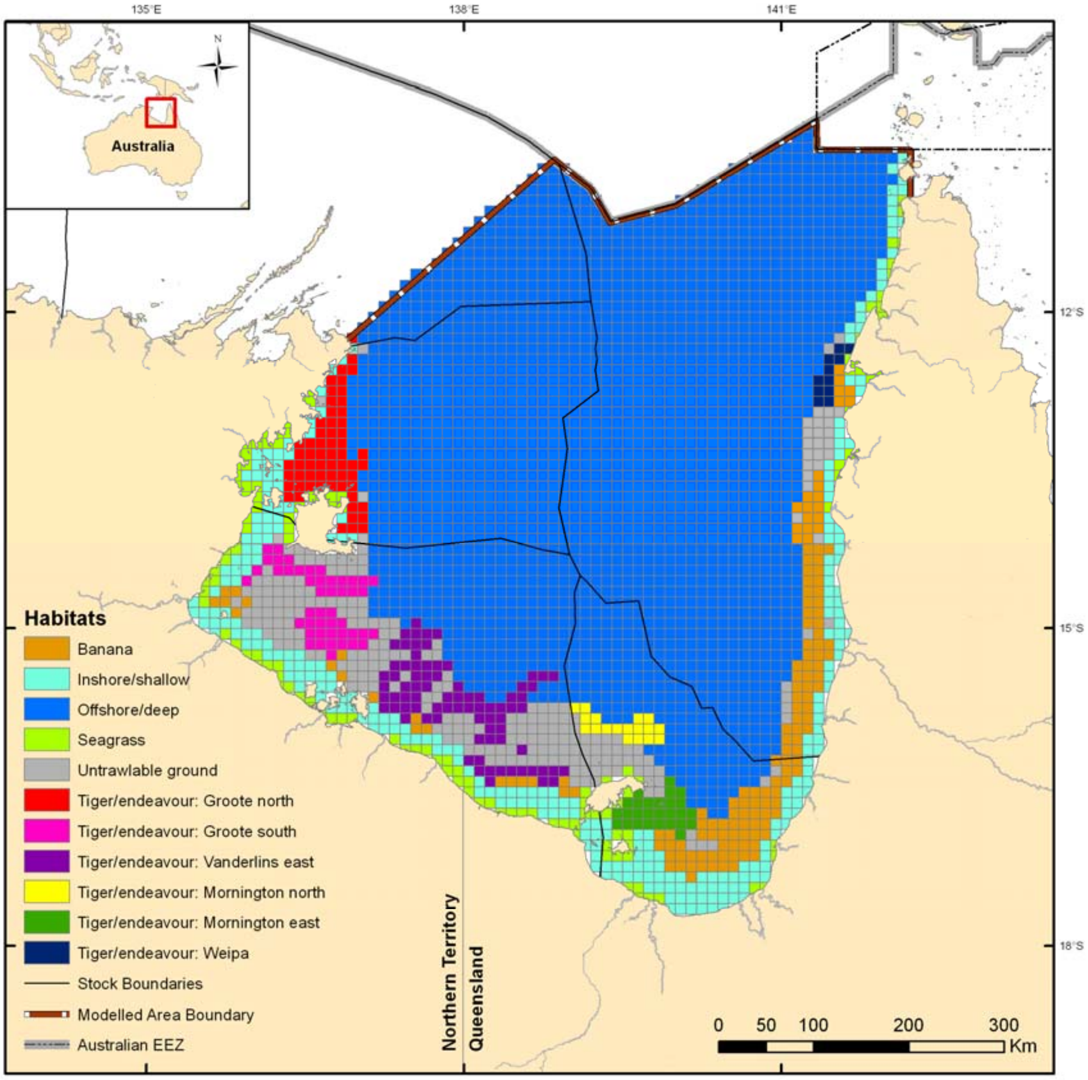

Figure 55. The area included in the present Ecopath model of the Gulf of Carpentaria, Australia. Colours represent the Ecospace habitats of the gulf whore descriptions are for which estimated areal extents are shown in Table 1.

9.3.4 MODEL STRUCTURE

The year 1990 was chosen to characterise a static description of the trophic flows in the GoC. The biota of the Gulf were aggregated into 60 functional biological groups based on the best information available (Table 37). These groups were chosen based on ecological 'guild' similarity criteria such as preferred habitat, feeding type and diet, sizes, and rates of production and consumption, in addition to the questions of interest in this study. Functional groups act as a single biomass pool, or species, even though these functional groups are made up of numerous species. Thus, the aggregation of species into these functional groups may affect model dynamics in some cases. The approach taken here was to emphasise disaggregation benthic and demersal groups in order to articulate the mechanisms and cascades that may be important in the real ecosystem, while aggregating other groups, such as pelagic fish, sharks and seabirds, that we considered not to have strong bearing on the outcomes of the simulations of interest in this project. Single-species stock models that characterise the interaction of different life stages (juvenile and adult) of functional groups represented by single species were embedded in the overall model to increase the robustness of the simulated dynamics.

9.3.5 SOURCES OF BASIC BIOLOGICAL PARAMETERS AND FISHERY DATA

The key biological parameters (biomass, productivity, diet composition) for each functional group in the model were estimated from stock assessments, primary research data, fishery data recorded in logbooks or by scientific observers, or the literature, and are shown in Table 2. The model is based upon a robust fish diet composition matrix (Table 39), which is based upon stomach content analyses from a concurrent study (Tonks et al., *this volume*; Dell et al. this volume).

The chosen year of 1990 that characterised the static description of the trophic flows in the GoC to provide a 'convenient starting point' for the dynamic simulations using the Ecosim model since good quality diet and fishery catch data were available for that time. However, in 1990 the shark, mackerel and some other fish stocks of northern Australia may have been recovering from over-exploitation by the Taiwan-Australia joint venture activities, Taiwanese and then Thai trawling would have had some impact at that time as well, but most of those activities were outside the GoC so it is difficult to ascertain what effects might have manifested in the GoC itself. Furthermore, other international fishing undoubtedly occurred during the 1970s and before because international waters occurred beyond 12 nautical miles until the late 1970s. It is best to choose a starting point during which major change is not occurring. Notwithstanding such complications, simulations should still indicate the system tendencies of the effects of examined changes.

The key biological parameters (biomass, productivity, diet composition, etc) for each species or taxon in the model were estimated from primary research data, fishery data, or literature (Figure 57). The model features a fish diet composition matrix (Table 2) based on stomach content analyses from a number of previous CSIRO projects (Salini *et al.*, 1990; Brewer *et al.*, 1991; Brewer *et al.*, 1995; Haywood *et al.*, 1998; Salini *et al.*, 1998; Griffiths *et al.*, 2007; Griffiths *et al.*, In Press), but also from the extensive dietary work undertaken in the present project, which has been reported in previous milestone reports.

However, owing to the high diversity of the GoC fish assemblage and their generally low commercial value in Australia, region-specific information on the basic biology of many of these fish species is scanty. Maximum recorded length was often the only biological parameter available for most species and so we resorted to using the simplest empirical equations. Where the maximum size of a fish from a particular species was recorded in scientific surveys to be within 10% of the species' maximum recorded length we assumed this to be a reasonable proxy of L_∞ in the study region. We then used the empirical equation of Pauly (1980) to estimate natural mortality (M). Since the vast majority of species in the region are not fished commercially or recreationally, we assumed that M was equal to total mortality (Z), though it probably generally underestimates total mortality and thus production rate (P/B) estimates for some species due to bycatch mortality, thereby tending to make the model more conservative, or less responsive to IUU fishing impacts. We resisted the use of biological parameters from other systems outside Australia, as much as we could, since the GoC is a unique system that likely functions very differently from most other systems in various ways, being a large shallow tropical soft sediment gulf of the southern hemisphere with unique monsoonal patterns and a somewhat unique fauna.

For fish species having detailed biological information we used M or Z estimates provided in published studies from the region. Where only growth parameters were available, we estimated M as:

$$M = 1.60K \qquad \text{(as of Jensen, 1996),}$$

where K is the von Bertalanffy growth parameter. For some elasmobranch species where maximum age was known, we calculated M by:

$$M = -\ln(0.01)/\omega \qquad \text{(as of Hoenig, 1983),}$$

where ω is longevity in years. For fish species we knew, or suspected, were fished either as a target or caught incidentally in reasonable numbers as bycatch but fishing mortality (F) was not available, it was assumed (for lack of a better assumption) that the population would be fished at MSY. A precautionary proxy of fishing mortality at MSY can be assumed to be $F = 0.8M$ (Gabriel and Mace, 1999) or $Z = 1.8M$.

Due to a lack of information of the movement dynamics of species in northern Australia, we assumed that there was no net movement of any functional group in or out of the model, that is, that immigration equalled emigration. Therefore, all mortality and prey consumption occurred within the system.

For each functional group, we found great variance in the estimated P/B values. To obtain the single value for a particular group, we excluded outlying values and any values we did not have confidence in based on the quality of biological parameters (e.g. VBGF parameters or the mortality estimation method. For example, several Lutjanid species are represented in the GoC ecosystem and can vary markedly in size. Although we had reliable VBGF parameters for some species (e.g. Newman *et al.*, 1996; Newman, 2002; Marriott *et al.*, 2007), VBGF parameters or maximum age estimates were only available from regions in the Indian Ocean, where fish seem to attain significantly smaller or larger maximum lengths than in northern Australian waters. Using these estimates would bias M estimates, and thus P/B. Therefore, we took the average of the most reliable values, and each weighted by the contribution of a particular species to the functional group in terms of biomass.

An estimate of Q/B was made for each fish species by using the empirical equation of Palomares and Pauly (1998):

$$\log \text{Q/B} = 7.964 - 0.204 \log W\infty - 1.965T' + 0.083A + 0.532h + 0.398d$$

where W_∞ is weight (in grams) at age infinity, annual mean water temperature $T' = 1000/(°\text{C}+273)$, A is the aspect ratio of the caudal fin = $h2/s$, h=1 and d=0 for herbivores, h=0 and d=1 for detritivores, and h=0 and d=0 for carnivores.
For fish species having little biological data, we used maximum length recorded in surveys (assumed L_∞) to estimate W_∞ using a length-weight relationship either from the literature if available, or from about 30 years of CSIRO unpublished survey data. To provide an overall Q/B value for each functional group we averaged values using the method used for P/B values.

9.3.6 SOURCES OF FISHERIES INFORMATION

The fisheries included in the model were: the Illegal Foreign Fishery (IFF), Northern Prawn Fishery (banana and tiger prawn fisheries separated); the Queensland (Qld) and Northern Territory (NT) line (primarily trolling for Spanish mackerel, but some demersal longline and dropline), gillnet (primarily sharks and grey mackerel) and pot (mud crab) fisheries; developmental fish trawl fishery (primarily red snappers); the indigenous fishery; and the recreational fishery. Annual catch and effort data for each of the primary state and Commonwealth fisheries that operate in the GoC were obtained from logbooks, effort for the IFF was derived from aerial surveillance surveys (Salini *et al.*, 2007), and from scientific surveys in the case of the indigenous fishery (Coleman *et al.*, 2003). The total biomass of each species caught in each fishery in 1990 was summed at the functional group level, and expressed as a biomass (t km^{-2}) within the modelled area.

Assigning fishery catches and bycatch to functional groups was difficult for some fisheries. Fishers are not required to record their catches at the species level in logbooks and are often recorded as broad taxonomic groups. This was a particular problem for the main shark fisheries of interest, namely gillnet and line fisheries, since all shark catches in logbooks were recorded as “shark – unspecified”, which encompasses several functional groups. In order to break down the unspecified catch to its constituent species for these two fisheries, we assumed that the relative contribution of each species to the catch recorded in logbooks was the same as the Qld and NT gillnet and line catch documented by Salini *et al.* (2006).

Complete catch data for the fish trawl fishery in the GoC (which comprises the state fisheries of NT and Qld) was not available, due to confidentiality of catch data in fisheries/areas where less than five boats operate. We were able to obtain permission to use catch data from the NT fish trawl fishery, but not for the Qld fishery. Since the same operators work in both fisheries and there is no spatial difference in the trawl bycatch near reef areas in the GoC (Stobutzki *et al.*, 2003), we scaled up the catch rate of each species (in terms of kg per boat day) of the NT catch to the total effort in the Qld fishery. The catch estimates from the NT and QLD fisheries were then combined to represent the entire fish trawl fishery in the GoC.

The catch composition and biomass of each species caught in the illegal foreign fishery in northern Australia during 1990 was unknown. Because the illegal foreign fishery uses similar gear as the domestic gillnet and longline fisheries that target sharks, it was assumed that the species composition of the domestic catch would probably be representative of the foreign catch. The catch composition was therefore calculated using data collected by scientific observers from commercial gillnet and longline catches in the GoC and reported by Salini *et al.* (2006). Species-specific catch biomass from the illegal foreign component was estimated by using the catch in the Australian domestic gillnet in the GoC in 1990. The illegal foreign catch was assumed to be 10% of the Australian domestic catch in 1990, a period when the illegal foreign effort was believed to be significantly less than present day.

Estimates of discards from the banana prawn and tiger prawn trawl fisheries was available and compiled by Dell et al. (2009) for the construction of this model. The species composition and biomass of the discarded catch was not available for the gillnet, line, pot, fish trawl, and illegal foreign fisheries. It is well accepted that there is negligible discarding from the pot and line fisheries, so in the model we assumed there was no discarding. Scientific observers routinely collect quantitative catch data for the gillnet and fish trawl fisheries, however, this was not available due to confidentiality policies of state fisheries agencies. Since the bycatch for both fisheries can be significant, it was important to include even crude bycatch biomass estimates. Despite the fish trawl fishery utilising nets with a slightly larger mesh size than nets in the NPF, confidential data (QDPI&F observer data) show that the bycatch composition is remarkably similar to the NPF, for which bycatch data are voluminous. Therefore, we scaled the NPF bycatch biomass, in terms of total effort in the fishery, headrope length and average trawl time, to reflect the fish trawl fishery. Species composition and biomass of the discarded catch in the gillnet fishery was based on data collected by CSIRO scientists during two fishery dependent surveys in February-March 2005. These data were then scaled up to the total number of days fished in the fishery in 1990. These data were also considered to be representative of the discards from the illegal foreign fishery due to the similarity in fishing methods. We scaled the discarded catch biomass to be 10% of the domestic gillnet fishery, as was undertaken for estimating the IFF retained catch.

9.3.7 FITTING THE ECOSIM MODEL TO CPUE DATA

Using the fitting procedure in Ecosim version 6 was used to calibrate this 1990 Gulf of Carpentaria Ecopath model to time series of biomass (t), fishing mortality (*F*), catch (t) data for adult tiger prawns derived from a recent stock assessment for 1970-2006 (Dichmont *et al.* 2010) and nominal effort data (number of boat days per year) for ten fisheries for the period 1990-2006 in order to increase the reliability of predictions from the Ecosim scenarios. Annual biomass estimates for adult tiger prawns were scaled to the value in the first year of the time series, effectively reducing the dataset to standardised biomass. Annual adult tiger prawn catches were divided by the model region and expressed as t km^{-2} to be consistent with other Ecopath model biomass inputs. Although a stock assessment exists for banana prawns in northern Australia (Zhou *et al.*, 2008), there were concerns that predicted biomass from the models were highly uncertain, primarily due to difficulty in correcting for changes in fishing power and recent changes in targeting practices. Therefore, only nominal effort and catch were used for this group in the Ecosim model calibration process and well as effort data for state-regulated fisheries (line, gillnet, pot, recreational, charter and fish trawl), aboriginal fisheries and IUU fisheries.

Data were imported into Ecosim and the 15 most sensitive interactions between predators and prey were identified using a non-linear search procedure. The prey vulnerability rate (v, see Equation 4) for the most sensitive predator-prey interactions were then iteratively adjusted (from the default of 2) by 1% until the sums of squares error (SS) was minimised to produce the best model fit to the time series data. In searching for the best combination of vulnerability values, the time series data were linked to an estimated primary productivity trend, in the form of a forcing function, forced upon the "phytoplankton" group, which best improved the Ecosim model fit to the time series. A variance value of 50 was used for the model fitting, implying that the model should capture large 'spikes' in the biomass and catch data. However, such high variance values can result in model 'over-fitting', meaning that the model may simply be fitting to noise in the data rather than true trends. Hence, the model was restricted to fit only seven parameters (one for each functional group) using 10 spline points in the primary productivity forcing function, which reduced risk of model over-fitting.
Once the optimal vulnerability search was complete, vulnerability values were inspected to ensure they were ecologically realistic and adjusted slightly manually if required. Although some adjustments to vulnerabilities and feeding time parameters often resulted in better visual fits of the model to observed data, they often resulted in a poorer statistical fit (lower SS) or required unrealistic parameter values, hence they were disregarded. Other sources provide further detailed descriptions of fitting Ecosim models to time-series data (Christensen and Walters, 2004; Christensen *et al.*, 2005).

9.3.8 DEVELOPING ECOSPACE

Ecospace is the spatial and temporal module of the Ecopath with Ecosim software package (www.ecopath.org; Christensen et al., 2005). Ecospace is a dynamic, spatial model that incorporates all key elements of Ecosim (including different vulnerabilities and split pools presented above). The Ecospace model is structured on biomass pools, linked by trophic relationships (i.e. predator-prey), which migrates among the grids of cells of proposed MPAs. Movements of functional groups are driven by parameters such as foraging behaviour, avoidance of predation, and dispersal rates that are linked to a range of defined habitats preferred by each functional group. Robust default estimation for these parameters based on life histories is built into Ecospace (Walters et al.,1999; Christensen and Walters, 2004).

Ecospace (Walters *et al.* 1998) models the feeding interactions of functional groups in a spatially explicit way. A simple grid represents the study area, and it is divided into a number of habitat types. Each functional group is allocated to its appropriate habitat(s), where it must find enough food to eat, grow and reproduce, while providing biomass to its predators and to fisheries. Each cell hosts its own Ecosim simulation and cells are linked through symmetrical biomass flux in four directions; the rate of transfer is affected by habitat quality. Optimal and sub-optimal habitat can be distinguished using various parameters such as the availability of food, vulnerability to predation and immigration/emigration rate. By delimiting an area as a protected zone, and by defining which gear types are allowed to fish there and when, we can explore the effects of MPAs and test hypotheses regarding ecological function and the effect of fisheries (see Walters *et al.*, 1998; Pitcher et al., 2005).

The biomass of functional groups is initially distributed over the modeled region and biomasses and fluxes among cells are governed by dispersal rates related to food availability and predation rates in the cells (Christensen *et al*, 2005). Then, the ecosystem is divided into a 2-dimensional grid of cells and each cell is defined according to its habitat type and the habitat preferences of each functional group. In the case of fishing pressure and landings, Ecospace incorporates the time series of effort per fleet per year that were defined in the Ecopath model. It should be noted that these fleets can be varied independently. The spatial distribution of this fishing effort is then controlled by a ‘gravity’ model, which allocates effort to each cell proportional to the relative profitability of fishing in each cell.

The Gulf of Carpentaria (GoC) model was represented by a grid of 6,400 cells (80 x 80 cells). The Ecospace habitat base map (Figure 1) was built based on the broad definition of “habitats” – i.e. for this model, habitats were spatially defined areas that contain a major biological, fishery, or geomorphic benthic feature that is directly related to the NPF. The major fishing grounds identified in the GoC: (1) tiger/endevour North; (2) tiger/endevour South; (3) tiger/endevour Vanderlins East; (4) tiger/endevour: Monington north; (5) tiger/endevour: Monington east; (6)) tiger/endevour: Weipa; (7) Banana; while the biological habitats were the (8) untrawlable grounds (hard/reefs); (9) seagrass; (10) inshore and shallow waters; (11) offshore and deep waters. All these habitats and their percentage of cover in the Ecospace map are presented in Table 1.

Table 37. Descriptions of each habitat type considered and defined in the GoC Ecospace model.

#	Type	Habitat Name	Acronym	Description
1	Fisheries ground	Tiger/endeavour Groote north	grtn	Northern Groote green-mud hot spot
2	Fisheries ground	Tiger/endeavour Groote south	grts	Southern Groote hot spot
3	Fisheries ground	Tiger/endeavour: Vanderlins east	vane	South-central harder-grounds hotspots
4	Fisheries ground	Tiger/endeavour: Mornington north	mtnn	North-Mornington hotspot (sponges & heart urchins grounds)
5	Fisheries ground	Tiger/endeavour: Monington east	mtne	South-east riverine-tidal sedimentary basin
6	Fisheries ground	Tiger/endeavour: Weipa	weipa	Offshore NE Weipa
7	Fisheries ground	Banana	banana	East-south (including inshore Weipa) banana hotspots
8	Biological	Untrawlable ground	utg	Rough grounds Submerged coral reefs
9	Biological	Seagrass	sg	Shore and coastal/tidal seagrass beds
10	Geomorphic	Inshore/shallow	inshore	Depth <15m, shallow and backreef/utg
11	Geomorphic	Off-shore/Deep	offshore	Depths ~>45m. oceanic-like but muddy largely off-trawling grounds
12	Geomorphic	Offshore-2 (Boundary of the Model)	Offshore2	North-west Arafura out of model area

controlled by a ‘gravity’ model, which allocates effort to each cell proportional to the relative profitability of fishing in each cell.

The Gulf of Carpentaria (GoC) model was represented by a grid of 6,400 cells (80 x 80 cells). The Ecospace habitat base map (Figure 1) was built based on the broad definition of “habitats” – i.e. for this model, habitats were spatially defined areas that contain a major biological, fishery, or geomorphic benthic feature that is directly related to the NPF. The major fishing grounds identified in the GoC: (1) tiger/endevour North; (2) tiger/endevour South; (3) tiger/endevour Vanderlins East; (4) tiger/endevour: Monington north; (5) tiger/endevour: Monington east; (6)) tiger/endevour: Weipa; (7) Banana; while the biological habitats were the (8) untrawlable grounds (hard/reefs); (9) seagrass; (10) inshore and shallow waters; (11) offshore and deep waters. All these habitats and their percentage of cover in the Ecospace map are presented inTable 37.

9.3.9 HABITAT ASSIGNMENT

Once habitats were defined, the functional groups defined in the Ecopath model were assigned to their 'preferred' habitat. 'Preferred' here means that the group in question has a higher feeding rate in the habitat. The group's survival rate is also higher here because the predation rate is higher in non-preferred habitats. The habitat assignment was made by experts from CSIRO and based on data from surveys carried out in the waters of the GoC. The habitat assignment and base dispersal rates of the functional groups considered in the model is presented in details in Table 38.

9.3.10 SPATIAL REPRESENTATION OF FISHERIES

In the case of the spatial representation of fishing in Ecospace, the model uses the multiple fishing fleets and their instantaneous fishing mortality rates (F) included in the Ecopath model. In Ecospace, the F's are distributed using a simple 'gravity model', where the proportion of the total fishing effort allocated to each cell is assumed to be proportional to the sum over groups of the product of the biomass. The catchability and profitability of fishing the target groups are also considered and included (Christensen et al., 2005). The Ecospace model considered the 14 fishing fleet included in the Ecopath model and their preliminary spatial distribution within the spatial model (Table 38).

9.3.11 DISPERSAL RATES

Each of the groups and species considered in the Ecopath model have an aggregated biomass (B_i) and they are not assumed to move within the modeled area of the GoC (Fig. 3). In Ecospace, a fraction of the biomass (B') of each cell is always on the move, according to

$$B' = m \cdot Bi$$

with m having the dimension of length/time (i.e. km/year) i.e., a velocity or 'speed'. However, m is not a rate of directional migration, as occurs seasonally in numerous fish populations. Rather, m should be regarded as dispersal and seen as the rate (km $year^{-1}$) of which the organism would disperse from a given ecosystem as a result of random movements (Christensen et al., 2005). As for the absolute value of m to be used in the simulation we use a default value of 300 km $year^{-1}$ (recommended by Christensen *et al.,* 2005) for all groups with high/medium motion activity (fish groups) and we use a default value of 3 km $year^{-1}$ (Christensen *et al.,* 2005) for those groups with very low motion (sessile groups and non-living groups). The movement rates in the model were revised by experts from CSIRO. No migration or advection of functional group was considered in this model.

Table 38. Spatial allocation by each habitat of the various functional groups included in the Ecospace model of the Gulf of Carpentaria (grtn= Groote north, grts= Groote south, vane= Vanderlins east, mtnn= Mornington north, mtne= Mornington east, weipa= Weipa, bananan= Banana east, utg= Untrawlable grounds, sg= Seagrass, inshore= Inshore/shallow, offshore= Offshore/deep). It also includes the values of the annual dispersal rates used for each functional group.

Group \ Habitat #	**grtn**	**grts**	**vane**	**mtnn**	**mtne**	**weipa**	**banana**	**utg**	**sg**	**inshore**	**offshore**	**Dispersal rate**
Percentage area	**1.9%**	**1.5%**	**2.5%**	**0.6%**	**0.9%**	**0.3%**	**4.7%**	**7.2%**	**4.4%**	**8.7%**	**67.3%**	**(km/year)**
1. Dolphins	5.0%	5.0%	5.0%	5.0%	5.0%	5.0%	10.0%	10.0%	0.0%	10.0%	40.0%	300
2. Dugongs	5.0%	5.0%	5.0%	5.0%	5.0%	5.0%	5.0%	5.0%	50.0%	15.0%	5.0%	50
3. Turtles	2.6%	1.2%	1.4%	0.0%	1.0%	0.4%	11.1%	9.0%	15.2%	32.5%	25.5%	300
4. Sea snakes	2.2%	3.9%	5.0%	2.2%	4.7%	2.0%	5.0%	33.0%	10.0%	22.0%	10.0%	100
5. Sea birds	3.0%	3.0%	3.0%	3.0%	3.0%	5.0%	10.0%	5.0%	0.0%	5.0%	60.0%	300
6. Large sharks	1.7%	1.2%	1.9%	0.5%	0.7%	0.2%	6.2%	7.7%	4.2%	11.2%	64.6%	300
7. Small sharks	3.2%	1.7%	2.3%	0.4%	0.4%	0.5%	4.0%	6.5%	5.0%	11.1%	64.9%	300
8. Sawfishes	3.9%	3.1%	15.0%	0.4%	0.3%	0.2%	6.8%	19.3%	12.6%	24.3%	14.0%	100
9. Rays	2.7%	2.2%	6.5%	0.6%	0.2%	0.4%	3.1%	7.8%	1.6%	5.1%	69.7%	100
10. Pelagic carnivores Fish	2.1%	1.6%	2.9%	0.5%	2.2%	0.3%	6.7%	9.9%	3.2%	10.4%	60.3%	100
11. Pelagic invert feeders Fish	3.3%	1.2%	1.6%	0.2%	1.4%	0.4%	6.4%	8.8%	6.2%	14.2%	56.4%	50
12. Benthopelagic carnivores Fish	3.4%	2.2%	2.9%	0.5%	0.5%	0.6%	6.5%	7.9%	3.1%	10.6%	61.9%	100
13. Benthopelagic invert feeders Fish	2.9%	1.4%	2.8%	0.2%	0.3%	1.1%	7.3%	9.3%	4.9%	15.3%	54.5%	50
14. Benthic carnivores Fish	1.6%	1.0%	1.8%	0.6%	0.3%	0.3%	2.2%	4.5%	1.6%	4.6%	81.5%	150
15. Benthic invert feeders Fish	1.3%	0.9%	1.3%	0.3%	0.6%	0.3%	10.7%	6.8%	6.7%	21.1%	50.1%	50
16. Red snappers Fish	0.9%	0.4%	2.0%	0.3%	0.4%	0.1%	3.5%	9.5%	2.1%	6.1%	74.7%	100
17. Reef assoc. carnivores Fish	4.1%	1.5%	2.0%	0.5%	0.3%	0.6%	4.9%	8.6%	3.7%	10.2%	63.6%	100
18. Reef assoc. invert feeders Fish	2.6%	1.1%	1.8%	0.4%	0.1%	0.4%	2.0%	4.5%	1.1%	3.7%	82.2%	50
19. Reef assoc. herbivores Fish	1.2%	0.7%	2.2%	0.3%	0.4%	0.3%	12.3%	11.3%	4.0%	16.5%	51.1%	50
20. Detritivores Fish	6.4%	0.5%	1.0%	0.0%	0.5%	7.2%	15.6%	9.1%	10.9%	31.0%	17.7%	20
21. Cephalopods	1.3%	0.7%	2.0%	0.4%	0.8%	0.2%	4.1%	7.6%	3.8%	11.8%	67.3%	30
22. Stomatopods	10.0%	5.0%	5.0%	5.0%	5.0%	2.0%	10.0%	8.6%	5.6%	17.0%	26.8%	20
23 Banana prawn juv	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	1.0%	1.0%	50.0%	48.0%	0.0%	1
24. Banana prawn adult	3.8%	0.9%	0.4%	0.0%	0.4%	0.7%	65.0%	3.0%	11.0%	10.0%	4.7%	1
25. Tiger prawn juv	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	5.0%	65.0%	30.0%	0.0%	10
26. Tiger prawn adult	12.0%	10.0%	11.4%	12.0%	12.0%	1.0%	3.0%	15.5%	6.1%	5.0%	12.0%	20
27. Other commercial prawns	12.0%	10.0%	11.0%	12.0%	12.0%	1.0%	5.0%	15.0%	5.0%	7.0%	10.0%	20
28. Thallasinid prawns	0.9%	0.4%	2.3%	0.4%	1.4%	0.4%	9.7%	5.3%	4.8%	15.8%	58.7%	20
29. Other non-commercial prawns	4.6%	2.7%	6.7%	9.4%	0.7%	0.7%	4.7%	11.7%	2.7%	8.5%	47.7%	20
30. Lobsters	1.4%	1.0%	10.0%	1.6%	1.4%	0.4%	6.3%	35.0%	3.0%	10.0%	30.0%	10
31. Mud crab	5.0%	4.0%	5.0%	10.0%	6.0%	1.0%	15.0%	10.0%	7.0%	15.0%	22.0%	10
32. Sand crab and other large crabs	4.1%	1.7%	5.6%	2.4%	1.1%	0.9%	8.5%	11.0%	3.7%	12.7%	48.2%	1
33. Large gastropod carnivore	1.3%	1.0%	2.0%	0.6%	1.4%	0.4%	7.2%	8.7%	3.1%	10.5%	63.8%	10
34. Holothurians	3.8%	3.1%	3.6%	1.1%	2.8%	0.1%	4.6%	12.8%	8.0%	19.0%	41.1%	1
35. Spatangoids	1.5%	6.0%	8.2%	0.4%	0.3%	0.0%	2.4%	12.9%	5.5%	12.1%	50.6%	2
36. Echinoids	2.4%	5.0%	5.4%	5.8%	0.4%	0.1%	2.8%	12.2%	3.6%	9.4%	52.9%	2
37. Ophiuroids	1.4%	0.4%	0.7%	0.2%	3.0%	0.3%	9.7%	6.3%	4.5%	14.0%	59.6%	2
38. Asteriods	5.0%	5.0%	5.0%	5.0%	5.0%	2.0%	10.0%	7.0%	3.0%	5.0%	48.0%	2
39. Sessile epibenthos	0.5%	0.8%	0.5%	8.0%	0.7%	0.5%	3.0%	65.0%	3.0%	8.0%	10.0%	0
40. Bivalves	4.6%	3.0%	2.8%	1.0%	0.7%	0.3%	6.4%	14.1%	11.5%	24.7%	31.1%	2
41. Small crustaceans	5.0%	5.0%	5.0%	5.0%	5.0%	3.0%	12.0%	20.0%	5.0%	15.0%	20.0%	1
42. Annelids detrit / carn	2.4%	2.3%	4.1%	1.1%	0.9%	0.2%	3.8%	9.3%	2.1%	6.8%	66.8%	2
43. Small gastropod omni / Small gastropod carn	5.0%	5.0%	8.0%	2.0%	2.0%	1.0%	9.0%	11.2%	3.8%	12.0%	41.0%	2
44. Infaunal detrit / carn	8.8%	5.0%	5.0%	2.0%	1.9%	1.0%	10.4%	6.5%	2.6%	11.0%	45.8%	2
45. Zooplankton	2.1%	1.8%	2.6%	2.8%	0.7%	1.6%	17.1%	11.5%	6.2%	27.1%	26.5%	10
46. Microbial heterotrophs	1.9%	1.5%	2.5%	0.6%	0.9%	0.3%	4.7%	7.2%	4.4%	8.7%	67.3%	50
47. Foraminifera	1.9%	1.5%	2.5%	0.6%	0.9%	0.3%	4.7%	7.2%	4.4%	8.7%	67.3%	50
48. Phytoplankton	1.9%	1.5%	2.5%	0.6%	0.9%	0.3%	4.7%	7.2%	4.4%	8.7%	67.3%	10
49. Microphytobenthos	1.9%	1.5%	2.5%	0.6%	0.9%	0.3%	4.7%	7.2%	4.4%	8.7%	67.3%	5
50. Seagrass	1.0%	1.0%	2.0%	1.0%	1.0%	1.0%	2.0%	2.0%	75.0%	14.0%	0.0%	0
51. Macroalgae	0.8%	0.2%	0.4%	0.1%	1.1%	0.3%	6.8%	22.1%	9.4%	30.7%	28.0%	0

Table 39. List of the 53 functional groups of the GoC benthic model and its basic input parameters.

Group name	Trophic level	Habitat area (fraction)	Biomass in habitat area (t/km2)	Z (/year)	Production / biomass (/year)	Consumption / biomass (/year)	Ecotrophic efficiency	Production / consumption
1. Dolphins	4.4	1	0.001		0.1	41.07	0.155	0.002
2. Dugongs	2.4	1	0.001		0.08	36.5	0.324	0.002
3. Turtles	3.4	1	0.008		0.192	3.5	0.138	0.055
4. Sea snakes	4.7	1	0.001		0.46	6.3	0.322	0.073
5. Sea birds	3.7	1	0.003		0.131	45.8	0.103	0.003
6. Large sharks	4.8	1	0.008		0.33	3.468	0.313	0.095
7. Small sharks	4.5	1	0.011		0.568	7.158	0.876	0.079
8. Sawfishes	4.3	1	0.003		0.29	2.66	0.506	0.109
9. Rays	3.9	1	0.163		0.565	4.561	0.195	0.124
10. Pelagic carnivores Fish	4.2	1	0.164		0.79	8.76	0.227	0.09
11. Pelagic invert feeders Fish	3	1	3.71		0.845	13.87	0.793	0.061
12. Benthopelagic carnivores Fish	4	1	0.35		1.792	10.788	0.338	0.166
13. Benthopelagic invert feeders Fish	3.4	1	0.995		2.62	10.927	0.815	0.24
14. Benthic carnivores Fish	4.1	1	0.225		1.6	7.567	0.529	0.211
15. Benthic invert feeders Fish	3.5	1	0.8		2.008	9.732	0.909	0.206
16. Red snappers Fish	4.4	1	0.132		1.505	5.667	0.522	0.266
17. Reef assoc. carnivores Fish	4.3	0.2	0.125		1.2	10.063	0.829	0.119
18. Reef assoc. invert feeders Fish	3.8	0.2	0.144		2.105	14.489	0.9	0.145
19. Reef assoc. herbivores Fish	3	0.2	0.179		1.3	31.413	0.957	0.041
20. Detritivores Fish	2	0.11	0.124		2.18	17.44	0.811	0.125
21. Cephalopods	4.1	1	0.17		3.4	17.338	0.59	0.196
22. Stomatopods	3.5	1	0.1		4.25	14.5	0.686	0.293
23 Banana prawn juv	3.2	0.24	0.004	3.42		55.636	0.182	0.061
24. Banana prawn adult	3.3	1	0.009	3.2		19.2	0.679	0.167
25. Tiger prawn juv	3.3	0.18	0.011	3.45		55.686	0.177	0.062
26. Tiger prawn adult	3.3	1	0.02	3.2		19.2	0.87	0.167
27. Other commercial prawns	3.3	1	0.18		3.2	19.2	0.605	0.167
28. Thallasinid prawns	3.1	1	0.963		4.65	19.2	0.633	0.242
29. Other non-commercial prawns	3.1	1	0.58		3.44	19.2	0.638	0.179
30. Lobsters	3	1	0.012		0.9	7.4	0.313	0.122
31. Mud crab	3	1	0.003		2.8	10.95	0.055	0.256
32. Sand crab and other large crabs	3.2	1	0.086		2.65	14.56	0.836	0.182
33. Large gastropod carnivore	2.9	1	0.08		0.72	3.755	0.74	0.192
34. Holothurians	2	1	0.055		0.6	2.077	0.3	0.289
35. Spatangoids	2	1	0.013		1.5	5	0.209	0.3
36. Echinoids	2.7	1	0.175		1.65	10.95	0.811	0.151
37. Ophiuroids	2.1	1			1.6	13.992	0.95	0.114
38. Asteriods	2.8	1	0.051		0.46	3.24	0.186	0.142
39. Sessile epibenthos	2.5	1	4.985		0.189	30.213	0.767	0.006
40. Bivalves	2	1	4.5		2.4	9.5	0.895	0.253
41. Small crustaceans	2.4	1	4.72		5.4	47	0.8	0.115
42. Annelids detrit / carn	2.5	1	4.81		4.6	15.3	0.925	0.301
43. Small gastropod omni / Small gastropod c	2.6	1	2.3		3.78	15.515	0.924	0.244
44. Infaunal detrit / carn	2.4	1	5.5		3.8	27.4	0.852	0.139
45. Zooplankton	2.1	1	12.6		17.3	173	0.807	0.1
46. Microbial heterotrophs	2	1			100	215	0.95	0.465
47. Foraminifera	2.1	1			12.5	25	0.95	0.5
48. Phytoplankton	1	1	17		118		0.848	
49. Microphytobenthos	1	1	3.241		706.496		0.069	
50. Seagrass	1	0.31	0.626		8.317		0.968	
51. Macroalgae	1	1	18		8.317		0.661	
52. Discards	1	0.75	0.123				0.465	
53. Detritus	1	1	285				0.349	

Table 40. Final diet matrix used in the NPF GoC Ecopath model.

FG #	Prey \ predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
1	Dolphins	0	0	0	0	0	0.0007966	0	0	0	0	0	0	0	0	0	0	0
2	Dugongs	0	0	0	0	0	0.0004799	0	0	0	0	0	0	0	0	0	0	0
3	Turtles	0	0	0	0	0	0.003839	0.0008281	0	0	0	0	0	0	0	0	0	0
4	Sea snakes	0	0	0	0	0	7.97E-05	8.28E-05	0	0	7.70E-05	0	0	0	0	0	0	9.15E-05
5	Sea birds	0	0	0	0	0	0.000643	0.0002465	0	0	0	0	0	0	0	0	0	0
6	Large sharks	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
7	Small sharks	0	0	0	0	0	0.0432832	0.0148859	0	0	0	0	0	0	0	0	0	0
8	Sawfishes	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
9	Rays	0	0	0	0	0	0.0191528	0.0049292	0	0	0	0	0	0	0	0	0	0.052728
10	Pelagic carnivores Fish	0.02	0	0	0	0	0.0230342	0.0216886	0	0	0	0	0.004151	0	0	0	0	0.0198974
11	Pelagic invert feeders Fish	0.208	0	0.0481154	0.0240481	0.553844	0.0489477	0.0828111	0.1998002	0.0200092	0.5078745	0	0.1037752	0	0.0978159	0	0.0518883	0.1452508
12	Benthopelagic carnivores Fish	0.001	0	0	0	0	0.0412696	0.0374622	0	0	0.0609849	0	0	0	0.0322692	0	0.026942	0.0477537
13	Benthopelagic invert feeders Fish	0.1	0	0.0192461	0.0821643	0.0620825	0.0527867	0.085713	0.2877123	0.0200092	0.076981	0	0.1471708	0	0.1055295	0	0.1846552	0.052728
14	Benthic carnivores Fish	0.05	0	0	0.3647294	0	0.0575855	0.0532357	0	0.0110146	0.0229923	0	0.0197173	0	0	0	0.0548818	0.0338255
15	Benthic invert feeders Fish	0.05	0	0	0.3577154	0.0510856	0.1915281	0.093684	0.2877123	0.1131495	0.0309897	0	0.1577384	0	0.2341749	0	0.2669352	0.1183893
16	Red snappers Fish	0	0	0	0	0	0.0374306	0.0374622	0	0	0	0	0.0103775	0	0.0282355	0	0	0.0248717
17	Reef assoc. carnivores Fish	0	0	0	0	0	0.1343662	0.0246462	0	0	0.003999	0	0.0010364	0	0.0020101	0	0.0074982	0
18	Reef assoc. invert feeders Fish	0	0	0	0.0801603	0	0.023994	0.0236603	0	0	0.0049988	0	0.0007255	0	0.0050252	0	0.0189954	0.0268615
19	Reef assoc. herbivores Fish	0	0	0	0	0	0.0374306	0.0325329	0	0	0.0049988	0	0.0031196	0	0	0	0	0.0269321
20	Detritivores Fish	0.166	0	0	0	0	0.0088294	0.0222401	0	0.0016007	0.0022992	0	0.0005189	0	0.0020101	0	0	0.0017888
21	Cephalopods	0.4	0	0.0481154	0	0	0.0719819	0.1114006	0	0.0070093	0.0359911	0.0006001	0.012437	0.0001002	0.0413449	0.0080215	0.033927	0.0278563
22	Stomatopods	0	0	0	0	0	0.0095976	0.0492923	0	0.0410543	1.70E-05	0.0008001	0.0103775	0.0010022	0.0363028	0.0050134	0.0269935	0.0238769
23	Banana prawn juv	0	0	0	0	0	0	0	0	0	0	0	0	5.00E-05	0	0	0	0
24	Banana prawn adult	0	0	0	0	0	0.0067183	0.0049292	0.041958	0.0020026	0.0007697	0	0.0005085	0.0001002	0.0020168	0.0005013	0	0
25	Tiger prawn juv	0	0	0	0	0	0	0	0	0	0	0	0	5.00E-05	0	8.00E-06	0	0
26	Tiger prawn adult	0	0	0	0	0	0.0047988	0.0078868	0	0.0070093	0.0006798	0	0.0031133	0.0006614	0.0020168	0.0010027	0.0008482	0.0008058
27	Other commercial prawns	0	0	0	0	0	0.0019195	0.0187311	0	0.0210278	7.20E-05	0	0.012453	0.0030066	0.0030252	0.0270725	0.0008482	0.0049743
28	Thallasinid prawns	0	0	0	0	0	0.0001152	0.0008281	0	0.0851125	0	0	0.0622652	0.1362987	0.0100841	0.0782094	0.0169635	0.0009153
29	Other non-commercial prawns	0	0	0	0	0	0.0047988	0.0325329	0	0.2283016	0.002999	0.0080013	0.0300948	0.005011	0.0332776	0.0401074	0.0189592	0.014923
30	Lobsters	0	0	0	0	0	0	0.0029575	0	0	0	0	0	0	0.000595	0	0	0.0029846
31	Mud crab	0	0	0.0004812	0	0	0	0	0	0	0	0	0	0	0	0	0	0
32	Sand crab and other large crabs	0	0	0.0098946	0	0	0.0002872	0.0029506	0	0	0.0003001	0	0.01524	0	0.0512808	0	0	0.0910555
33	Large gastropod carnivore	0	0	0	0	0	0	0	0	0	0	0	0	0.0004309	0	0.0010027	0	0.0917857
34	Holothurians	0	0	0	0	0	0	0	0	0.003004	0	0	0	0	0	0.0008122	0	0
35	Spatangoids	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
36	Echinoids	0	0	0	0	0	0	0	0	0.0001903	0	0	0	0.0007316	0.0009177	0.0050134	0.0029936	0.000189
37	Ophiuroids	0	0	0	0	0	0	0	0	0.0008511	0	0	0.0051888	0.0120264	0.000595	0.1133034	0.000449	0.000189
38	Asteriods	0	0	0	0	0	0	0	0	0	0	0	0	0.0002906	0.0005748	1.30E-05	0	0
39	Sessile epibenthos	0	0.25	0.292969	0	0	0	0	0	0.0001702	0	0	0	0.0030066	0.0010084	0.0020054	0.0006187	0.0003582
40	Bivalves	0	0	0	0	0	0	0	0.007992	0.003004	0	0.0050008	0.0072643	0.0280615	0.0231935	0.0701879	0.0039914	0.0179076
41	Small crustaceans	0	0	0	0	0	0	0.0561932	0	0.3554696	0.005998	0.1230197	0.270504	0.4068916	0.1939733	0.2085584	0.107768	0.0775998
42	Annelids detrit / carn	0	0	0	0	0	0	0	0	0.0110146	0	0.0020003	0.0031133	0.0240527	0.0272271	0.3669825	0.0019957	0.0002985
43	Small gastropod omni / carn	0	0	0	0	0	0	0	0	0.0020026	0	0	0.001017	0.0140308	0.0050421	0.0060161	0.0008482	0.0009053
44	Infaunal detrit / carn	0	0	0	0	0	0	0	0	0.000791	0	0	0.0002491	0.0004009	0.0211766	0.0250671	0	7.36E-05
45	Zooplankton	0	0	0.4822229	0	0	0	0	0	0.0001402	0.005998	0.6180989	0.0197173	0.3537752	0.0292439	0.0310832	0	8.46E-05
46	Microbial heterotrophs	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
47	Foraminifera	0	0	0	0	0	0	0	0	0	0	0.0004401	0	0	0	0	0	0
48	Phytoplankton	0	0	0	0	0	0	0	0	0	0	0.075012	0	0	0	0	0	0
49	Microphytobenthos	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
50	Seagrass	0	0.73	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
51	Macroalgae	0	0.02	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
52	Discarded Bycatch	0.005	0	0	0	0.1499878	0.0067183	0.0078868	0.007992	0.0110146	0	0	0	0	0	0	0	0
53	Detritus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
54	Import	0	0	0.0989555	0.0911824	0.183	0.1673694	0.1683023	0.1668332	0.0550473	0.2309801	0.1670267	0.0981266	0.0100209	0.0100335	0.0100185	0.171	0.0920984
	Sum	1	1	1	0.9999999	0.9999999	0.9997816	1	1	0.9999999	0.9999999	1	1	1	1	1	1	0.9999999

Table 39 (cont.): Final diet matrix used in the NPF GoC Ecopath model.

FG #	Prey \ predator	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40	4
1	Dolphins	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
2	Dugongs	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
3	Turtles	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
4	Sea snakes	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
5	Sea birds	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
6	Large sharks	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
7	Small sharks	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
8	Sawfishes	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
9	Rays	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
10	Pelagic carnivores Fish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
11	Pelagic invert feeders Fish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
12	Benthopelagic carnivores Fish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
13	Benthopelagic invert feeders Fish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
14	Benthic carnivores Fish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
15	Benthic invert feeders Fish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
16	Red snappers Fish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
17	Reef assoc. carnivores Fish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
18	Reef assoc. invert feeders Fish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
19	Reef assoc. herbivores Fish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
20	Detritivores Fish	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
21	Cephalopods	0	0	0	0	0	0	0.0390985	0	0	0	0	0	0	0	0	0	0	
22	Stomatopods	0.003	0	0	0	0	0	0.0020051	0	0	0	0	0	0	0	0	0	0	
23	Banana prawn juv	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
24	Banana prawn adult	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
25	Tiger prawn juv	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
26	Tiger prawn adult	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
27	Other commercial prawns	0	0	0	0	0	0	0.0020051	0.0131515	0	0	0	0	0	0	0	0	0	
28	Thallasinid prawns	0	0	0	0	0	0	0.0020051	0.0070816	0	0	0	0	0	0	0	0	0	
29	Other non-commercial prawns	0	0	0	0	0	0	0.0020051	0.0131515	0	0	0	0	0	0	0	0	0	
30	Lobsters	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
31	Mud crab	0	0	0	0	0	0	7.72E-05	0	0	0	0	0	0	0	0	0	0	
32	Sand crab and other large crabs	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
33	Large gastropod carnivore	0	0	0	0	0	0	0.0080202	0	0.005005	0	0	0	0	0	0	0	0	
34	Holothurians	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.005	0	0	
35	Spatangoids	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
36	Echinoids	0	0	0	0	0	0	0.1824599	0	0.04004	0.017982	0	0	0	0	0.02	0	0	
37	Ophiuroids	0.074	0.0002497	0.1541079	0.1119832	0.01	0.01	0	0	0	0	0	0	0.01	0	0	0	0	
38	Asteriods	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
39	Sessile epibenthos	0	0.0249735	0	0	0	0	0	0	0	0	0	0	0.242	0	0.03	0	0	
40	Bivalves	0.186	0.2497353	0.2731912	0.2259661	0.18	0.135	0.0471188	0.2063766	0.3803804	0	0	0	0	0	0.09	0	0	
41	Small crustaceans	0.225	0.1668232	0.1661163	0.1909714	0.09	0.105	0.0020051	0.1345494	0.2502503	0.1148851	0	0	0.007	0	0.02	0.0079995	0	
42	Annelids detrit / carn	0.246	0.0419555	0.0440308	0.159976	0.18	0.16	0.0020051	0	0.1301301	0.3006993	0	0	0.19	0	0.07	0	0	0.00
43	Small gastropod omni / carn	0.192	0.305676	0.3332333	0.2489626	0.21	0.24	0.1453664	0.1446659	0.0740741	0.1678322	0	0	0	0	0.26	0	0	
44	Infaunal detrit / carn	0	0.0008091	0.0003002	0.00015	0.05	0.03	0	0	0.039039	0.026973	0	0	0.01	0	0	0	0	
45	Zooplankton	0	0	0	0	0.07	0.09	0.0020051	0	0	0	0	0	0	0.011	0	0.1729901	0	0.37
46	Microbial heterotrophs	0	0	0	0	0.02	0.03	0	0	0	0	0	0	0	0.11	0.06	0.0979944	0	
47	Foraminifera	0.055	0.0079915	0.0090063	0.026996	0.03	0.032	0	0	0	0	0	0	0	0	0	0.17499	0	
48	Phytoplankton	0.019	0	0	0.0029996	0.04	0.01	0	0	0	0	0	0	0	0.055	0	0.3109823	0.16	0.2
49	Microphytobenthos	0	0	0	0	0.053	0.09	0	0	0	0	0.1	0.1	0	0	0.07	5.70E-05	0.12	
50	Seagrass	0	0.0349629	0	0	0	0	0.0771946	0	0	0	0	0	0	0	0	0	0	
51	Macroalgae	0	0	0	0	0	0	0.2446165	0.0485592	0.019019	0	0.1	0	0.324	0	0	0	0	0.36
52	Discarded Bycatch	0	0	0	0	0	0	0	0.0131515	0.01001	0	0	0	0	0	0	0	0	
53	Detritus	0	0	0.020014	0.0239964	0.06	0.06	0.0120303	0.0849786	0.0520521	0.3716284	0.8	0.9	0.217	0.824	0.375	0.2349866	0.72	
54	Import	0	0.1668232	0	0.0079988	0.007	0.008	0.2299823	0.3343343	0	0	0	0	0	0	0	0	0	
	Sum	1	1	1	1	1	0.9999999	1	0.9999999	1	1	1	1	1	1	1	0.9999999	1	

9.3.12 SIMULATION SCENARIO SETTINGS.

Using the balanced and calibrated ecosystem model for the GoC, we first explored a *base case,* where we simply let the Ecosim model simulate the historical dynamics of fishing from 1970 to 2009, using largely the observed catch and effort time series of the tiger prawn fishery in the NPF. We then used the models to evaluate the latest 5 years of fishing (2005-2010), to assess the ecosystem consequences of the substantial reductions of the NPF's fishing effort and fleet size – i.e. ~30% annually (Dichmont et al 2008). Observed catch or effort data were unavailable for 2010 so we used the results and recommendations of latest bioeconomic stock assessment for the tiger-endeavour fishery (Dichmont et al; 2010). For these scenarios we assessed the relative changes in biomass (i.e. biomass at end/start) of the various functional groups. We also assessed the historical biodiversity consequences of trawling on the predicted changes of the mean trophic level (TL) of the ecosystem in relation to the changes in the commercial catches, as the industry develops and adapt to changes. Lastly, we used Ecospace to evaluate the consequences to fishing and biodiversity after the hypothetical implementation of MPAs.

9.4 RESULTS

9.4.1 THE FOOD WEB

The basic parameters used are consistent with expert knowledge of the food web in the GoC ecosystem. The Figure 56 shows the trophic flows between each functional group in the GoC model indicating prey resource use (connecting lines from top to bottom), their relative biomass (size of the grey circles) and their position along their trophic level (vertical axis). As expected, the biomass of the primary producers and primary consumer groups represent that largest biomass, while the biomass of the remaining functional groups generally decreased with increasing trophic level (Figure 56). The intermediate trophic levels were largely represented by invertebrate groups, including commercially important prawns (Figure 56).

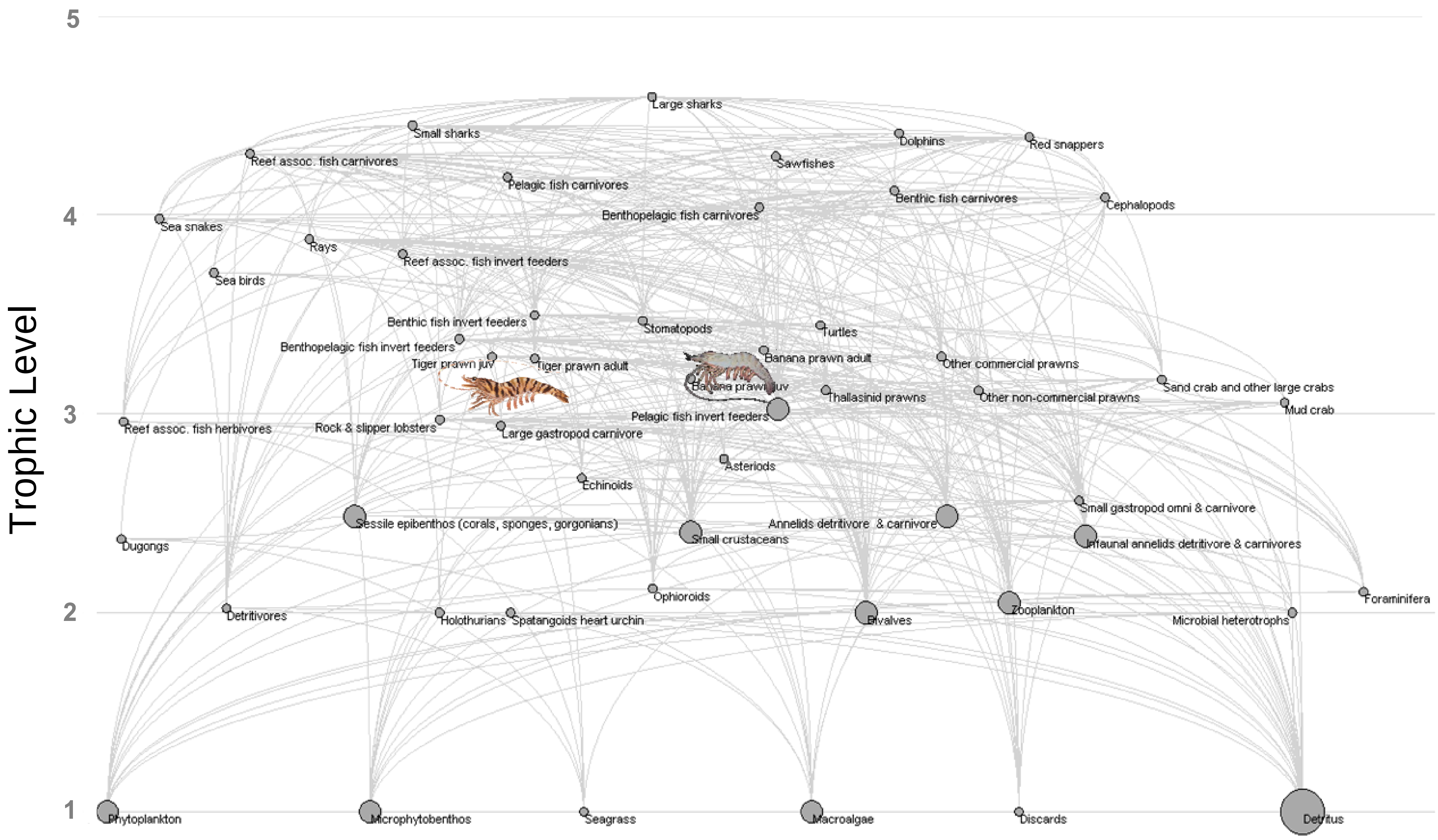

Figure 56. Trophic flow diagram of the benthic ecosystem of the Gulf of Carpentaria in relation to their trophic level (TL).

Using the Keystone Index (KI) of Ecopath's network analyses in relation to the group's relative trophic impact, we found that the adult tiger prawn is the group that has the most trophic influences (up and down the food web) in the modelled GoC ecosystem (Figure 57). This analysis also showed that, as expected, predatory large sharks and benthic fishes are also of high trophic impacts, and were followed by small gastropods, zooplankton and small crustaceans (Figure 57). This highlight that in addition to their commercial value, tiger prawns play also a keystone role in the ecosystem, both as consumer and as prey and their effct is disproportionate to their abundance in the ecosystem (Powers et al. 1996, Libralato et al. 2006).

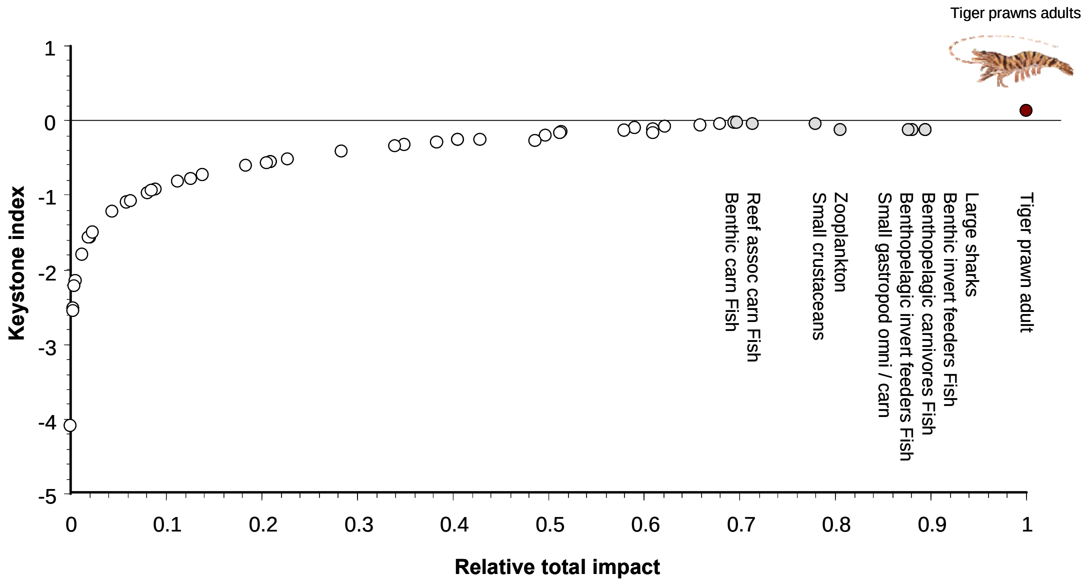

Figure 57. Keystoneness Index (KI) for all consumer functional groups of the marine trophic web of the Gulf of Carpentaria. The keystoneness index (y axis) is reported against overall relative effect (x axis) relative to the maximum effect measured in the trophic web (scale between 0 and 1). Labels are provided for the top-10 functional groups.

9.4.2 ECOSIM MODEL FITTING & CALIBRATION

The Ecosim model performed well in reproducing the 41 years of standardised biomass data for adult tiger prawns (Figure 58). However, the model did not capture well the variation in the data for the first three years of the biomass time series, which is often due to the Ecosim model having a difficulty of including high values in the forcing function that will allow the capture of the initial values, but also values later in the time series. This poor start-up fitting can also be due to uncertainty in these historic values predicted for the fishing effort, that were derived from the current stock assessment models (Dichmont et al. 2010). The Ecosim model in general has a better fit with catches values that were instead derived from the observed industry and management logbook system.

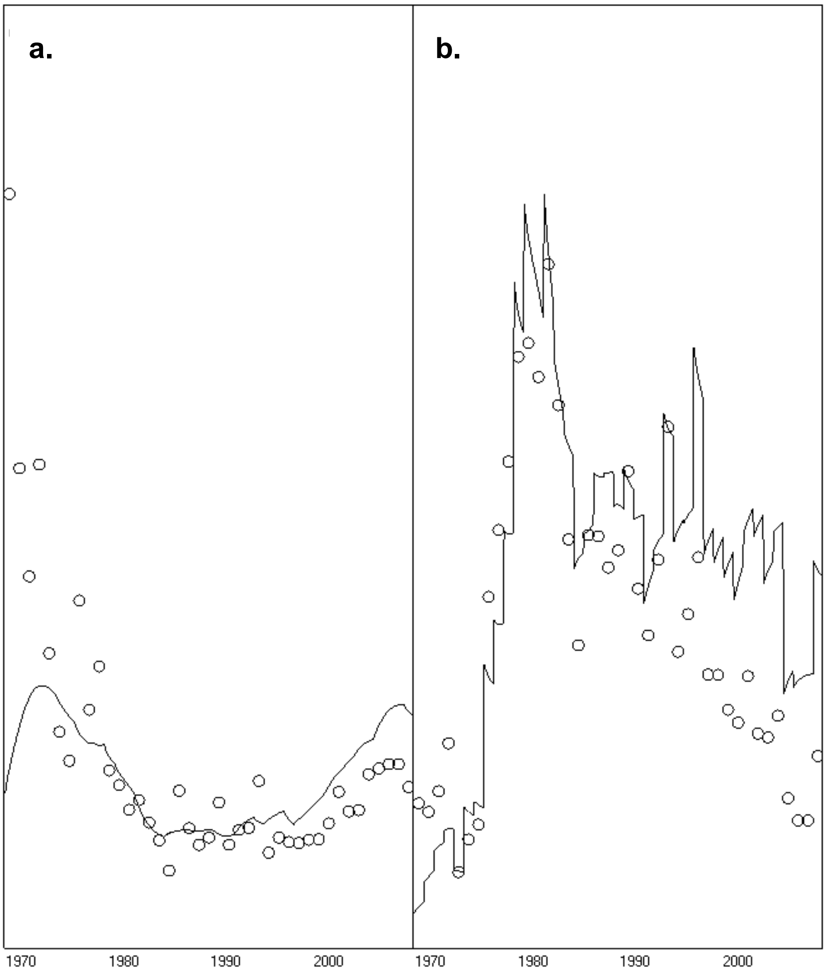

Figure 58. Fits of the NPF GoC Ecosim model to (a.) standardised relative biomass index and (b.) catch data for tiger prawns between 1970 and 2010. Circles are the observed values while the line are fitted by the model to the respective time series.

9.4.3 ASSESSING HISTORICAL AND RECENT EFFECTS OF THE NPF ON THE BENTHIC ECOSYSTEM

The NPF has experience a classical development of most fisheries worldwide, consisting of rapid growth and increase of fishing effort, and as catches decline due to overexploitation, reductions and adjustment to overall fishing practices are undertaken to maintain the fishery. Figure 4 depicts annual changes in fishing effort (boat/days) for 1970-2010. Most of fishing effort growth occurred in the 1970-80 period, with some variable changes in the 80s and 9s, followed by drastic and continual decreases from 2000 (Figure 59).

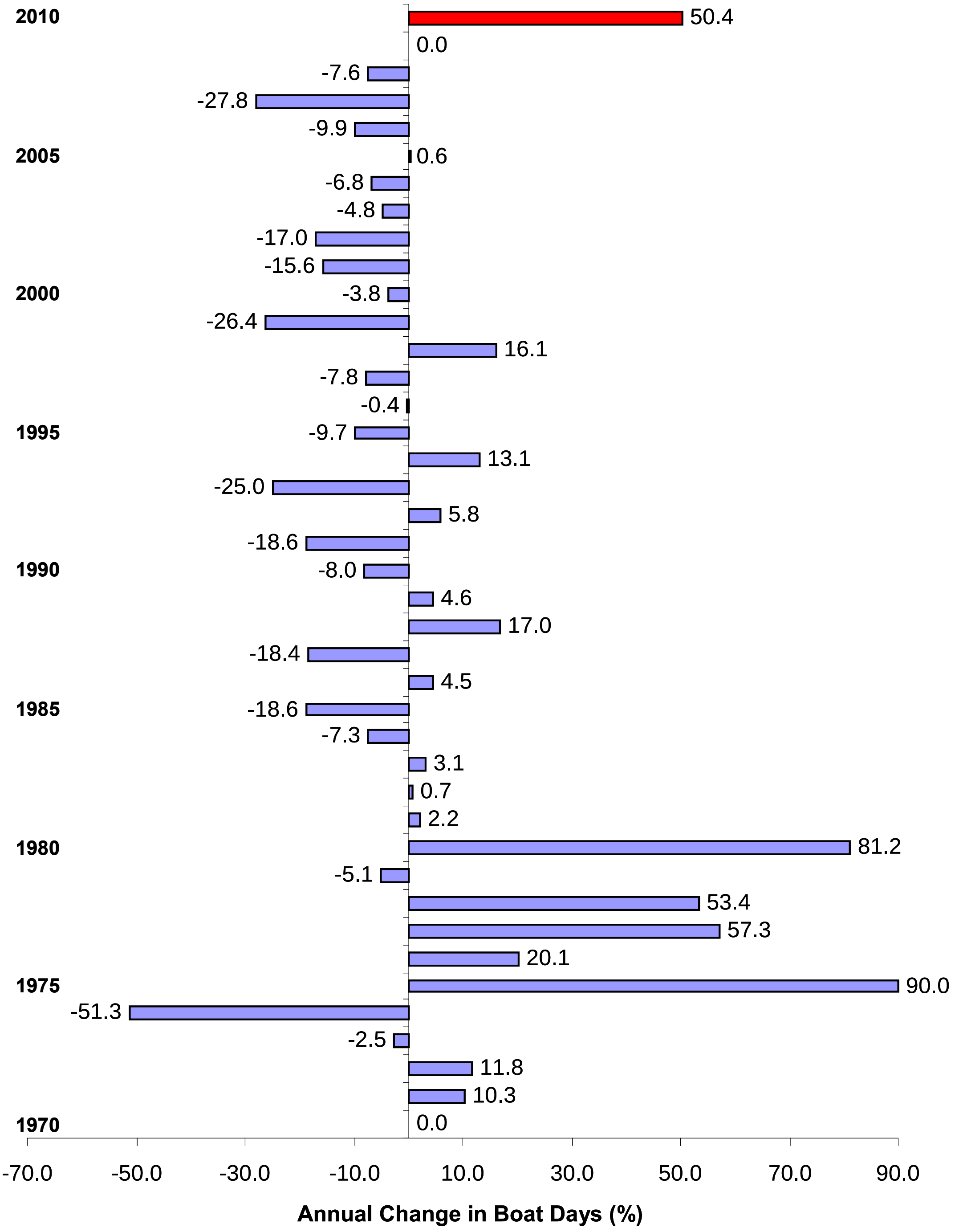

Figure 59. Annual relative changes (%) in fishing effort for the NPF (1970-2010)

Figure 60 depicts the predicted overall changes in the relative biomass of all 53 functional groups after 41 years of trawling in the GoC. It is clear that the historical trawling has a range of positive and negative effects in the various groups (Figure 60). Only the discarded bycatch group showed a large increase in relative biomass (>200%), while most positive and negative changes for the other groups ranged from less than 1% up to 30% (Figure 60). Overall, the most negatively affected groups were small sharks, banana prawns, mud crabs and echinoids (starfishes and urchins). As can be expected in an expanding fishery, the discarded bycatch increased substantially as did tiger prawns, sand crabs and large shark groups (Figure 60). All increasing groups were benthic feeders and/or groups known to consume discarded bycatch. The increase of tiger prawn is likely to be related to the drastic fishing effort reductions and probably less influenced by a trophic-related effect (Figure 60). The Ecosim model predicted that the tiger prawn population is in a rebuilding trajectory, largely explained by the active year to year management of the fishery.

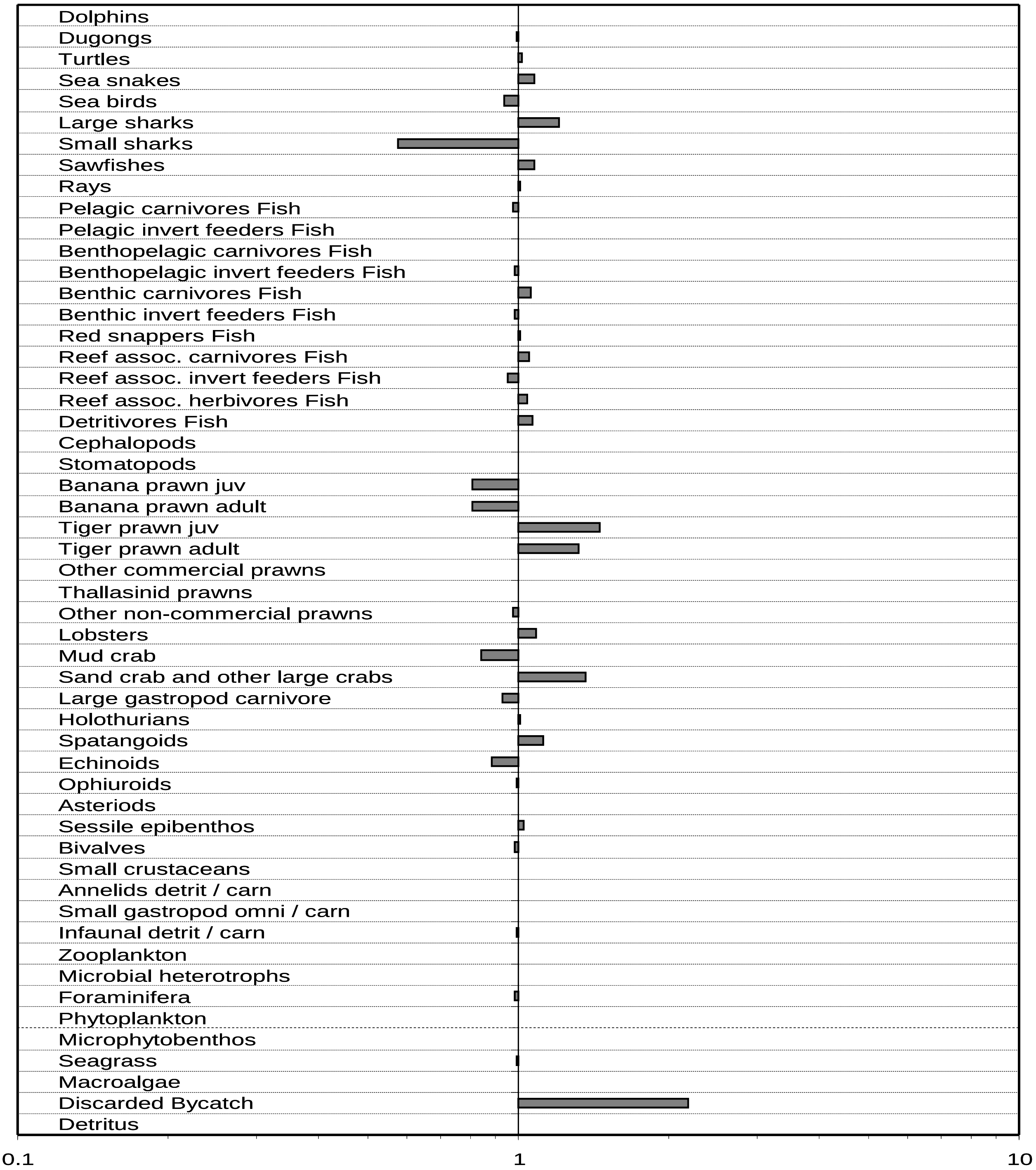

Figure 60. Relative change in biomass for each functional group at the end of simulation (2010) divided by their value at the start of the simulation (1970). Values are shown in a logarithmic scale (greater than 1 indicates increasing while less than indicate decreased biomass).

Our simulation of the recent changes of the NPF (2005-10) involving decreased fishing effort (Figure 59) have had relatively small effects on the biomass of most functional groups in the GoC model

(Figure 60). Again, small sharks, large gastropods and echinoids were negatively affected while large sharks, tiger prawns and sand crab exhibited an increased biomass at the end of the simulation (Figure 60). No major changes (<10%) were shown for discarded bycatch as with most other functional groups.

The results shown in Figure 61 show a clear picture of how the accumulated effects of trawling have affected the trophic structure of the ecosystem. We found a tight relationship between prawn catches and changes in the mean trophic level (TL) of the catch. Since 1970 until around 1980, the TL decreased from 3.78 to 3.46, while catches increased (Figure 61). However, as regulations and management decreased fishing effort to ameliorate decreases in prawn catches, the TL began to return to initial values of around 3.60 (Figure 61).

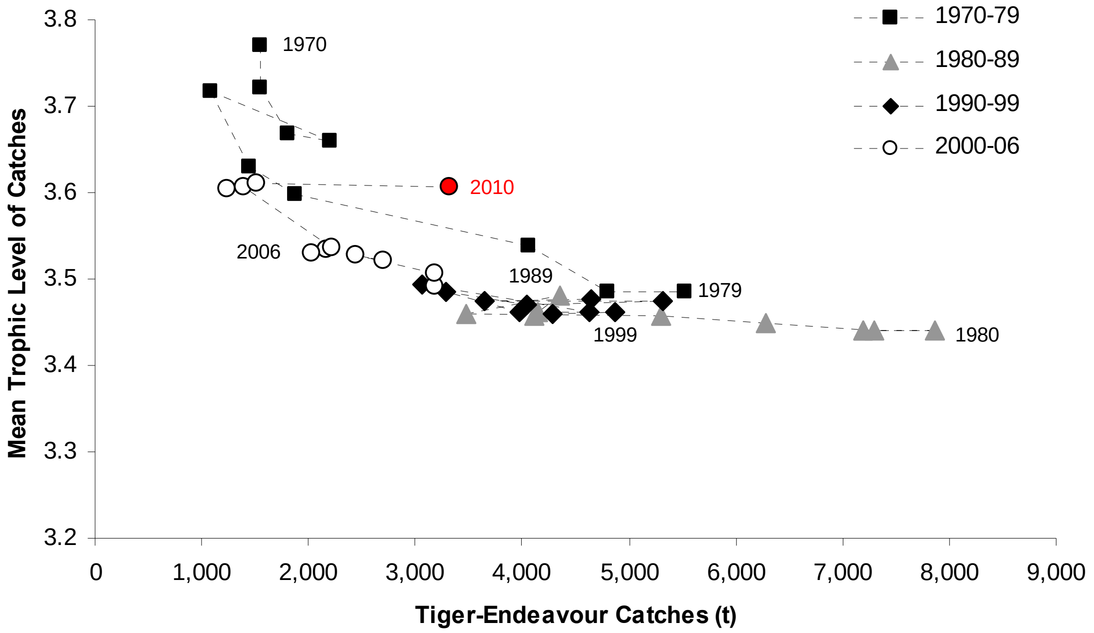

Figure 61. Decadal relationships between the observed tiger-endeavour catches and the predicted mean trophic level (TL) of the catches of all combined fisheries of the GoC. Catch value for 2010 is derived from its stock assessment.

9.5 DISCUSSION

Ecopath models provide a useful theoretical framework in which the ecological effects of fishing and other perturbations, such as climate change, can be explored in isolation or in unison. The various mathematical assumptions of the Ecoapth model, the requirement of vast quantities of input parameters (for which there are many unknowns), difficulty in calibrating, and validating model results are often cited as weaknesses that have limited the uptake of the model predictions into fisheries management policies (Pauly et al., 2000). However, in recognising these limitations, Ecopath models can be very useful for strategic fisheries management, rather than tactical fisheries management (e.g. setting Total Allowable Catches) where the rigor of single-species stock assessment models is required.

The enormous complexity of trophic interactions within species rich ecosystems like the GoC, which vary on various spatial and temporal scales, means that ecosystem models like Ecopath may be one of the few ways in which the ecological effects of fishing may be explored. Predictions can be made as to the direction and magnitude of change to the biomasses of specific groups and the flow-on effects throughout the system, which may not be immediately obvious direct predator-prey effects.

Our model provided unexpected results in that reducing the effort of demersal trawling in over the past decade had only negligible effects on the biomass of any functional group in the GoC ecosystem. However, this is not to say that demersal trawling does not have an effect on the ecosystem structure in the long-term. For example, by reconstructing historic biomass estimates by calibrating the model with stock assessment estimates of adult tiger prawn biomass, it was clear that the mean trophic level of the ecosystem was higher prior to the peak in effort in the 1980s. A decline in trophic level after this period may be a result of removal of small and large sharks, as this was prior to the introduction of Turtle Excluder Devices (TEDs) which became a mandatory requirement only relatively recently in 2000 (ref). Although a relatively small number of sharks are occasionally caught in the fishery during the present day, there has been a significant change in the attitudes of fishers, who now release sharks alive. The ban on taking sharks in 2008, primarily for finning, has probably also contributed to the rebound of sharks in recent years and the increase of the mean trophic level of the ecosystem.

The lack of dramatic changes in ecosystem structure as a result of changes in fishing effort in recent years highlights an important point about how reference points for single species (or functional groups) and ecosystems used my fishery managers may be misleading. Our results demonstrate the potential dangers of a shifting baseline, where the true assessment of a population or ecosystem status may be flawed without information on the historic state of the fishery, especially prior to fishing. Ecosystem models, such as Ecosim, have shown in this case to be able to provide very useful information on the state of the ecosystem and temporal changes in the biomasses of non-commercial species groups that are generally not monitored over long periods. This should help guide future studies that can fill key knowledge gaps in order for ecosystem and single species models to be improved and refined in future.

Ecosystem models are complex and require quantitative data for a large number of biological, ecological and fishery parameters, for which there are many unknowns. In this study, we resisted using parameters from models of similar system in other parts of the world, as this produces models that do not accurately describe the ecosystem, and are of little use to guide ecosystem based fisheries management. Instead, we used direct field observations to construct spatially explicit predictive models to estimate the biomass of many functional groups for which these types of data are difficult or expensive to collect or estimate. We believe this is one of the main strengths of our ecosystem model, since it is comprised of regionally-specific and species-specific parameter estimates. Nonetheless there are numerous shortcomings in our model that may be addressed in future to further improve the model.

Understanding the basic biology of key species representing functional groups and their exploitation by fisheries – as a target or bycatch - is imperative for building a high quality model that is representative of the ecosystem being modelled. In particular, growth studies are required to produce estimates of P/B, while laboratory or 24-hour sampling of animals in the wild is required to estimate consumption rates (i.e. Q/B). It is acknowledged that these studies can be laborious and expensive, but until accurate quantitative data is collected, the outcomes of ecosystem models may be biased in unknown directions. With regards to fishery exploitation, the model could be greatly improved by obtaining more accurate fishery landings and discard data. For all fisheries included in the model, catch data is recorded by fishers on a daily basis in compulsory logbooks. The reliability of logbook data is verified in many of these fisheries by scientific observers, who also collect detailed data on discard composition and biomass. Unfortunately, confidentiality policies of NT and QLD fisheries agencies prevent the release of data from fisheries, or specific regions, where fewer than 5 boats operate. This poses a problem for GoC fisheries since many fisheries are comprised of only a few vessels, for example, the N9 gillnet fishery and the fish trawl fishery. For future models to be improved, access to such datasets is imperative since they inform the model of additional mortality sources. This may involve directly contacting fishers to request the use of their logbook data.

An important point to note about fisheries data used in the model is that it has several shortcomings in terms of quality and taxonomic resolution. The largest problem was that reporting of catches in logbooks was only at high taxonomic levels, such as “shark” or “reef fish”. These groups can contain several species that may be members of different functional groups. Unfortunately, there are few independent data sources (e.g. scientific observer data) that can provide an accurate breakdown of these groups to the species level. As a result, the fishery landings or discards allocated to particular functional groups may be underestimated or overestimated. For fishery landings and discard data to be accurately represented in future ecosystem and single species models, logbook reporting is required at the species, or at the very least genus, level. Although this is beginning to be attempted for some species groups in some fisheries (e.g. sharks in gillnet fisheries), the taxonomic resolution of the catch that is currently reported is inadequate and maybe improved by education programs in each respective fishery to improve the species identification skills of fishers and to promote the importance of high resolution data for better informing fishery models and subsequent management. Indeed, the Northern Territory and Queensland jurisdictions are obliged to improve logbook shark identifications and both have or will introduce upgraded logbooks. They are also obliged to provide observer monitoring of the gillnet fisheries to help improve species identification and logbook verification.

An important procedure in Ecosim models is calibrating the model to time series of known - or good indices of - biomass, fishing mortality and catch. This helps to improve the reliability of predictions made by the model, since the model should be able to recreate past known trends before it can forecast into the future. Unfortunately, long time series (>10 years) are uncommon, except for commercially important species. However, these commercially important species often represent a small number of species within a restricted range of trophic levels. For example, time series for pelagic ecosystems are often only available for apex predators such as tunas, billfish and sharks that comprise the main target, byproduct or bycatch species in commercial fisheries. In contrast, in demersal trawl fisheries, time series are often available for species occupying lower trophic levels such as forage fishes (e.g. blue grenadier) and prawns. This means that when calibrating the model to a few groups occupying the same trophic level, the model cannot be informed about the impacts of changes in biomass on predator and prey species, and thus estimates of the vulnerability parameter become highly uncertain. If fishery managers wish to make a serious attempt at using ecosystem models to implement ecosystem based fisheries management, long time series of abundance of a range of species representing different trophic levels is required to improve dynamic model predictions.

A final recommendation for improving ecosystem model prediction relates to spatially explicit models such as Ecospace. These models require information on the preferred habitats of each functional group and the degree to which they move between these habitats, and also the proportion of time they may spend in marginal habitats where they may be more vulnerable to predation. These issues may be addressed using new advancements in tagging technologies, such as acoustic and satellite tagging. For example, pop-up archival satellite tags (PSATs) provide detailed information on depth and temperature preferences as well as movement, residency patterns and swimming speed, without require the animal to be recaptured or sacrificed. Increasing miniaturisation of these tags means that much smaller animals can now be tagged, which are likely to occupy lower trophic levels than species that have traditionally been tagged using these methods, such as sharks, turtles and tunas.

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APPENDIX 10 A SPATIALLY-EXPLICIT FRAMEWORK UNDER ECOSYSTEM-BASED MANAGEMENT TO ASSESS AND MANAGE THE IMPACT OF FISHING OF THE NORTHERN PRAWN FISHERY

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10.1 INTRODUCTION

Fisheries management is inherently complex as it involves a resource that is a national public asset, yet (often) only entrusted to a few fisher owners (Gordon 1954, Beddington et al. 2007). This means that individual business objectives such as maximising profits need to be traded against the medium-term impacts of fishing on the ecosystem, and inter-generational needs for a clean and sustainable environment (Grafton et al. 2007, Costello et al. 2008). Defining objectives and trade-offs for fisheries management is therefore often extremely complex and politically sensitive (Gislason et al. 2000, Mardle and Pascoe 2002, Hilborn 2007). Ecosystem based (fisheries) management (EBM) or the Ecosystem Approach (EA) has an overall objective to “sustain healthy marine ecosystems and the fisheries they support” (Pikitch, Santora et al. 2004), so it is therefore important to investigate the impact of fishing at target species and ecosystem scales.

The implementation of EBM approaches require integration of a range of data sources, tools and models that can be complex, involving bioeconomics, ecology and biophysics (Christie et al. 2006, Pitcher et al. 2008). One approach available is management strategy evaluation (MSE), a simulation-based framework that allows evaluating broader ecosystem-based fishery management strategies via the use of integrated models coupled with management decision rules (Smith 1994, 1999, Smith et al. 2007, A’mar et al. 2008, Mapstone et al. 2008, Dichmont et al. 2008). An MSE models the whole management and biological systems together allowing comparison and evaluation of the relative performance of different management strategies (see also Sainsbury et al 2000). The MSE consists of an operating model that can be considered as a ‘virtual’ resource and is seen as a representation of the ‘true’ underlying dynamics of the system and the fishery. The operating model for a EBM approach would include the biology of the components affected by fisheries, such as the benthos and associated ecosystem, and all the processes that control the dynamics of that system. The operating model also generates all the data needed within the management strategy. This management strategy remains ‘ignorant’ of the ‘truths’ included in the operating model other than the data given it. Each combination of the types of data used, the assessment related analysis method applied, and the decision rules used constitutes a different management strategy. The outcome of the management strategy (e.g. the level of effort, which areas and times are open to fishing etc.) is fed back to the operating model and is used to determine the dynamics of the ‘true’ situation being managed. The key component that links the management strategy and the operating model is a fleet dynamic model that applies the effort spatially and temporally, or as required.

Biological populations and their communities and habitats have an inherent spatial structure often linked by oceanographic processes whose interactions operate on a number of spatial scales (Botsford et al. 1997, Worm and Myers 2003; Ware and Thompson 2005). Our knowledge about this spatial patchiness is based on how oceanographic processes change over space and time, and how multispecies assemblages interact with each other and with forces at different spatial and temporal scales (Sanchirico 2005). Spatial management in fisheries is therefore one of the many tools that can be used to address fisheries management objectives. They are certainly not a silver bullet but can be a powerful tool to address many aspects of fisheries management (Pelletier, Mahevas 2005, Garcia et al. 2007).

Networks of marine protected areas (MPAs) and spatial planning are the centre pieces for management and conservation of Australia’s marine biodiversity (Commonwealth of Australia. 1998, 2006). These spatial management provisions are been rolled out within its bioregional planning process that has divided Australia’s EEZ in five major bioregions (Heap et al. 2005, Commonwealth of Australia 2006) where detailed spatial assessments of the conservation values and its threats are been produced (Commonwealth of Australia 2008).

In the Australian Commonwealth’s fisheries harvest policy, controlling total effort or catch in a fishery is generally managed through direct and often non-spatial Total Allowable Catch (TAC) or Total Allowable Effort (TAE) controls (DAFF 2007). However, spatial management is used extensively for other purposes in Australian fisheries management such as protecting key habitats and protecting key components of the population such as spawning or juvenile grounds. Pascoe et al (2009) provides four high-level generic objectives for Australian fisheries management: “enhance economic performance, ensure sustainable resources (commercial species), minimise environmental impacts and minimise externalities (encompassing both commercial and social objectives)”.

In this paper, we use a tropical prawn trawl fishery to evaluate the resulting impacts of different spatial management scenarios that include various forms of closing to fishing while achieving the biodiversity or specific fisheries management objectives. The Northern Prawn Fishery (NPF) is a small fishery of about 50 vessels in the far north of Australia (Figure 62) and is managed by a transferable gear unit system, seasonal closures and extensive spatial closures. Most of the present closures are aimed at minimising the capture of small prawns (a.k.a. shrimps) and avoiding key habitats such as seagrass beds (Kenyon et al. 2005). In this study we therefore do not test these since it is difficult to separate these effects from other management options. The paper therefore tests spatial management options with objectives other that those for target species management.

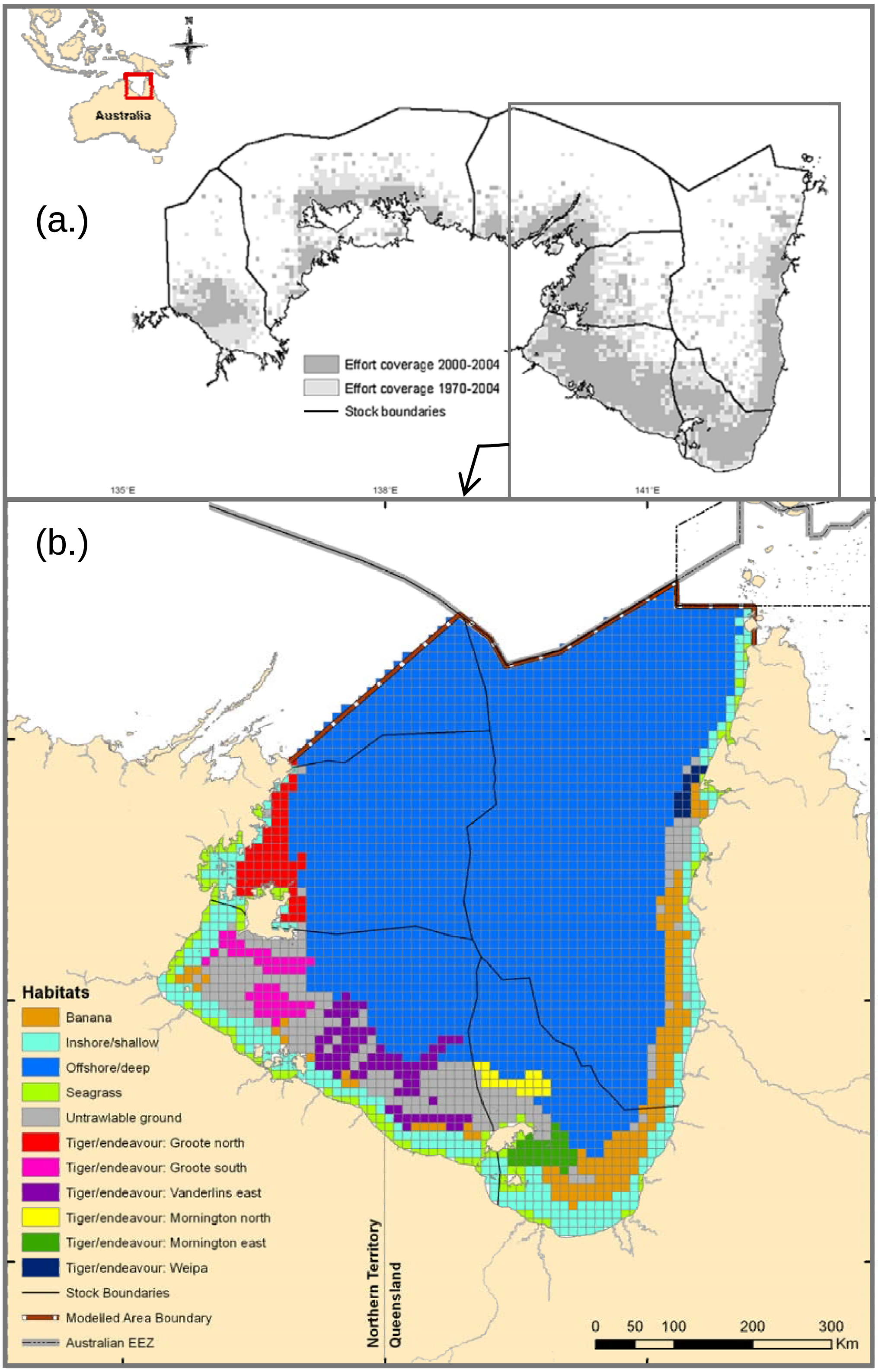

Figure 62. Map of the overall span of the Northern Prawn Fishery (a.), its main stock regions (as thin black lines), its main fishing grounds (shades of gray) and (b.) the details of the main benthic habitats within the Gulf of Carpentaria (c.) defined in a 6-minute grid and used by the EcoSpace operating model.

The NPF falls with the North marine Bioregion, and it expected that a network of new marine reserves and their zoning arrangements will be implemented in 2011/12. This network of MPAs is likely to include areas that are have been historically used by fishing, particularly within the Gulf of Carpentaria (Figure 62), where the most valuable fishing grounds occur. This type of spatial management will need to monitor and evaluate the consequences of both the additional closures to fishing as well as the effect of fishing on the benthos on the open areas.

Given the above, we investigate the effects of a range of spatial closures (beyond those already in use) that include a) mocked up large biodiversity-centric MPAs (since the actual ones are yet to be made available for the public) to remove all fishing, b) ecological risk assessment (ERA) closures aimed to reduce the number of non-target species that are assessed as being at risk in the Commonwealth risk assessment/management approach (Hobday et al 2007) despite present management actions (Griffiths et al. 2007, AFMA 2009, Deng et al. in prep.), and c) closures that have at their core the objective to reduce the overall impact of trawling on key taxa groups. We attempt to schedule these closures so that the impact on the fishery is minimised wherever possible and shift effort given that effort is managed through a TAE. The different closures are assessed in terms benthic and ecosystem impact while assuming effort is shifted with no economic or target species impact. This means that this present method is limited to small closures within trawled areas. Although the closures used in this paper are examples only, they allude to a potential framework of spatial closures to mitigate the impacts of fishing on the ecosystem for various specific objectives.

10.2 METHODS

10.2.1 BACKGROUND

The basic Management Strategy Evaluation (MSE) framework has been described extensively elsewhere (e.g. Kell, Mosqueira et al. 2007). Conceptually, however, it includes three key components: a) the operating models that describe “reality”; b) the management strategies that are to be evaluated; and c) the performance measures that will be used to evaluate the performance of each management strategy in relation to the objectives. The use and development of MSE principles are well developed within the NPF for target species (Dichmont, Deng et al. 2006), and including economics (Dichmont, Deng et al. 2008). However, here we expand this framework so that a) the operating model is a spatially explicit ecosystem model (EcoSpace – Walters, Pauly et al. 1999), and b) the management strategies include the present bio-economic assessment and rules, but is extended to include an effects of trawling model (Ellis and Pantus 2008) that uses a set of rules to adaptively closures (but not re-open) mostly low impacted grids within a habitat. Performance measures now are more focused on ecosystem, benthic impact, in particular based on risk assessment conducted on non-target taxa (see section 10.3.3 below) and indicate how these various components are linked for the NPF Spatial MSE. In this case, the MSE concentrated on the stocks within the Gulf of Carpentaria (Figure 62), where, on average, 87% and 78%% of the tiger (*Penaeus esculentus* and *P. semisulcatus*) and endeavour (*Metapenaeus endeauvori* and *M. ensis*) prawn species are caught.

Figure 63 depicts the overall system, the relationship among the spatial MSE framework components and the data types communicated among components. There is a clear separation along the horizontal axis of the large and small spatial scale components, which distinguishes between the overall stock-region prawn models and the fine scale 6-min grid ecosystem, risk assessment or benthic impact models (Figure 63). A hierarchical fleet dynamic model which starts at stock level and then places effort at the fine scale level within a stock.

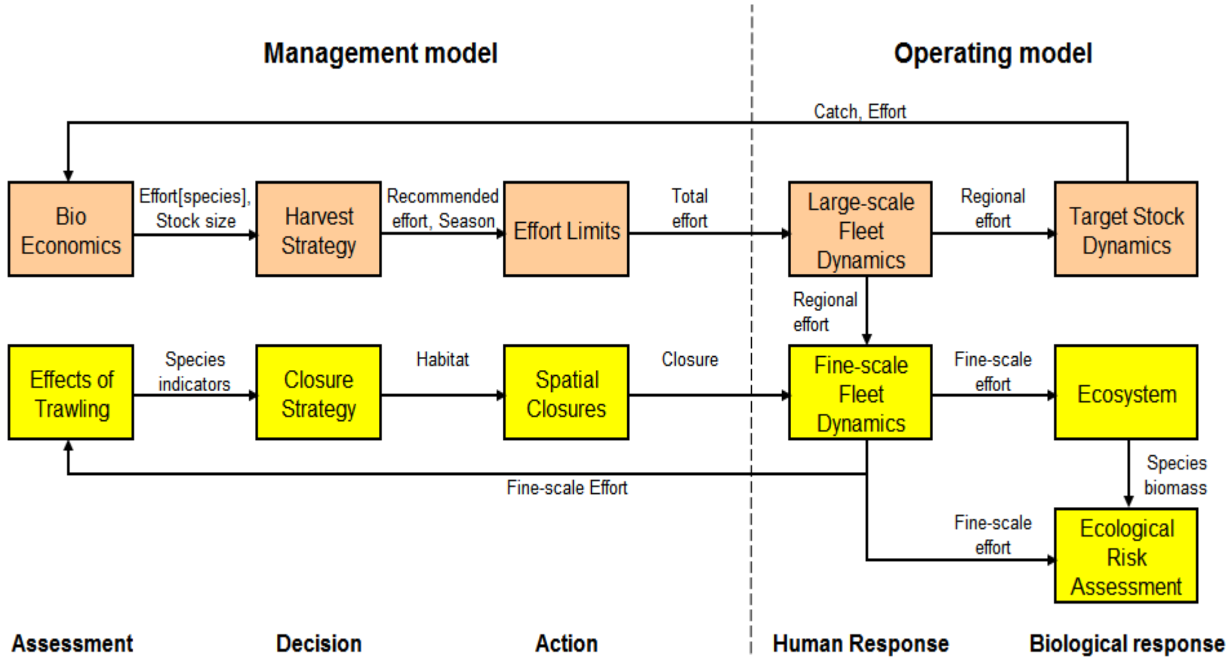

Figure 63. Schematic diagram of the MSE system. Each box represents a separate sub-model. Communication between sub-models and the kind of data supplied are shown by the labelled arrows. The dashed line separates the management model from the operating model. The models also split into large-scale components (*brown*) on the scale of stock regions, and fine-scale components (*yellow*) on the scale of 6-minute management cells. The only link between the large-scale and fine-scale components is the regional effort from the large-scale fleet dynamics sub-model.

10.2.2 OPERATING MODEL

10.2.2.1 ECOSYSTEM

The ecosystem is described using EcoSpace, the spatial-explicit version of EcoPath with EcoSim (Pauly, Christensen et al. 2000). This model builds on two previous ecosystem models developed for illegal fishing in the region (Pascoe et al. 2008) and for the recruitment dynamics of banana prawns (Okey et al. 2007), however these models were only used for the scoping and start up specification of the structure of a largely benthic model develop by Griffiths et al. (in prep, see Appendix 9). This is the model used here and we believe it reflects recent research as well as an ecosystem that would be more useful for questions based on NPF effort.

Ecospace is the spatial and temporal module of the Ecopath with Ecosim software package (www.ecopath.org; Christensen et al., 2005). Ecospace is a dynamic, spatial model that incorporates all key elements of Ecosim (including different vulnerabilities and split pools presented in Appendix 9). The Ecospace model is structured on biomass pools, linked by trophic relationships (i.e. predator-prey), which migrates among the grids of cells of the marine park map. Movements of functional groups (FG) are driven by parameters such as foraging behaviour, avoidance of predation, and dispersal rates that are linked to a range of defined habitats preferred by each functional group. Robust default estimation for these parameters based on life histories is built into Ecospace (Walters et al. 1999; Christensen and Walters, 2004). We did a sensitivity analysis to explore which of these parameters have a strongest impact of the overall biomass predictions, an important step in the understanding of the modeling framework. Ecospace still needs much work in order to make it a useful policy exploration tool for marine ecosystems (i.e. seasonal and ontogenetic migration; Christensen et al. 2005). A discussion of the capabilities and limitations of Ecospace approach can be found in Christensen and Walters (2004).

Ecospace (Walters et al. 1998) models the feeding interactions of functional groups in a spatially explicit way. A simple 6-minute grid represents the study area, and it is divided into a number of habitat types (Figure 62). Each functional group is allocated to its appropriate habitat(s), where it must find enough food to eat, grow and reproduce - while providing energy to its predators and to fisheries. Each cell hosts its own Ecosim simulation and cells are linked through symmetrical biomass flux in four directions; the rate of transfer is affected by habitat quality. Optimal and sub-optimal habitat can be distinguished using various parameters such as the availability of food, vulnerability to predation and immigration/emigration rate. By delimiting an area as a protected zone and by defining which gear types are allowed to fish there and when this occurs, we can explore the effects of marine protected areas (MPAs) and test hypotheses regarding ecological function and the effect of fisheries. Previous authors have used Ecospace in this capacity (e.g., Walters et al., 1998; Pitcher et al., 2005).

The biomass of functional groups is initially distributed over the modeled region, and biomasses and fluxes among cells are governed by dispersal rates related to food availability and predation rates in the cells (Christensen et al, 2005). Then, the ecosystem is divided into a 2-dimensional grid of cells. Each cell is defined according to their habitat type and the preferences of each functional group. In the case of fishing pressure and landings, Ecospace incorporates the time series of effort per fleet per year that were defined in the Ecopath model. It should be noted that these fleets can be varied independently. The spatial distribution of this fishing effort is then controlled by a ‘gravity’ model, which allocates effort to each cell proportional to the relative profitability of fishing in each cell.

The habitat types used in the EcoPath model (Table 41) are assumed to be well (but broadly) defined and are therefore used in the performance measures and also in the management model to assess the impacts of trawling by habitat. This assumption is valid because the relevant habitats (the ones in which fishing occur) are based on effort and VMS data which are both accurate at the 6-minute scale. The Gulf of Carpentaria model is represented by a grid of 6,400 cells (80 x 80 cells). The Ecospace habitat base map (see Figure 62) was built based on the major fishing grounds identified in the gulf: (1) tiger/endeavour North; (2) tiger/endeavour South; (3) tiger/endeavour Vanderlins East; (4) tiger/endeavour: Monington north; (5) tiger/endeavour: Monington east; (6)) tiger/endeavour: Weipa; (7) Banana; (8) Untrawlable ground (9) seagrass; (10) inshore and shallow waters; (11) offshore and deep waters (Table 49).

Table 41. Habitat type, name, acronym with description used within the EcoSpace model.

Type	Habitat Name	Acronym	Description
Fisheries ground	Tiger/endeavour Groote north	grtn	Northern Groote green-mud hot spot
Fisheries ground	Tiger/endeavour Groote south	grts	Southern Groote hot spot
Fisheries ground	Tiger/endeavour: Vanderlins east	vane	South-central harder-grounds hotspots
Fisheries ground	Tiger/endeavour: Mornington north	mtnn	North-Mornington hotspot (sponges & heart urchins grounds)
Fisheries ground	Tiger/endeavour: Monington east	mtne	South-east riverine/tidal sedimentary basin
Fisheries ground	Tiger/endeavour: Weipa	weipa	Offshore NE Weipa
Fisheries ground	Banana	banana	East-south (including inshore Weipa) banana hotspots
Biological	Untrawlable ground	utg	Rough grounds Submerged coral reefs
Biological	Seagrass	sg	Shore and coastal/tidal seagrass beds
Geomophic	Inshore/shallow	inshore	Depth <15m, shallow and back reef /Untrawlable ground
Geomorphic	Off-shore/Deep	offshore	Depths ~>45m. Oceanic-like but muddy largely off-trawling grounds

10.2.2.2 PRAWNS

The population dynamics of prawns were modelled in the operating model in the same way as in (Dichmont, Deng et al. 2008). Endeavour (*Metapenaeus endeavouri* and *M. ensis*) and tiger prawn (*Penaeus esculentus* and *P. semisulcatus*) species are described using a multi-species, multi-stock model (boundaries are defined in Figure 62). The population dynamics of each species in each region that they occur are governed by a delay-difference model that operates on a weekly time step. The conditioning of the operating model involved estimating annual recruitments from data on catches and standardised catch-rates and using these to estimate the parameters of a Ricker stock-recruitment relationship (cf. Dichmont et al., 2003).

The values for natural mortality, growth, weekly availability and recruitment and spawning proportion were based on tagging data, on analyses of experimental data, and on analyses of survey data. Targeted effort for one species in a given region leads to catches of all species that are found in that region (i.e. technical interactions are included in the operating model). Unlike tiger prawns, there is no directed (target) fishery for endeavour prawns (although, anecdotally, operators sometimes target endeavour prawns towards the end of the fishing season) so the fishing mortality for endeavour prawns is the fishing effort targeted at the two tiger prawn species multiplied by species-specific catchability coefficients (computed using the methods of Wang, 1999). Economics are included as a profit function that includes cost and revene based on catches from the operating model. The bioeconomic operating model can therefore produce the following performance measures, (i) S_y/S_{MSY} (%) as the spawning stock size in year, 'y', relative to the spawning stock size at Maximum Sustainable Yield expressed as percentage, (ii) S_y/S_{MEY} (%) as the spawning stock size in a year relative to the spawning stock size at Maximum Economic Yield as a percentage and (iii) the 5-year mav(S_{y-5-y}/S_{MSY}) (%) is the 5-year moving average of the spawning stock size between year 'y' and year 'y-5' inclusive relative to the spawning stock size at Maximum Sustainable Yield expressed as percentage.

10.2.3 RISK ASSESSMENT

A risk assessment method described in Zhou and Griffiths (2008) is used to assess the risk of fishing placing species within the ecosystem at risk. This quantitative Sustainability Assessment for Fishing Effects (SAFE) method consists of two components: calculating fishing mortality, and assessing this with respect to limit reference points. The mean fishing mortality rate (μ) depends on the fishing activity overlapping with each species' core distribution area within the Gulf of Carpentaria, which is then adjusted by the probability of being caught by the prawn trawl. This means that fishing mortality here is not the fishing mortality rate as estimated in stock assessments, but rather the fraction of the population killed by the fishery. A component of this calculation requires the population biomass, which is provided by the EcoPath model. The reference points described in Zhou and Griffiths (2008) and a residual risk assessment (AFMA 2008), showed that exceeded the maximum sustainable fishing mortality rate for 19 species, and exceeded the minimum unsustainable fishing mortality rate for 9 species, mostly threatened marine species of seasnakes. A subsequent ecological risk management assessments conducted by AFMA showed that only 7 non-reptilian species are now considered at priority risk –i.e. two chondrichthyans as extreme high risk, two teleosts as precautionary extreme high risk and three invertebrates as residual high risk (AFMA 2009). Table 2 lists the main taxa included in the ERA and their model parameters. Of course, since only functional groups and not species occur within EcoPath, their respective functional groups were used instead. These groups were assessed each year for the whole of the GoC as well as within key habitats of the projection period so as to determine whether a spatial closure was able to protect the species over time. Two reference points were used: μ_{msm} which is the fishing mortality rate that corresponds to the maximum sustainable fishing mortality (MSM) at the biomass that supports MSM (B_{msm}) (equivalent to Maximum Sustainable Yield for target species; and μ_{crash} which is the minimum unsustainable fishing mortality rate that, in theory, may lead to population extinction in the long term. If the fishing mortality rate is greater than u_{crash}, then that species is at extreme relative risk (Table 50). The ERA was used to establish various spatial-explicit management scenarios for conservation management objectives. A different spatial map of the functional groups was used for this process rather than that used in the operating model. This is to avoid assuming perfect knowledge of the spatial structure of the relevant functional group.

Table 42. List and parameters of the marine threatened species of the Gulf of Carpentaria included in the ecological risk assessment and in the spatial MSE.

Taxa	Family	Common Name	Catch Rate	Escape Rate	μ_{msm}	μ_{crash}
Marine reptile						
Acalyptophis peronii	Hydrophiidae	Horned Seasnake	0.47	0	0.163	0.299
Aipysurus duboisii	Hydrophiidae	Dubois' Seasnake	0.47	0	0.178	0.325
Aipysurus eydouxii	Hydrophiidae	Spine-tailed Seasnake	0.47	0	0.161	0.296
Aipysurus laevis	Hydrophiidae	Olive Seasnake, Golden Seasnake	0.47	0	0.092	0.175
Astrotia stokesii	Hydrophiidae	Stokes' seasnake	0.47	0	0.096	0.184
Disteira kingii	Hydrophiidae	spectacled seasnake	0.47	0	0.146	0.271
Disteira major	Hydrophiidae	Olive-headed Seasnake	0.47	0	0.146	0.271
Emydocephalus annulatus	Hydrophiidae	Turtle-headed Seasnake	0.47	0	0.146	0.271
Enhydrina schistosa	Hydrophiidae	Beaked Seasnake	0.47	0	0.146	0.271
Hydrophis caerulescens	Hydrophiidae	Dwarf seasnake	0.47	0	0.138	0.257
Hydrophis elegans	Hydrophiidae	Elegant seasnake	0.47	0	0.104	0.196
Hydrophis mcdowelli	Hydrophiidae	Seasnake	0.47	0	0.138	0.257
Hydrophis melanosoma	Hydrophiidae	Black-banded robust seasnake	0.47	0	0.138	0.257
Hydrophis ornatus	Hydrophiidae	Seasnake	0.47	0	0.139	0.259
Hydrophis pacificus	Hydrophiidae	Large-headed Seasnake	0.47	0	0.139	0.258
Lapemis hardwickii	Hydrophiidae	Spine-bellied Seasnake	0.47	0	0.112	0.212
Teleost fish						
Filicampus tigris	Syngnathidae	Tiger Pipefish	0.3	0	0.982	1.000
Trachyrhamphus longirostris	Syngnathidae	Long-nosed Pipefish, Straight Stick Pipefish	0.3	0	0.982	1.000
Chondrichthyan fish						
Anoxypristis cuspidata	Pristidae	Narrow Sawfish	1	0.733	0.139	0.235
Pristis clavata	Pristidae	Dwarf Sawfish	1	0.733	0.369	0.601
Pristis microdon	Pristidae	Freshwater Sawfish	1	0.733	0.097	0.185
Pristis pectinata	Pristidae	Wide Sawfish	1	0.733	0.095	0.181
Pristis zijsron	Pristidae	Green Sawfish	1	0.733	0.095	0.181

10.2.4 FLEET MODELS

The fleet dynamics models that describe the movement of the fleet both at a stock level and then subsequently at a 6-minute grid level within a stock are described in Venables et al. (2009). The models consist of two phases to provide the operating model with effort at the necessary spatial and species (dis)aggregated scale. The first phase is a vessel movement model that assigns effort at the prawn stock and species level at a weekly time-step. The second phase takes the effort allocated to a grid in that year and assigns this to 6-minute grid squares, as needed by the benthic impact model. This model assumes that fleet behaviour is a reflection of past habits but modified based on what was caught in the previous week and at that time in previous years. Based on these assumptions, closures that removed very large areas of the main fishing ground were not included in the analysis as these assumptions may not be valid under such circumstances.

10.2.5 MANAGEMENT STRATEGIES

Management strategies normally consist of monitoring, assessment and decision, normally framed within an adaptive cycle. The target species monitoring and assessment is described in detail in Dichmont et al. (2008) and in brief below. Since it has been shown that the bio-economic model achieves the best profit pathway (compared to other models and strategies tested), while keeping both the target species sustainable and benefits the impact of trawling on the benthos, this model has been retained to set effort levels for each tiger prawn stock over time. Furthermore, this bio-economic assessment is presently used in the fishery. Since the spatial closures are not aimed at managing total effort and are assumed to be small enough to not substantially change the fleet dynamics between stocks (i.e. only displacing effort within stocks), each scenario has the same effort by year and stock, with the differences between scenarios only occurring at the grid level. Effort on banana prawns is assumed to be kept constant over time. Once the effort level is set for the tiger-endeavours fishery, this effort is applied at a 6-minute grid scale (assuming a uniform effort distribution) over a 10-year horizon the benthic impact model (Dichmont et al. 2008).

10.2.5.1 TARGET SPECIES

In all the scenarios, the fishery is managed based on setting effort to maximise profit thereby moving the fishery to Maximum Economic Yield (MEY). Given a new assessment (undertaken every alternate year), this involves first determining whether any of the species is “overfished” (defined as the average spawning stock size over the most recent five years being below 0.5 S_{MSY}), in which case the fishery is closed until that species has recovered to above 0.5 S_{MSY}. If none of the species is assessed to be overfished, the sequence of future effort levels is calculated so that profit over a 50-year projection period is maximized. The estimation has a lower bound so that effort in a given year is not less than half of that directed towards *P. esculentus* in 2007 (a value recommended by industry and management). The future effort for the first seven years (with the effort for the eighth and all subsequent years set to that for the seventh year) is selected to maximise total profit – this is therefore a dynamic bio-economic model.

The profit function is based on parameters described in Dichmont et al (2008), which forms the basis for estimating the time-trajectory of future effort, and accounts for costs due to labour, capital, fuel and other causes. Cost parameters were derived from economic surveys of the fishery undertaken by the Australian Bureau of Agricultural and Resource Economics, ABARE (Rose and Kompas 2004, Dichmont et al. 2008, Kompas et al. 2009,). Endeavour prawns are essentially caught as a bycatch of targeting tiger prawns. This means that management is essentially aimed at tiger prawns, and endeavour prawns are only considered in the bio-economic model as added revenue, while the catches of endeavour prawns only increase costs through catch-associated costs (i.e. packaging, labour etc.). All spatial closure scenarios use the same effort by stock, year and week so that target species actions are kept consistent.

10.2.5.2 BENTHIC IMPACTS

The Effect of Trawling (EoT) models the estimated primary benthic impacts of repeated trawling on the biomass of benthic organisms, ignoring any long-term consequences of that removal on the ecosystem, including any effect on prawn productivity (Ellis et al. 2008). It consists of three components given input on effort by 6-min grid cell: (i) a component that calculates the depletion of the biomass on a range of benthic species (or taxonomic groups) that occur in the NPF given repeated trawling; (ii) a component that determines the rate of recovery of each of the species; and (iii) a component that distributes the biomass (initially) uniformly over space (Dichmont et al. 2008). The model is described in detail in Dichmont et al. (2008).

Since the Ecopath model works at a functional group and not a species level, only functional groups that include a large proportion of benthic animals at risk are included here (Table 2). A data poor environment is assumed in which the distribution of animals is assumed equally distributed i.e. assumed to be essentially unknown. This would be the case in data poor situations. The benthic impact model is forward projected for 10 years and using the following decision rules, a decision is made on whether to close a set of grids. This is done for the EoT model indicator for species *s* and habitat *h* is

$$I_{hs}(t) = \frac{1}{N_h} \sum_{g \in h} B_{sg}^{\infty}(t) \text{ where } B_{sg}^{\infty}(t)$$

is the projected steady-state relative biomass (i.e. projection for infinity years, not just 10 years) under recent spatial effort distribution and current E_{mey} at time *t*. The indicator is the average projected relative biomass within the habitat.

10.2.6 SCENARIOS

Several scenarios were tested trying to address several spatial closures that could be used (beyond those currently existing for target species and seagrass beds) to protect species within the ecosystem (Table 43). The Base Case (BC) is a scenario that describes the presently used fisheries management system where no additional closures are in place and where effort is set based on the bio-economic stock assessment model. In all cases, the operating model was kept consistent (beyond spatial closures of course) and the management model used the bio-economic model to set effort levels. The other spatial management scenarios were developed by; (i) using an externally (to the MSE) derived risk assessment model aimed at moving the classification of “at risk” functional groups to “not at risk”, (ii) are dynamically and adaptively set when the EoT model triggers a set threshold, or (iii) are set in an ad-hoc manner using expert knowledge like in the case of MPAs (Table 43)

Table 43. Scenarios used to test various spatial management options thereby describing the operating model, which management scenario is applied and which additional spatial closures were used. Low spot= low fishing effort; Hot spot= high fishing effort; L= low ERA species biomass; H= high ERA species biomass; MPA = conservation closure; EoT70 = 70% biomass change threshold in the EoT model.

Scenario	Name	Acronym	Management scenario	Closures
Status quo. No changes in fishing and set as 1990 as reference year	Base Case	BC	Bio-economic model	None
Examples of representative closure of benthic biodiversity for conservation reasons	Marine Protected Area	MPA	As above	Ad-hoc closures as in Figure 9-3a
Adaptive closure via simulating the effect of trawling (EoT model) with % threshold. Once cells are closed remains closed.	EoT 70%	EoT70	Effects of trawling model with a limit reference point of 70% change of initial biomass for target functional group in a habitat	Set closures adaptively, as in Figure 9-3b.
ERA closure on low biomass density (low spot) and low fishing effort. Cells close low fishing effort and externally derived biomass density cells	Low spot Low	LdLf	ERA closures	Close cells when ERA threshold criteria (Table9- 4) are reached as in Figure 9-3c.
ERA closures on low biomass density (low spot) and high fishing effort. Only close cells from those with low density and high effort	Low spot High	LdHf	As above	Close cells when ERA threshold criteria (Table 9-4) are reached as in Figure 9-3d
ERA closure on high biomass density (hot spot) and low fishing effort. Close cells from high biomass density and low fishing effort cells	Hot spot Low	HdLf	As above	Close cells when ERA threshold criteria (Table 9-4) are reached as in Figure 9-3e

10.2.6.1 DEVELOPING SPATIAL CLOSURES

Based on the SAFE method (Zhou and Griffiths 2007), the species at high-risk within overall functional groups of the modelled ecosystem were determined. This process was undertaken external to the MSE, with the resultant closures imported to the MSE. Key to the SAFE method is the overlap with these groups' spatial distributions with that of the fishery and this is also a major aspect that can be controlled through management. In order to minimise the impact on the fishery, two of the three sets of spatial closures were developed that target locations where effort is limited (called "Low spots"). Regional patterns of the distribution and abundance were based on the biophysical species distribution modelling (Brown et al. unpubl), but are not the same as those used for the operating model. An empirical data transform model using a random forest estimator as a non-parametric method for modeling the biological response to environmental gradients was used. All available biological and environmental data were used. Based on these models, the biomass in a 6-minute grid of the key functional groups that contained at risk species were calculated and used to identify possible spatial closures. This biomass density was plotted against effort (the most recent 5-year average in boat days) for each grid for each high-risk functional group in each habitat. These plots were used to identify grids that a) are only lightly fished with less than 50 boat days, b) are lightly fished with less than 200 boat days, and c) are only lightly fished with less than 50 boat days plus grids within the "hot spots" where functional group biomass density is large (Table 44). The latter biomass thresholds were based on independent (of the MSE) tests using the SAFE model that showed that only removing "low spots" for certain of the high risk functional groups in certain habitats would not reduce the risk status within their habitat of these groups.

The resultant closed grids for each scenario were the sum of grids identified for each functional group. Risk was assessed at the habitat and whole of GoC level.

Table 44. Biomass thresholds used to determine grids that should be included in the ERA hot spot scenario.

Habitat	Large sharks	Rays	Sawfishes	Sea snakes	Small sharks
North Groote (grtn)	> 475	> 2,000	> 1200	> 25	> 2,500
South Groote (grts)	> 500	N/A	> 600	> 25	N/A
East vanderlins (evdl)	N/A	N/A	> 2,000	N/A	N/A

The effects of trawling closures were determined adaptively within the MSE's management model – i.e. the EoT model identifies whether a benthic functional group in a specific habitat was below a pre-set threshold depending on the scenario, in this case 70% for the "EoT 70" scenario. The most recent year's effort is taken from the logbook data (by grid in the specific habitat) and sorted, and a certain quantile of the grids are closed starting from the lowest effort grids. In the "EoT 70" the quantile is 0.5, i.e. the grids in which recent effort is less than the median recent effort for the habitat. Initially, various "EoT" scenarios were tested (10, 20, 30), but closures were never initiated since the thresholds were never triggered. We decided to set a high threshold to test the utility of such adaptive method.

It is assumed that effort at the stock level remains the same with or without a closure and therefore the small-scale fleet dynamic model is used to shift the effort from the closed grids to the remaining grids within a stock based on historical patterns, which therefore considered that certain grids are more favoured than others (as in Ellis et al. 2008, and Dichmont et al. 2008).

10.2.7 INTEGRATION AND OPERATIONAL ISSUES

Ideally, the prawn operating model would not be needed and should not be used in the strictest definition of an ideal MSE. However, the EcoSpace model works at monthly or annual time steps whereas the fleet dynamic model required a weekly time step. Furthermore, EcoSpace only works at a functional group (FG) level whereas the effort needed to be divided by fleet which is essentially a target-species level concept. For convenience the prawn operating model was only used at the stock levels for the fleet dynamic model. The outputs of the models were compared to test whether they were essentially showing similar trends.

10.2.8 PERFORMANCE MEASURES

All results were expressed as measures of central tendency at 2-quantile or median, whose variability was expresses by their, and 5th and 95th percentiles. The model variability was estimated from 30 replicate model runs conducted by the spatial MSE. The base case (BC) was set at the stock and fisheries levels of 1990. In general, all results for each of modelling scenarios were contrasted against the BC. The model projections were also made from the ecosystem and fishery state at 1990, whereas forecast was conducted for the 2008-2016 period.

The performance of the benthic biomass for each functional group from the operating model at any given scenario was calculated in relation to the biomass of the same functional group in the base case expressed as percentage and calculated as,

$$(Scenario_{medianFG} / BC_{meadianFG}-1) * 100$$

The results of the spatial MSE framework can be integrated into several form of graphical reporting, such as time, habitats, regions, scenarios, and various combinations among them. The results presented below are demonstrative examples of the capability of the modelling system, showing the overall performance of a measure (e.g. median biomass of FGs or mean trophic level) for each scenario, habitat, and regions or through time and space. The habitats used here were those defined in Table 49 for Ecospace while the regions were defined as spatial integrations of related habitats that include; (i) all tiger-endeavour fishing grounds, (ii) all banana fishing grounds, (iii) all inshore, untrawlable grounds and seagrasses habitats combined and (iv) all offshore habitat.

Table 45. Performance measures used in the spatial MSE.

Type	Performance measure	Expression
Biological	Ratio of the stock size at 2016 from the operating model relative to the stock size at MSY	S_{2016} / S_{MSY}
Ecosystem	Benthic relative biomass index	$B^{O}_{t,h,g,f,r}$ is the benthic biomass index from the operating model, O, for the year, t, in habitat h for functional group f, r is the replicate and g is the 6-minute grid
Ecosystem	Mean trophic level of the system after fishing	$T^{O} = \frac{\overline{T}_{2016}}{\overline{T}_{2008}}$ where T is the mean trophic level in 2016 relative to the Ecopath year, 2008

The results of the various spatial management strategies evaluations for any given performance measure were presented mostly by comparing their performance in relation to the base case (BC or status-quo). In this way the MSE results are directly comparables and allows to gain a quick understanding of the trade-offs between strategies. Since the modelled ecosystem is composed of 53 functional groups, the following results are examples and operational proof-of-concept of the capabilities of the spatial MSE.

10.2.9 COMPARATIVE ASSESSMENT

In order to visualize and synthesize the multiple and complex results, the various indicators, performance measures and metrics were expressed into measures of central tendency across various groups and management scenarios (see section 10.4.8).

The relative biomass was expressed by the median (5th,95th) percentiles of the average density in kg per hectare. All the percentage change for each scenario was relative to the base case BC.

For the BC, let $B_{m,r,f,g,y}$ be the density of functional group *f* in grid cell *g* at the beginning of year *y* in simulation *r* for MSE scenario *m*. Define $B_{m,r,f,g,y}$ to be the average density over the Tiger region in 2016 for simulation *r* of scenario *m*:

$$\mathrm{B}_{m,r,f} = \frac{1}{N_{\text{Tiger}}} \sum_{g \in \text{Tiger}} B_{m,r,f,g,2016}$$

Thus we have performance measures $\mathrm{B}_{m,f}^{\text{med}}$, $\mathrm{B}_{m,f}^{5\%}$ and $\mathrm{B}_{m,f}^{95\%}$, which are respectively the median, 5th and 95th percentiles of this average density over all simulations, i.e.:

$$\mathrm{B}_{m,f}^{\text{med}} = \underset{r}{\text{median}} \left\{ \mathrm{B}_{m,r,f} \right\}$$

with $\mathrm{B}_{m,f}^{5\%}$ and $\mathrm{B}_{m,f}^{95\%}$ defined similarly.

For the scenarios, define $\Delta_{m,r,f} = B_{m,r,f} \big/ B_{\text{BC},r,f} - 1$ to be the change relative to the BC scenario of the average density in simulation r for MSE scenario *m*. Then the performance measures are $\Delta_{m,f}^{\text{med}}$, $\Delta_{m,f}^{5\%}$ and $\Delta_{m,f}^{95\%}$, which are respectively the median, 5th and 95th percentiles of this relative change over all simulations.

Each column shows median (5th,95th) percentiles of the EOT indicator in Weipa and aggregated to whole Tiger region

For biomass indicator EoT70, define $I_{m,r,f,h}$ to be the EoT indicator in habitat *h* for functional group *f* in simulation *r* for MSE scenario *m* (see section 10.3.5.2 for the definition of the EoT indicator). Then the performance measures are $I_{m,f,h}^{\text{med}}$, $I_{m,f,h}^{5\%}$ and $I_{m,f,h}^{95\%}$, which are respectively the median, 5th and 95th percentiles of this EoT indicator over all simulations.

For the overall biomass we used the median (5th,95th) percentiles of the percentage change in total biomass in 2016 relative to 2008. Where we define $B_{m,r,y}$ to be the total biomass over the Tiger region in year *y* for simulation *r* of scenario *m*:

$$\mathrm{B}_{m,r,y} = \sum_{f} \sum_{g \in \text{Tiger}} B_{m,r,f,g,y}$$

Define $\Delta_{m,r} = B_{m,r,2016} \big/ B_{m,r,2008} - 1$ to be the change in 2016 relative to that in 2008 of the total biomass in simulation *r* for MSE scenario *m*. Then the performance measures are Δ_{m}^{med}, $\Delta_{m}^{5\%}$ and $\Delta_{m}^{95\%}$, which are respectively the median, 5th and 95th percentiles of this relative change over all simulations.

For the changes in fishing effort, let $E_{m,r,y}$ be the total effort displaced by closures in the Tiger region in year *y* for simulation *r* of scenario *m*. Then define $E_{m,r}$ to be the average annual displaced effort over 2011–2015:

$$\mathrm{E}_{m,r} = \frac{1}{5} \sum_{y=2011}^{2015} E_{m,r,y}$$

Then the performance measures are E_m^{med}, $E_m^{5\%}$ and $E_m^{95\%}$, which are respectively the median, 5th and 95th percentiles of this average displaced effort over all simulations.

10.3 RESULTS

10.3.1 SPATIAL MANAGEMENT SCENARIOS

The six spatial management strategies scenarios developed here had the aim of modifying and altering the spatial distribution of trawling affecting the benthos while also achieving the NPF's fisheries management targets (stock and economics). In essence, the resulting scenarios created a range of fisheries closures of various sizes that were designed to illustrate likely spatial closures relevant to the current NPF management needs that could be implemented given different EBM criteria. The criteria used here include conservation and ecological risk criteria that satisfy the current fishery management objectives.

The Figures 3a-e shows the resulting spatial closures for each simulated management scenario aimed at reducing the impact of trawling on the benthos, changing the stutus of "at risk" species to "not at risk" or increasing the biodiversity as a whole. Trawl closures were defined as individual or clusters of 6-minute grid. The number of closed cells varied according to the scenario and for all but the Eot70 scenario, the closed cells remains the same in all 30 replicated simulations and over time within a simulation i.e. they are implemented at the start of the projection period and are permanent. However, the Eot70 is a adaptive given a performance indicator i.e. closed cell are triggered when the threshold of 70% biomass change of a key functional group is reached in a given group-habitat. This means that the number of closed grid cells varied from replicate to replicate and the ones depicted in Fig. 3b correspond to an illustrative example of a single simulation.

The MPA ad-hoc scenario clustered all closed cells into compact closures that crossed different depth ranges in different regions. All the other closures were grids scattered due to the method used to design these (see Table 44).

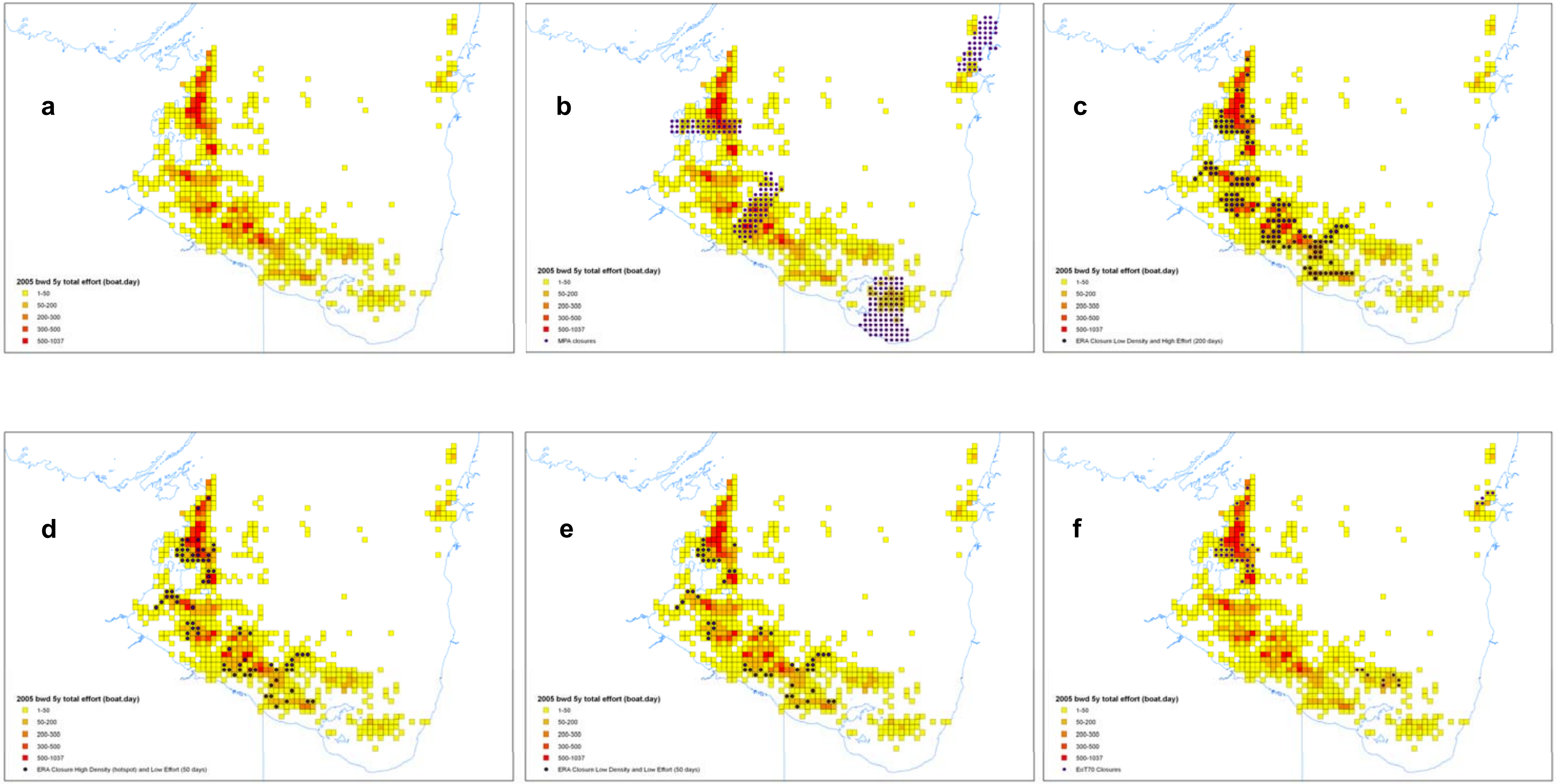

Figure 64 **a-e**. Maps the Gulf of Carpentaria depicting the 5-year average fishing effort from the NPF's logbook data (boat days) in 6-minute grid cells and showing the spatial closures scenarios for: (a.) Base Case (no closures), (b.). MPA), (c.) Low spot High, LdHf, (d.) Hot spot Low HdLf, (e) Low spot Low, LdLf, and (f.) EoT70 (illustrative example for a single replicate).

The Table 6 depict the resulting amount of fishing area being excluded from trawling according to the different spatial management scenarios. In the modelled area of the GoC (396,483 km^2 of surface), trawling for tiger and endeavour prawn species is concentrated in only 20.8% of largely the southern and western shallows (<40-45m) parts of the GoC (Figure 64, Table 46) It was clear that the closures imposed by the scenarios are really small when the whole GoC is considered, since the maximum area closed is 6.6% in the case of the MPAs (Table 46). In terms of closure sizes, the overall four MPA scenario had the largest closure with ~26% of the tiger-endeavour trawling grounds being closed, while the EoT70 scenario had the smallest closure with only ~6% of fishing ground closed (Table 46).

Table 46. Amount of area closed by area and % for the tiger-endeavour trawled areas and the whole GoC for the various spatial management scenarios.

Scenarios	Acronym	# grid cells	Area (km^2)	Closure in relation to trawling ground (%)[b]	Closure in relation to the GoC (%)
Global modelled area GoC	*GoC*	*3,211*	*396,483*	-	-
Base case (status quo)	BC	669	82,606[2]	0.0	0.0
MPAs	MPA (total)	211	26,053	31.5	6.6
	MPA-Grt	37	4,569	5.5	1.2
	MPA-Vnd	52	6,421	7.8	1.6
	MPA-Wei	50	6,174	7.5	1.6
	MPA-Kar	72	8,890	10.8	2.2
ERA closures	LdHf	129	15,928	19.3	4.0
	HdLf	79	9,755	11.8	2.5
	LdLf	53	6,544	7.9	1.7
EoT 70% threshold	EoT70[a]	49±29	6,050±3,581	7.3±4.3	1.5±0.9

[a]= median ± standard deviation; [b]= total area of trawling grounds

10.3.2 FISHING EFFORT

The Figure 4 depicts the 1970-2016 time series of the median annual trawling effort for the two tiger prawn species. From 2008 to 2016 the figure shows the predicted median annual effort with its 95% CI. This series was derived from the bieconomic stand-alone model that satisfied the fisheries management target species objectives –i.e. the stock to be also at Maximum Economic Yield (S_{MEY}) whereafer effort will be at E_{MEY}. All the scenarios used the same effort series by year and simulation so that the only differences between them are the spatial closures. These effort series were then passed into Ecospace to evaluate the ecosystem consequences at the functional group and spatial regions and habitat levels. The effort series show clearly the historical fishing expansion from 1970 to mid 80s, followed by a sharp and steady decline of effort as results of prawn catches declines until the late 90s, largely due to the drastic fishing cuts and industry adjustments (Figure 65). The predicted effort also shows the small but steady increases in effort that commenced in the 2007-8 which is consistent with the MEY approach given the resource is on a path to S_{MEY} (Figure 66). At the end of the projection, the resources are at S_{MEY} for tiger prawns whereas endeavour prawns are still to reach the target. All species are well above the limit reference point – the 5-year moving average of S_{MSY} should be greater than 50% of S_{MSY}.

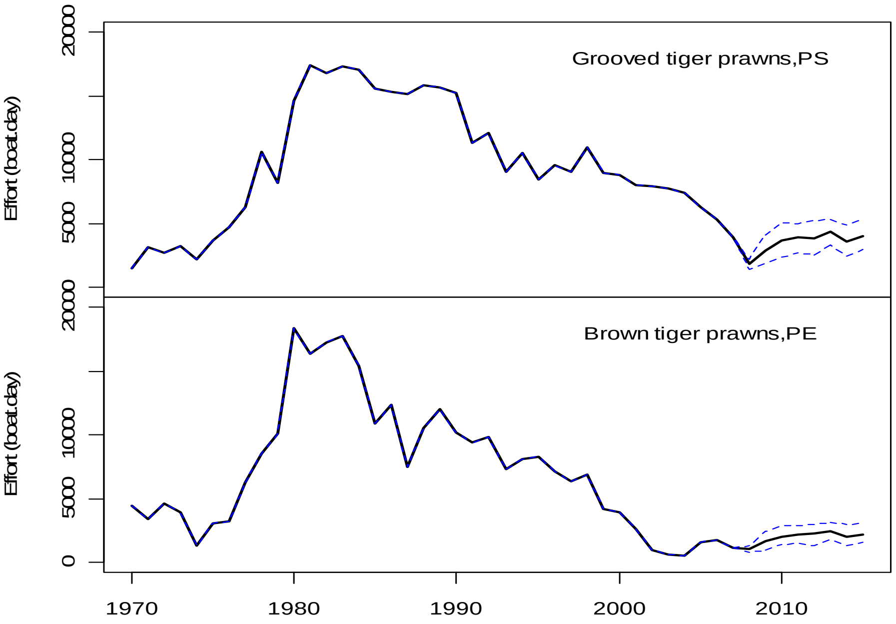

Figure 65. Time series of the fishing effort measured as median trawling effort in boat/day for the two main tiger prawn species with 95% confidence intevals for the projections; *Penaeus esculentus* (brown) and *P. semisulcatus* (grooved). These series were used in the in the fine scale regional effort as input for the spatial MSE.

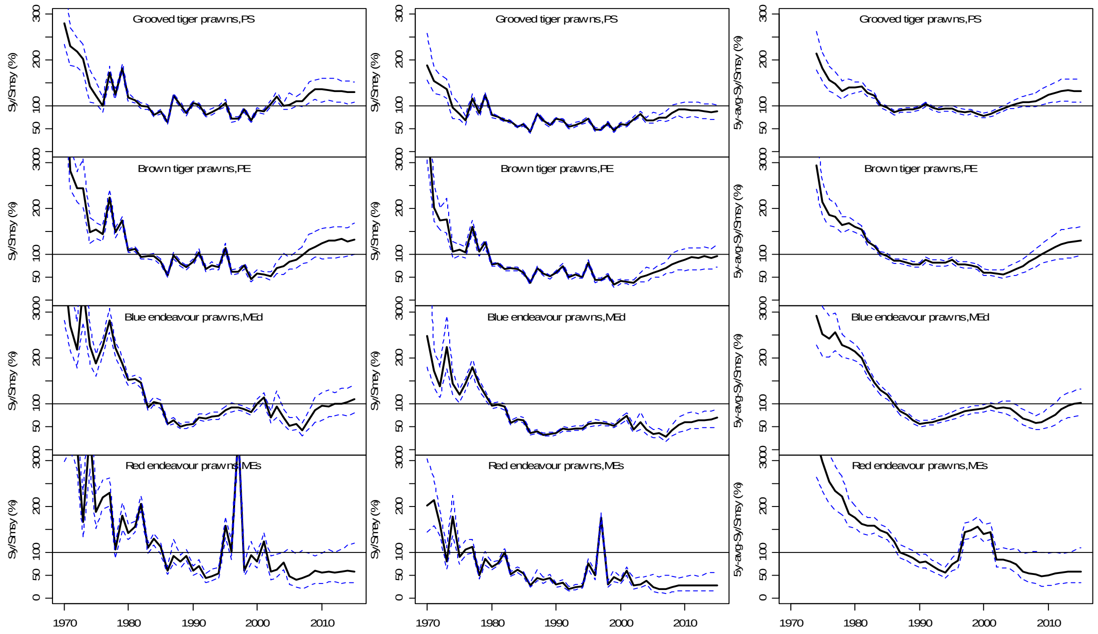

Figure 66. Time series of the three main Sy/Smsy (%), Sy/Smey (%) and 5y-avg S/Smsy (%) performance measures under the predicted mean fishing effort (Fig. 9-4) and generated by bioeconomic stock assessment.

10.3.3 EVALUATION OF ECOSYSTEM CONSEQUENCES FOR ACHIEVING FISHERIES MANAGEMENT TARGETS

10.3.3.1 EOT BIOMASS INDICATOR RELATIVE TO 1990 FOR DIFFERENT HABITATS AND SCENARIOS

Using the Effect of Trawling operating model (EoT), we evaluated the consequences of achieving the NPF's bieconomic target (MEY) on two functional groups across all habitats and under all management scenarios. These two groups have been chosen to (i) demonstrate the capability of the spatial MSE and (ii) because these have been previously shown to contain some of the most susceptible taxa in the GoC to the effects of trawling (Haywood et al. 2005, Dichmont et al. 2008). The Figure 65 depicts the temporal trajectories of the median predicted biomass relative to the reference year, 1990, of the sessile epibenthos and large gastropods carnivore functional groups (Figure 67a and b respectively). The dotted line in Fig. 6 crossing at 0.7 (or 70%) indicates the trigger or threshold point that defines a trawling closure (however of course, this is based on the EoT component model in the management model). This evaluation shows that overall, the performance of the indicator showed small variations that did not differ greatly across scenarios and habitats (<10% change), and only from 2010 onwards there was some temporal variations in the indicator's trajectories between scenario (Figure 67 a,b). In the case of sessile epibenthos, no scenario triggers a fishing closure at any of the 12 habitats and 6 scenarios (Figure 67a), and only on the Weipa and Mornington north fishing habitats were there some larger (between 10 and 30%) changes in relative biomass (Figure 67 a). The trajectories of the indicator for the large gastropod carnivore group showed more changes –i.e. there were larger changes through time and all but the MPA scenario crossed the set threshold in the Weipa fishing habitat (Figure 67b).

This evaluation also shows that at the start (2008) the value of the indicator tended to be around 1 (or 100%), but from 2010 onwards does vary among of the various habitats and scenarios (Figure 65 a,b).

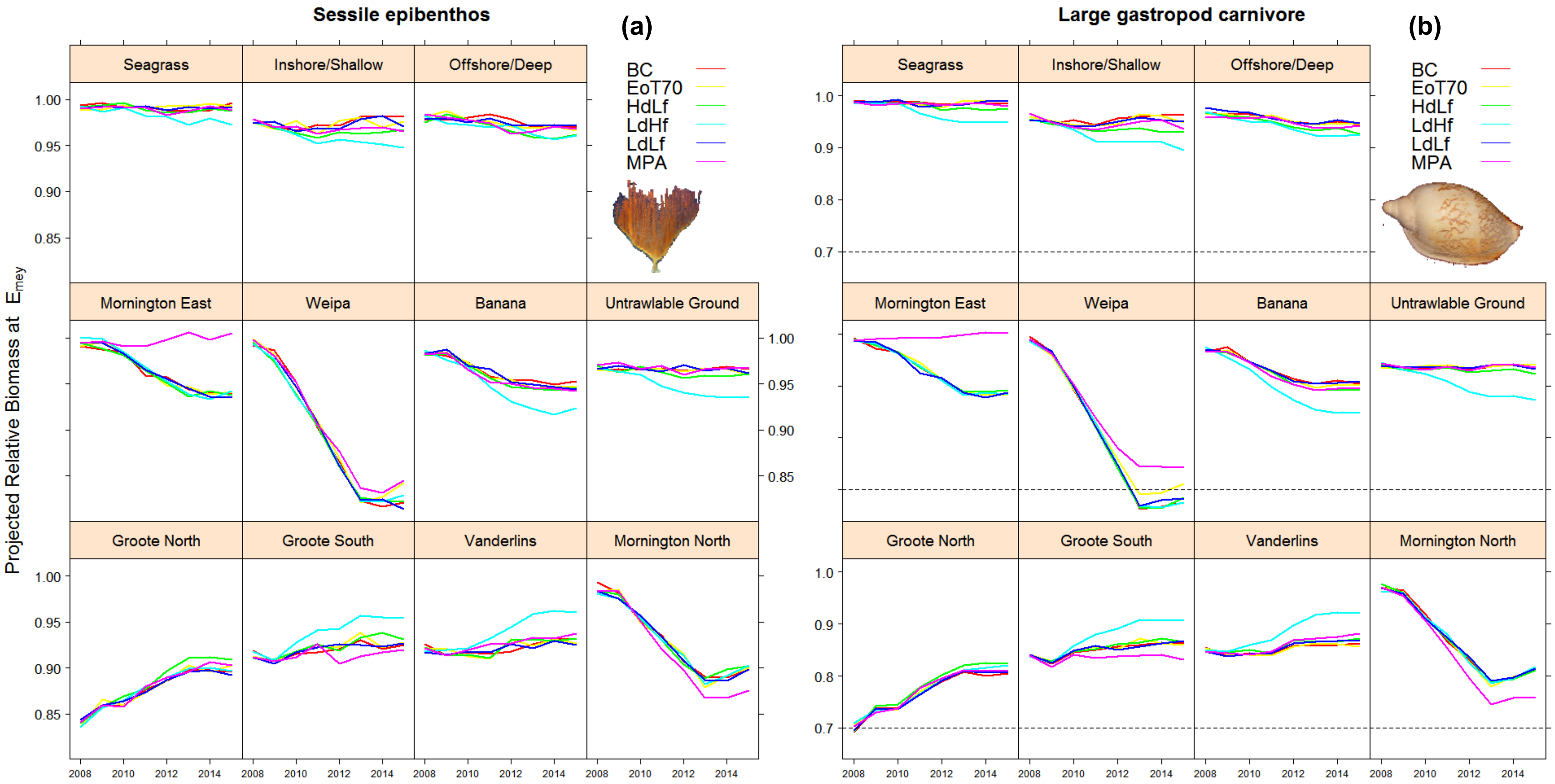

Figure 67 **a-b**. Indicator of the assessing model of the biomass relative to 1990 – the operating model version of the EoT indicator – for the spatial management scenarios and for two illustrative functional groups. The evaluation is done by the EoT operating model.

10.3.4 REGIONAL BIOMASS

To evaluate the spatial footprint of trawling under multiple spatial management strategies (closures), the modelled area of the GoC was divided into 12 major habitats (Fig 1, and Table 1), where trawling occurs mostly in a few habitats types (see 5-year average fishing effort in Figs 3. Results are therefore only presented for areas where trawling occurs with consistency and some intensity.

Figure 68 a-b and Figure 69 a- b show the median projected biomass in 2017 relative to the biomass of the Base Case (BC - no spatial closure) for four major fished regions –i.e. tiger, banana, inshore, offshore. Note that the scales for Figure 68b and Figure 69 a and b are one-to-two orders of magnitude smaller than in Figure 68 a.

It is clear that either no or small changes in biomass were experienced for most functional groups (Figure 68, Figure 69) in relation to the BC. These results also confirm that most of the significant changes, both positive and negative occur in the tiger region (Figure 68a), where most of the trawling for tiger-endeavour prawns occurs. Negligible changes occurred in the inshore and offshore region (Figure 69 a and b) where there is very little or no trawling (Figure 64). The biggest change to the predicted median biomass did not exceed 2% beyond what was experienced in the BC scenario with the ERA LdHf, showing the most changes, followed by MPAs and HdLf (Figure 68, Figure 69). The adaptive spatial closure of EoT70 and the ERA LdLf scenarios resulted in little change relative to the BC across all regions. Despite the small changes detected across all scenarios, the relative biomass of prawns were slightly affected –e.g. adult tiger prawns were positively affected by ERA LdHf closures but negatively by MPAs (Figure 68a). In general all closures tended to predict increases in top predator groups like sharks (Figure 68 a, Figure 69a) that in turn fed on secondary consumers, like prawns, which could explain the decrease on prawn biomass within MPAs (Figure 68a).

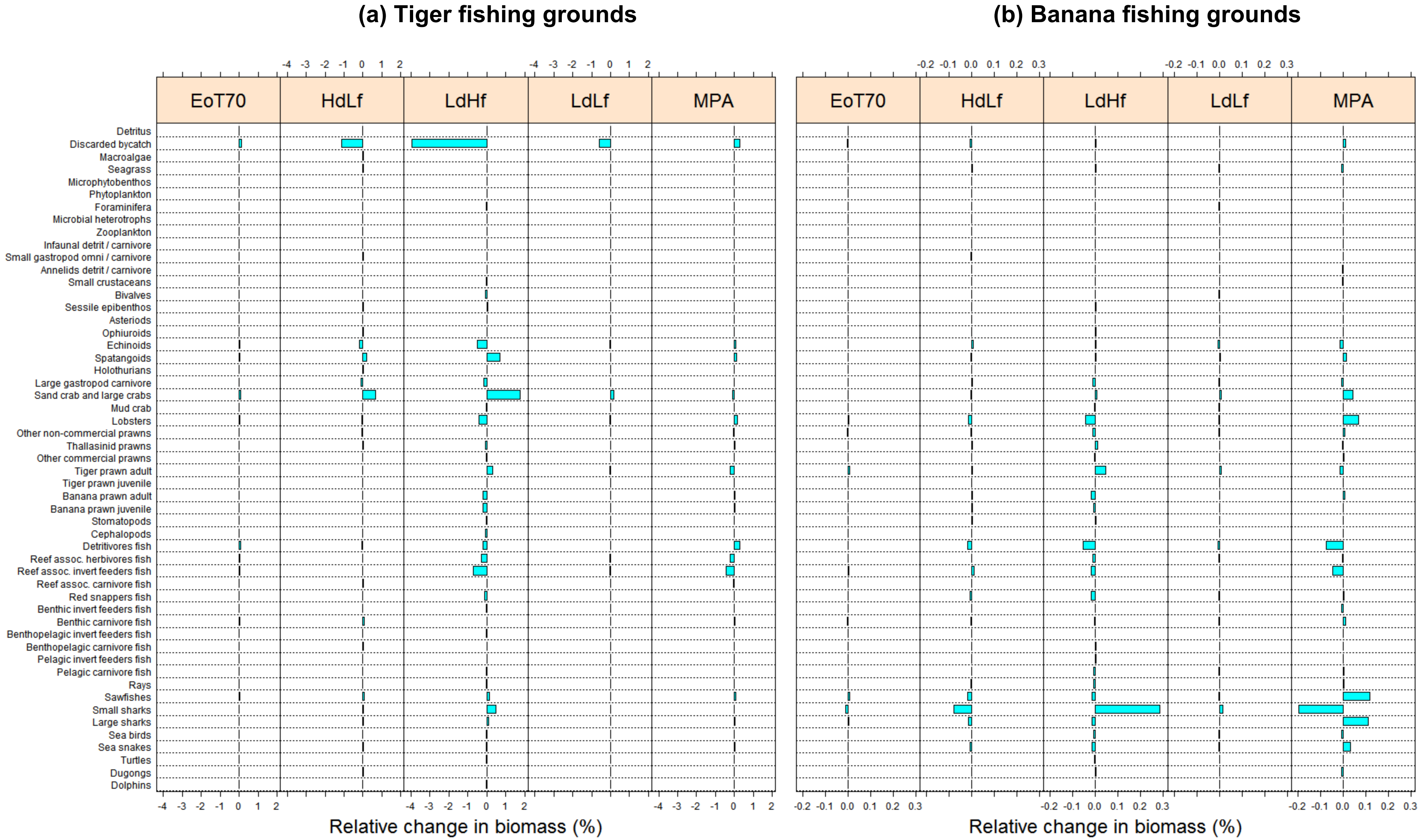

Figure 68 **a-b.** Median changes of relative median biomass predicted for all FGs for all six modelled scenarios projected at 2016 in relation to the base case (BC) in 1990. Values are expressed in %s and reported for the (a) tiger and the (b) banana fishing grounds treated as single regions. Note that the scale for (7b) is one order of magnitude smaller than in (7a).

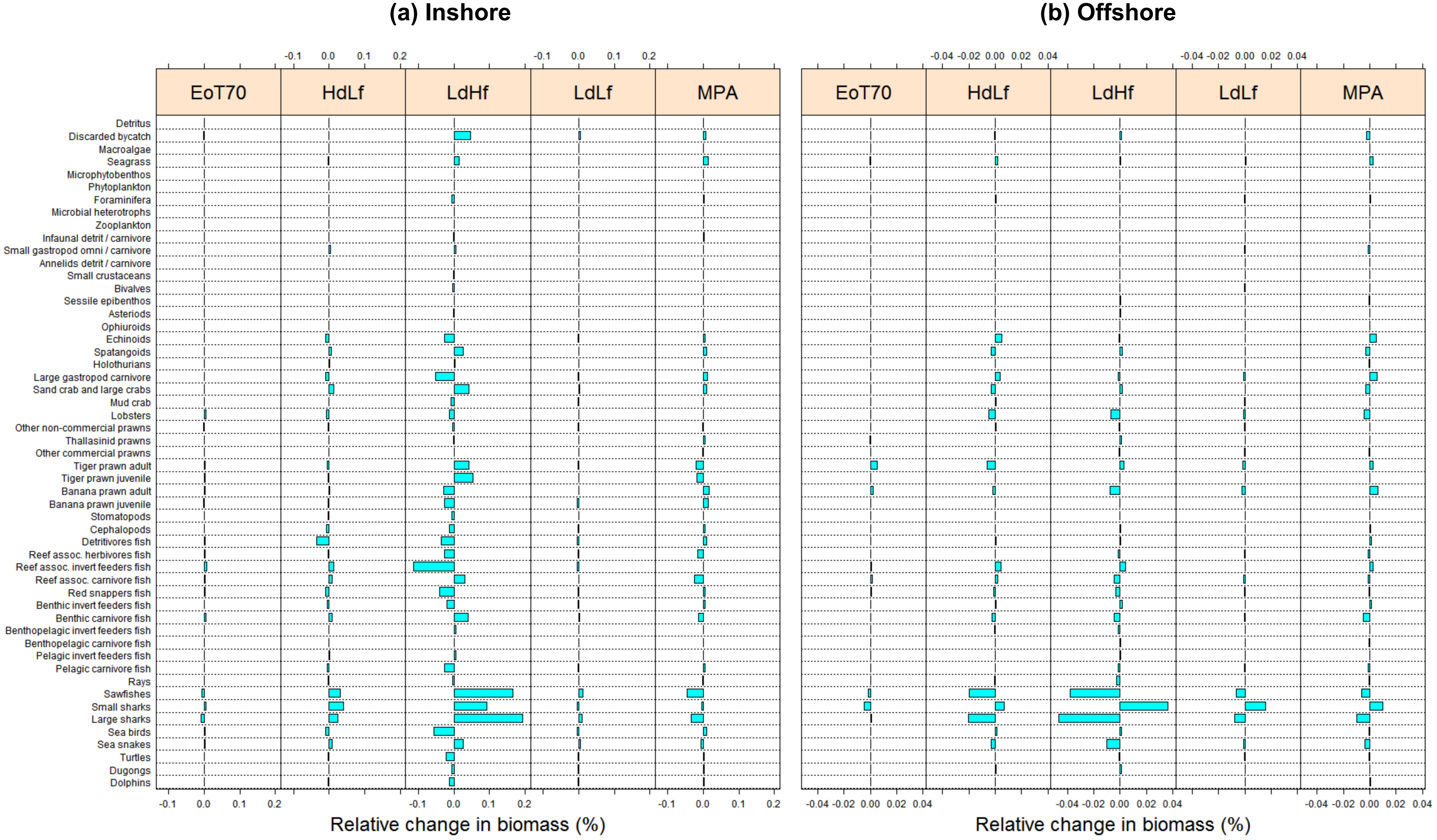

Figure 69 **a-b.** Changes of relative median biomass predicted for all FGs for all six modelled scenarios projected at 2016 relative to the base case (BC) in 1990. Values are expressed in percentes and for (a) inshore and (b) offshore regions. Note that the scale for (9-8a) and (9-8b) are one and two order sof magnitude smaller than in (9-7a).

10.3.5 TEOMPORAL BIOMASS VARIATION PER HABITATS

Below are a few examples of biomass relative to the BC from the 53 functional groups that were selected from the aggregated results presented in Figure 67 for: sand crabs, (Figure 70a), adult tiger prawns (Figure 70b), large sharks (Figure 71a) and large gastropods (Figure 71b). Al predictive biomass per habitat figures for all functional groups are presented in the Appendix 13A.

These show clearly that the spatial effects of the different modelled habitats are very different by scenario, especially in trawled habitat (Figure 70, Figure 71). The least changes for most groups were found to be in the inshore, the seagrasses and offshore habitats while the tiger-endeavour fishing habitats showed the largest deviation from the BC (Figure 70, Figure 71). When changes were observed in habitats with little or no trawling, these variations can be explained by the trophic-induced (or predator-prey) dynamics of the predicted biomass of the various functional groups flowing into these habitats (e.g. large sharks inFigure 71 a).

The performance of the various spatial management strategies was also evident at the temporal and habitat scales for the simulated period. It was clear that for each group and each habitat, even where the trawling effect is strong (on fishing grounds), most scenarios remain similar to the BC, with ±5% variation from the BC as the biggest inter-scenario variation (e.g. sand crabs Figure 70 a). In general, the scenarios that showed most differentiation were the LdHf and MPAs (Figure 70, Figure 71). Despite the inter functional group and scenario deviation from the BC, some habitats showed similar trends –e.g. in Wiepa most scenarios differ above and below the BC, in Vanderlins most scenarios were the same of greater than above the BC, or in Mornington north most scenarios were equal or below the BC (Figure 70, Figure 71). These results were in general consistent for most other (not shown) functional groups. These results also showed that, despite the scenario, deviations from the BC increase towards the end of the simulated period, most likely due to the fact that fishing effort increases from 2010 onwards (Figure 70, Figure 71).

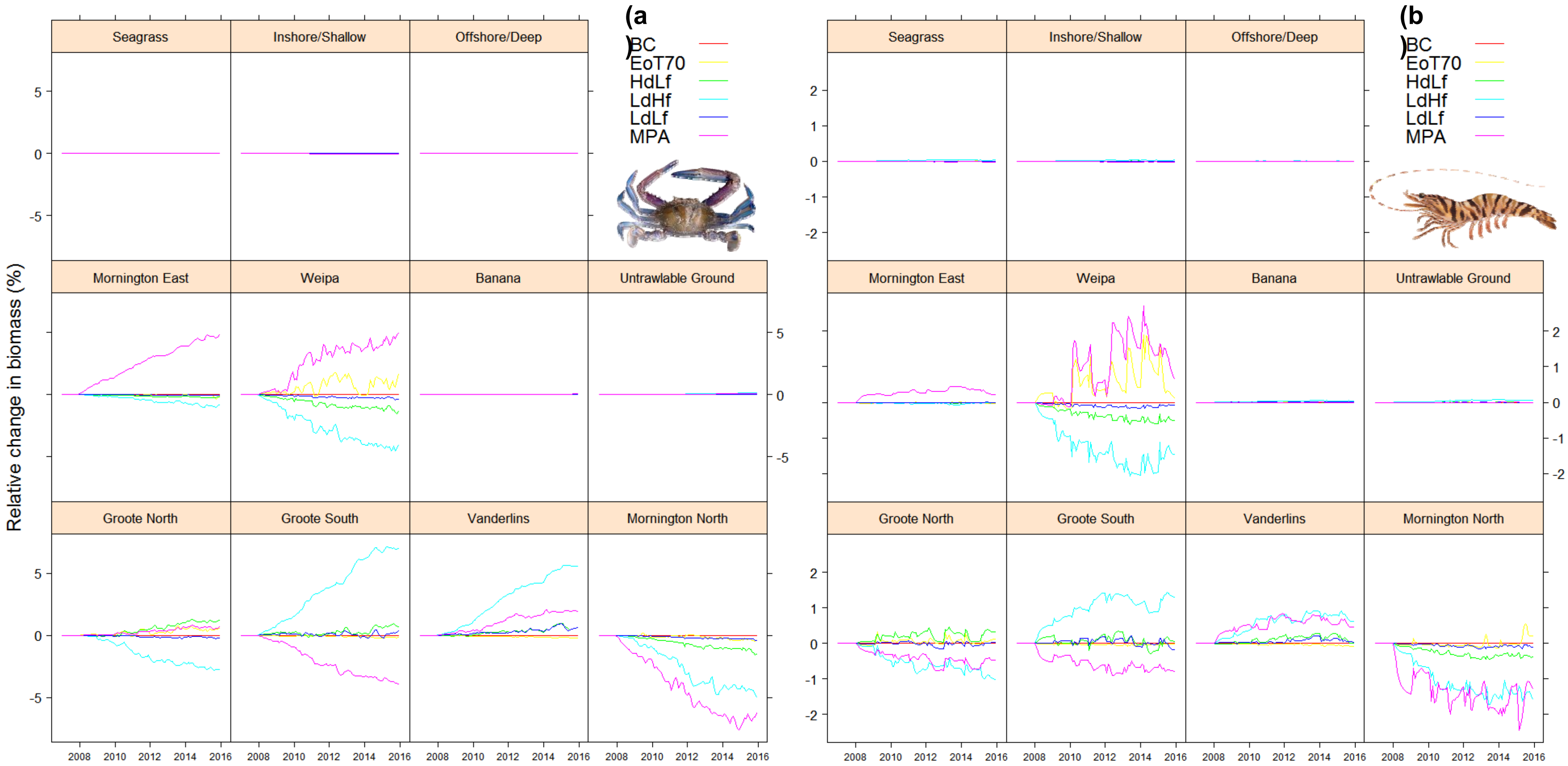

Figure 70 **a-b**. Changes to the predicted biomass relative to the Base Case for all six modelled over the projection period for the (a.) sand crabs and other large crabs and (b.) adult tiger prawn groups. All habitats are shown in different panels.

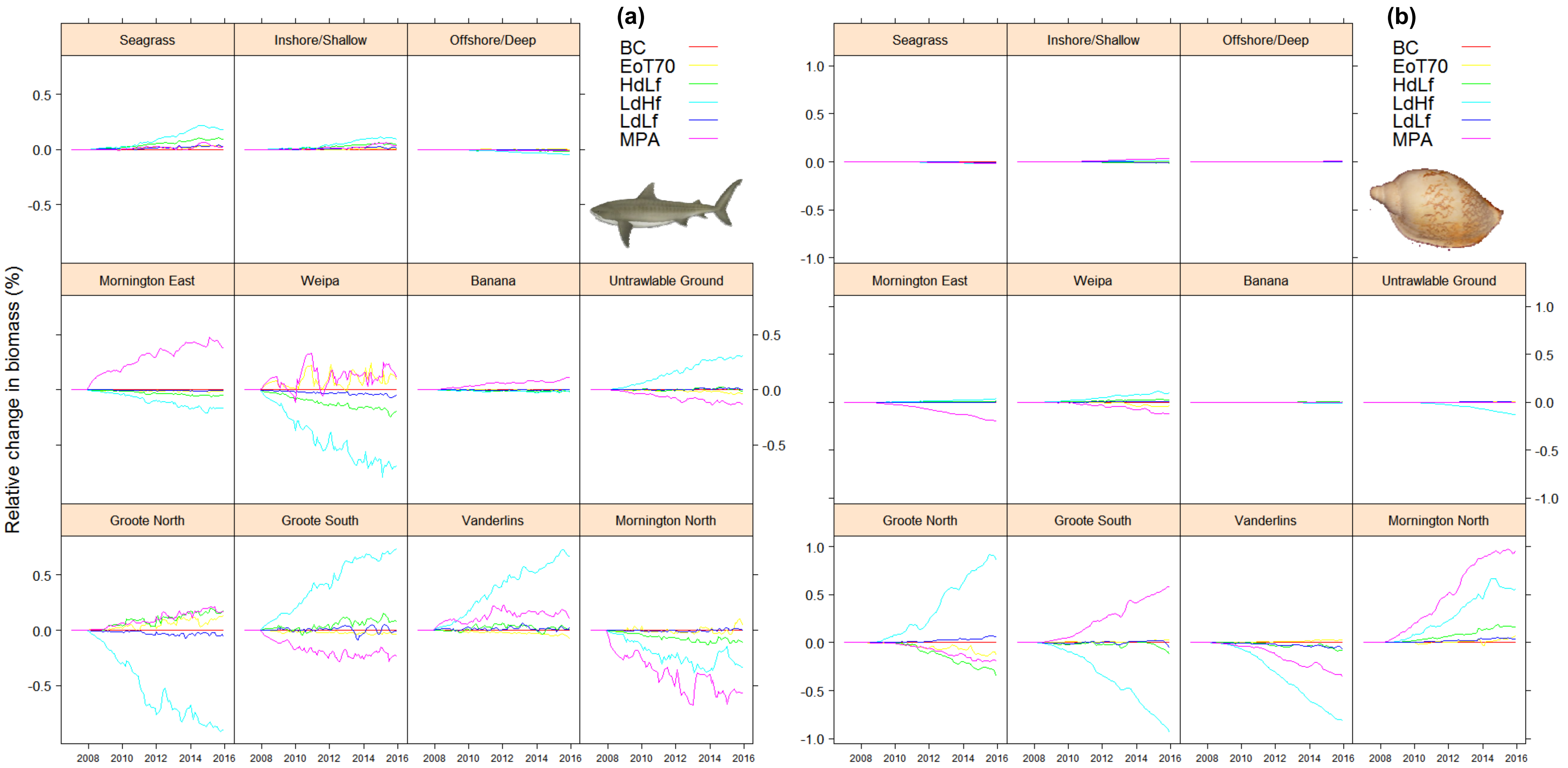

Figure 71 **a-b**. Changes of the predicted biomass relative to the Base Case for all six modelled scenarios for the projection period for the (a.) large shark and (b.) large gastropod carnivore groups. All habitats are shown in different panels.

10.3.6 ASSESSING THE SPATIAL PREDICTIONS OF THE ECOSYSTEM BIOMASS

The spatial MSE also allows assessing the detailed spatial distribution of the predicted biomass for any functional group across any time step. The Figure 72, Figure 73 are showing illustrative examples, for various functional groups, of their relative biomass for each management scenarios, in relation to the BC (status quo) for the end-of-simulation year 2016. Al predictive biomass at fine-scale spatial figures for all functional groups are presented in the Appendix 13B. Is important to note that the results depicted below are of all different scales, so the colour prediction for one group is no the same for another –i.e. each figure allows comparison among the scenarios but not across functional groups.

As with other biomass evaluations (regional, habitats), these spatial predictions are showing clearly that positive (towards red) and negative (towards light blue) responses were found for all functional groups in response to the spatial management scenarios (Figure 72 a-d). This is suggesting that some groups are being more, nil or less affected by trawling. These results are also confirming that different groups respond spatially different among the scenarios. Here, some groups are predicted to have their biomass increased inside spatial closures like with sessile epibenthos (Figure 72a), and seasnakes (Figure 72b), while in others their predicted biomass decreases, as with echinoids (Figure 72c), and bivalves (Figure 72d), as examples. We found also that most groups that respond negatively to closures are those that are in general preys of secondary consumers and predatory groups like cephalopods (Figure 73a) and small crustaceans (Figure 73b). Conversely, most top predatory groups increased their biomass inside and around closures rays (Figure 73c) and benthic carnivore fishes (Figure 73d). These findings are consistent with the expected underlying food web dynamics that is being imposed by trawling.

Spatially, these results confirm also that the LdHf scenario showed the most spatial variation in relation the BC (status-quo) across all functional groups, while the EoT70 is the scenario with least changes (Figure 72, Figure 73). These results also showed that the different functional groups respond spatially different to the regional and habitat variation of the trawling effects, the only changing factor in the simulations. Is interesting to note, that the imposed closures have in cases a clear spilling-over effects into the surrounding non-closed cells, like in the cases of the sea snakes and cephalopods (Figure 72 b and Figure 73a, respectively). This spilling over is normally swamped in the cells subjected to most intensive trawling, like for example in the fishing hot-spot north of Groote Eyland (Figure 72b,c and Figure 73c,d).

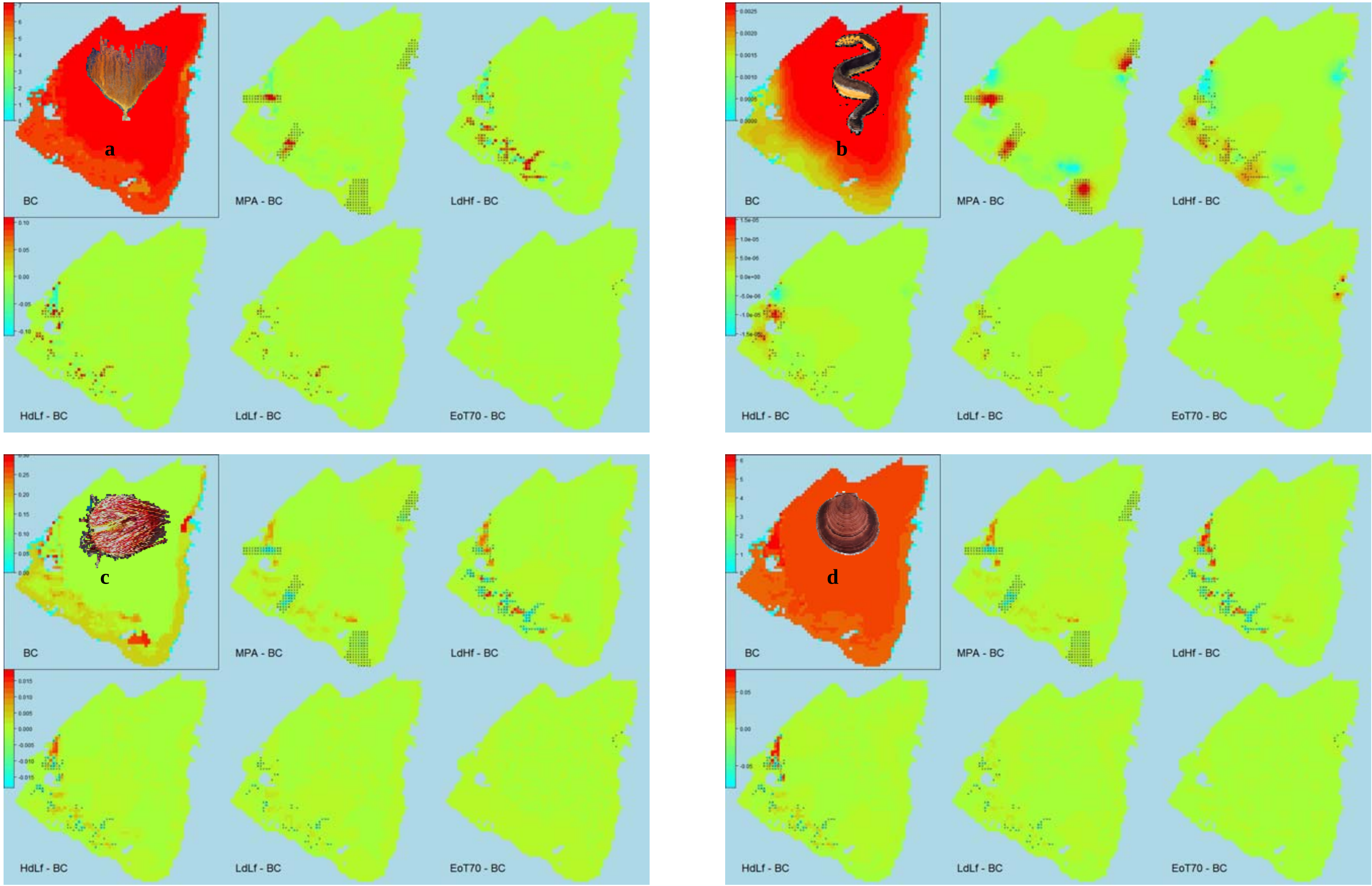

Figure 72 **a-d**. Predicted changes at 2016 of the relative biomass of the functional groups; (a) sessile epibenthos, (b), seasnakes, (c) echinoids, (d), bivalves. Different panels show the spatial prediction of each modelled scenarios in relation to the base case (biomass - biomass BC). Spatial closures are shown and black crosses. Results are of a single simulation.

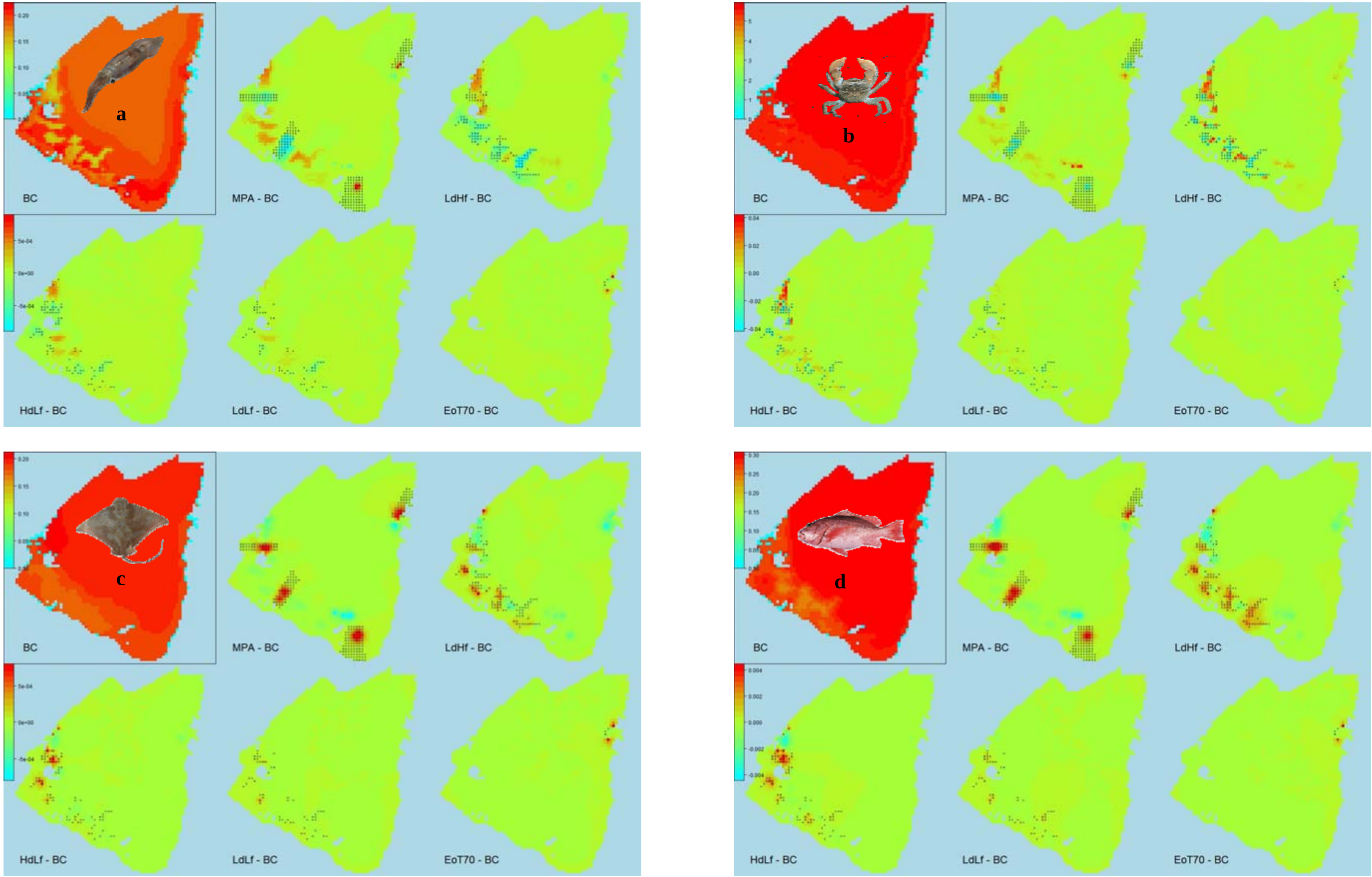

Figure 73 **a-d**. Predicted changes at 2016 of the relative biomass of the functional groups; (a) cephalopods, (b) and small crustaceans, (c) rays, (d) benthic carnivore fishes Different panels show the spatial prediction of each modelled scenarios in relation to the base case (biomass - biomass BC). Spatial closures are shown and black crosses. Results are of a single simulation.

10.3.7 MEAN TROPHIC LEVEL

We evaluate here the relative performance and trajectories of the mean trophic level of the ecosystem under the various scenarios in relation to the BC (Figure 74). Our results shown that the different scenarios in the current simulation period and under the current and predicted fishing efforts, are inducing little and very small changes (of less than 1%) in the mean trophic level of ecosystem under trawling (Figure 74). This is not unexpected for this type of performance measure, since the historical variability (i.e. 1970 to 2010) of the NPF's mean trophic level ranged between 3.77 and 3.38 (see Figure 61, Appendix 9). What is important here, are not the absolute values but their temporal trends and variability of the performance measure under the different management scenarios.

Despite the small changes, we found pronounced differences among the scenarios (Figure 74). As with other biomass results, the HdLf, LdLf, and EoT70 scenarios were not very different for the BC, although they showed increasing variability through time (Figure 74). The LdHf, and in lesser manner the HdLf, scenarios shown a steady and increasing trajectories, while the MPA scenario showed high interanual fluctuations and a decreasing trend in the last 2 years of the simulation (Figure 74). These fluctuations and differences can also be explained by the expected food dynamics –i.e. the spatial management scenarios are inducing changes in the benthic ecosystem that in turn is altering their trophic structure and function.

Figure 74. Changes of the mean trophic level of the ecosystem under all spatial management scenarios in relation to the base case (BC status quo).

10.3.8 COMPARATIVE ASSESSMENT

The Table 47 list the summary of likely performace measures for selected groups in relation to the various metric and simulation scenarios. The colours in this table allows the quick visualisation of the best and worst case scenarios for a range of groups,where the red indicate the worst case, the green the best case, while the yellow the nil case scenario for that particular group and/or metric.

It is evident again that the overall values are largely small and evidence little variation across the range of measures and metrics (Table 47). These are consistent with the previous results and demonstrate again that the current levels of trawling in the overall modelled benthic ecosystem are low.

This comparison showed that the various spatial closures can act as best, nil and and worst case scenarios for any particular group or metric. An important result is that in general and in relation the the base case BC for example, the scenarios that have the greatest spatial closures (MPA and LdHf) have the best and worst responses across all measures and metrics (Table 48). In particular, and consistent with other results, the LdHf scenario seems to be the one that have the most negative and positive responses, followed by the MPA and HdLf scenarios (Table 48). This compartive assessment confirms that the level of closures will create the most responses across the whole benthic ecosystem, while the scenarios that have the least closures have no or relatively little responses in relation to the base case BC, (Table 48).

This comparison also shows that the coarser the metrics, like in the case of overall relative biomass for the whole tiger-endeavour fishing grounds are poor and little informative since they are too spatially coarse to detec any difference across scenarios (Table 48).

Table 47. Summary of the predicted values for 2016 for a range of performance measures for key example groups and metrics for all 6 spatial management scenarios. Red= worst case, Yellow = intermediate case, Green= best case scenario.

	Scenarios in 2016					
Performance measures	**BC**	**MPAs**	**LdHf**	**HdLf**	**LdLf**	**EoT70**
Median (5th,95th) percentiles of the EoT indicator in 2016 for Weipa and Tiger regions for:						
• Weipa – Large gastropod carnivore	0.68 (0.56,0.82)	0.75 (0.65,0.83)	0.68 (0.56,0.82)	0.68 (0.56,0.82)	0.68 (0.56,0.82)	0.71 (0.59,0.84)
• Weipa – Sessile epibenthos	0.82 (0.73,0.91)	0.84 (0.81,0.90)	0.82 (0.73,0.91)	0.82 (0.73,0.91)	0.82 (0.73,0.91)	0.84 (0.75,0.91)
• Tiger – Large gastropod carnivore	0.84 (0.77,0.88)	0.85 (0.80,0.89)	0.87 (0.81,0.90)	0.85 (0.78,0.89)	0.84 (0.77,0.88)	0.84 (0.77,0.88)
• Tiger – Sessile epibenthos	0.91 (0.87,0.94)	0.92 (0.90,0.94)	0.93 (0.90,0.95)	0.92 (0.88,0.94)	0.91 (0.88,0.94)	0.91 (0.88,0.94)
Relative median (5th,95th) percentiles of the average density in kg per hectare biomass in 2016 per scenario relative to the BC in Tiger region for:						
• Dugongs	0.5 (0.5, 0.5)	-0.02 (-0.04, -0.01)	0.01 (0.00, 0.01)	0.00 (0.00, 0.01)	0.00 (0.00, 0.00)	0.00 (-0.00, 0.00)
• Turtles	8.1 (8.1,8.1)	-0.00 (-0.01, 0.00)	-0.02 (-0.03 ,-0.02)	-0.00 (-0.00, 0.00)	-0.00 (-0.00, 0.00)	0.00 (0.00, 0.00)
• Sea snakes	1.9 (1.9, 1.9)	0.03 (0.01, 0.04)	-0.04 (-0.06, -0.03)	0.01 (0.01, 0.02)	-0.00 (-0.01,-0.00)	0.01 (-0.00, 0.02)
• Large sharks	20.9 (19.9,20. 1)	0.06 (0.02, 0.09)	0.10 (0.05, 0.13)	0.04 (0.03, 0.06)	-0.00 (-0.01, 0.01)	0.00 (-0.03, 0.02)
• Sawfishes	12.0 (12.0,12.0)	0.09 (0.05, 0.12)	0.11 (0.08, 0.15)	0.05 (0.03, 0.07)	-0.00 (-0.02, 0.01)	0.00 (-0.02, 0.03)
• Cephalopods	166.5 (166.3, 166.6)	0.00 (-0.01, 0.01)	-0.08 (-0.11, -0.06)	-0.01 (-0.02, -0.01)	-0.00 (-0.00, 0.00)	0.00 (-0.00, 0.01)
• Tiger prawn adult	13.4 (13.3, 13.6)	-0.09 (-0.25, 0.03)	0.42 (0.12, 0.78)	0.07 (-0.09, 0.20)	0.02 (-0.06, 0.12)	-0.00 (-0.03, 0.19)
• Lobsters	11.9 (11.8, 12.0)	0.24 (0.09, 0.37)	-0.32 (-0.44, -0.22)	-0.02 (-0.05, 0.01)	-0.02 (-0.04, -0.01)	0.03 (0.01, 0.09)
• Large gastropod carnivore	73.0 (72.7, 73.3)	0.01 (-0.04, 0.04)	-0.16 (-0.22, -0.12)	-0.10 (-0.12, -0.07)	0.00 (-0.00, 0.01)	-0.01 (-0.04, 0.02)
• Sessile epibenthos	6,349 (6,338, 6,355)	-0.01 (-0.02, 0.01)	0.07 (0.05, 0.09)	0.02 (0.01, 0.03)	0.00 (0.00, 0.01)	0.00 (-0.01, 0.01)
• Seagrass	204.9 (204.4, 205.1)	0.01 (-0.00, 0.05)	-0.01 (-0.04, 0.03)	-0.01 (-0.03, 0.02)	0.01 (-0.00, 0.02)	-0.00 (-0.01, 0.01)
Percentage change in total biomass in 2016 relative to 2008 for the overall Tiger region						
Median (5th,95th) percentiles of the	0.010 (0.006, 0.012)	0.010 (0.006, 0.012)	0.010 (0.006, 0.012)	0.010 (0.006, 0.012)	0.010 (0.006, 0.012)	0.010 (0.006, 0.012)
Mean TL – BC in 2016	0	–0.00004	0.0003	0.00005	0.00001	0.00001
Number 6-minute cells closed on Tiger region	0	211	129	79	53	49±29 ^{a}
Percentage of area closures on Tiger region	0	31.5%	19.3%	11.8%	7.9%	7.3±4.3% ^{a}
Median (5th,95th) percentiles of ffort	0 (0, 0)	1,752 (1,053, 6,475)	2,436 (1,774, 3,214)	1,531 (1,125, 1,764)	330 (181, 487)	223 (62, 434)
Median (±95%CI) number of groups at risk in	2±1	2±1	0	1±1	2±2	2±2
Percentage of 30 simulations where $\mu>\mu_{msm}$ in Tiger region for 2016 for:						
• Large sharks	36.7%	26.7%	0.0%	20.0%	30.0%	30.0%
• Rays	3.3%	0.0%	0.0%	0.0%	0.0%	0.0%

a= median ±95% CI

10.4 DISCUSSION

Evaluating the ecosystem effects of demersal trawling has attracted substantial research efforts in the past decades (Jennings and Kaiser. 1998, Hall 1999, Gislason et al. 2000, Collie et al. 2000, Thrush and Dayton 2002, Kaiser et al. 2006, Watling and Norse 2008, Trush and Dayton 2010, Thurstan et al. 2010). These efforts have been largely biased towards studies in the northern hemisphere and temperate and deep water ecosystems, where the impacts have been demonstrated to be substantial (e.g. Thrush et al. 1998, Cryer et al. 2002, Heath 2005, Heifetz et al 2009). The results of such studies have strongly influenced the perceptions of trawling impacts.

Similar studies elsewhere however have not occurred for tropical soft-sediment systems, normally associated with fishing for Peaneids and crustaceans species. The comparatively few studies conducted in tropical and sub-tropical ecosystems have been largely focus on bycatch assessments (e.g. Harris & Poiner 1990, Hendrickson and Griffin 1993; Brewer et al. 1998; Ortiz 1998; Robins et al. 1999; Stobutzki et al. 2001a,b). Foster (2009) made the interesting point that most of these impact studies are concentrated in developed countries, creating a strong knowledge bias to simile ecosystems in developing nations, where most trawling is conducted in tropical soft-sediments habitats.

Australia has led the assessment of ecosystem impacts of fishing in particular the impacts of trawling, with a range of studies in tropical sub-tropical (e.g. Moran and Stephenson 2000, Pitcher et al. 2000, Sobutski et al. 2001, Wassenberg eta al 2002, Sumpton et al. 2005, Courtney et al. 2006, Pitcher et al. 2009), temperate (e.g. Bax and Williams 2001, Tanner 2003, Williams et al. 2006, Svane et al. 2009) and deep ocean ecosystems (e.g. Koslow et al. 2001, Althaus et al. 2009, Williams et al. 2010). The findings of such studies has been varied, ranging from irreversible (Williams 2010), to strong (Svane et al 2009), and to less obvious and in cases undetectable impacts (Burridge et al. 2005). Despite this wide variations, trawling does have local and specific impacts, particularly when the fishing grounds overlap with with vulnerable biota and the impact assessments are done at the appropriate temporal and spatial scales (Pitcher et al. 2009). The significance of these works is that they have set scientific basis and ecological knowledge needed for this study.

Table 48. Comparative cost-benefit summary of the best–and-worst case scenarios for various performance measures against a set of likely management objectives for key functional groups.

Management Objectives	Performance measure	Best scenario	Worst scenario	Pro’s	Con’s	Comments
Sustainability of byproduct						
• Cephalopods	Relative biomass	none	LdHf, & HdLf	– no changes in biomass	– fewer catches – higher displaced effort – higher costs ($)	– MPA spatial design will be crucial to achieve NPF’s least effort displacement
• Lobsters (mostly bugs)	“	MPA	LdHf, Hd & LdLf	– higher biomass		
Protecting habitat-forming taxa						– Seagrass protected by historical closures – MPA more echinoids and sea turtles that less epibenthos
• Seagrasses	Relative biomass	LdHf & HdLf	LdHf, HdLf & EoT70	- higher biomass	– non–adaptive closures – more displaced effort	
• Sessile epibenthos	“	LdHf & HdLf	MPA	- higher biomass		
Reducing “at risk” species						– ERA need to be at smaller spatial scales (habitats or less) otherwise no taxa is detected at risk.
• Overall number	μ and # taxa at risk	LdHf & HdLf	LdLf & EoT70	– effective to remove & minimize taxa at risk	– need to close more fishing grounds – no clear differentiation, need ancillary criteria	
• Sharks	“	all	BC, LdLf, & EoT70	– highest risk		
• Rays	“	all	BC	– zero risk		
Protecting TE&P species						
• Turtles	Relative biomass	none	MPA	– less effort displaced	– “spotty” and small single closures.	– Closures will require post–hoc consolidation – MPA favour turtle predators
• Dugong	“	LdHf,	MPA	– higher biomass	– need to close more fishing grounds	– MPA favour dugong predators
• Sawfishes	“	MPA, LdHf, & HdLf	none	– higher biomass	– need to close more fishing grounds	– All closures favour sawfishes
• Sea snakes	“	MPA & HdLf	LdHf	– higher biomass	– need to close more fishing grounds	– Trawl baycach is the biggest effect, closures full protection
Biodiversity						
• Biomass (increase)	Relative biomass	—	—	—	—	– Measure too spatially coarse, no differentiation among scenarios
• Trophic level (increase)	Median trophic level	LdHf	MPA	– higher TL in trend to pristine (1970)	– need to close more fishing grounds	– MPA protects predators that decrease prawns and TE&P
Minimize effort displacement						
• Tiger fishing grounds	% closed	LdLf & EoT70	LdHf	– minimal closures	– adaptive data needs – poor performance in other measures	– Closures are not focussed on fishing but related to other measures (conservation or risk based)
• Displaced effort	Boat days	EoT70	LdHf			

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APPENDIX 11 VOYAGE SUMMARY RV SOUTHERN SURVEYOR SS03/2005

Biogeophysical characterization of northern Australia marine ecosystems: Assessing biophysical relations, ecosystem biodiversity, and surrogacy

11.1 ITINERARY

Departed Cairns, Queensland 1330 hrs Wednesday 23 February 2005

Arrived Weipa, Queensland 1000 hrs Monday 21 March 2005

11.2 PRINCIPAL INVESTIGATOR(S)

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11.3 SCIENTIFIC OBJECTIVES

The survey provided quantitative data for the development of knowledge and tools for managing the effects of prawn trawling on the benthic ecosystems and communities of the Northern Prawn Fishery. It characterised the impacts of trawling on benthic communities and key ecosystem processes in the Gulf of Carpentaria using a wide array of sampling equipment simultaneously on the RV *Southern Surveyor*.

11.4 VOYAGE OBJECTIVES

The survey design was done around a natural experiment (i.e. the existing human “manipulation” of different levels disturbance on the benthos) to test the effects of trawling on benthic biota and ecosystem processes. There were a total of 127 sampling stations. At each location we sampled in the day and night. The stations were distributed among three trawling intensity strata and day/night within three key habitat types (Regions). At each sampling station we: (1) sampled biota with a prawn sampling net and epibenthic sled; (2) collected surface sediments with a box-corer; (3) collected water samples with a CTD; and (4) measured currents with an underway ADCP. In addition, we will also deployed moored instruments to measure currents and sediment flux at two sites.

11.5 VOYAGE TRACK

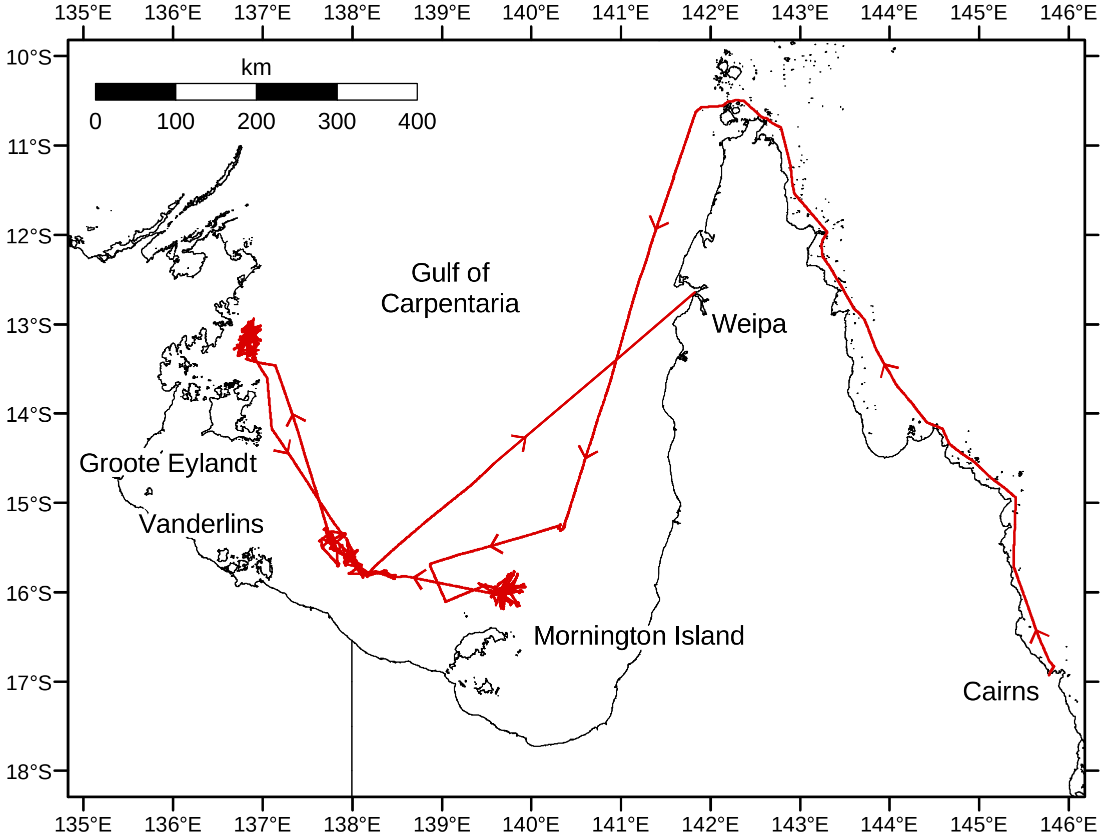


Figure 75. Cruise track for SS03/2005 departing Cairns on 23 February 2005 and arrivingWeipa 21 March 2005. The cruise track is approximately 7000 km long.

11.6 RESULTS

This cruise was a collaborative effort between CSIRO Marine Research (CMR), Geosciences Australia (GA) and the Department of Environment and Heritage – National Oceans Office (DEH/NOO). The cruise objectives were designed to satisfy a shared set of objectives. Further SS03/2005 would be followed by a complementary cruise (SS04/2005) manned by GA and CMR scientists and funded by GA and DEH/NOO.

11.6.1 GEOLOGICAL SAMPLING

The geological sampling program overall was highly successful, with 127 sites sampled, including the three target regions, reef areas, and current meter and BRUCE mooring sites. A total of 125 box cores were sub sampled using a 90 mm diameter minicore. An additional 23 bulk samples were taken from selected box cores to provide duplicate samples or where the sediments were unsuitable for coring.

Samples to measure suspended sediment concentrations were taken from 125 CTD samples. These will be analysed to provide information on sediment concentration and composition in the water column.

Nine (9) rock samples were also collected from benthic sled hauls, mainly in the region of the Vanderlins where there is much hard ground. Some of these samples appear to be reefal limestones, but require further analysis.

The box corer yielded excellent results with typical recoveries of 20 to 22 cm depth. Surface sediments were particularly well preserved in the bulk of samples. The sample sites encompass a range of habitat types, so will provide a good data set for investigating surrogates for benthic habitats.

The minicores collected will be analysed using a multi-sensor core logger which provides data on density and other sediment changes downcore. These data together with analysis of sediments and microfossil content will be used to interpret changes in sea floor conditions and biota through the last sea level rise. Surface minicore and bulk samples will be analysed for grain size, using sieve and laser techniques, to provide information on modern environments.

The diverse range of habitats sampled will provide an excellent dataset for testing relationships between geological and biological distributions, with sampling prior to and following Cyclone Ingrid allowing an unexpected opportunity to study the effects of these events on the sea floor environments.

The Geoscience Australia instrument frame, BRUCE (Benthic Research frame for Underwater sediment Concentration Experiments) was deployed on the top of the reef at the north-western end of the reef at location 15° 42.024' S, 138° 52.029' E in 26.0 m water depth. The Acoustic Doppler Current Profiler (Angus, son of BRUCE) mooring was deployed in the narrow tidal channel located mid-way between the BRUCE mooring site and Mornington Island at location 16° 5.706' S, 139° 2.418' E in 40.8 m water depth. Both deployments were left for recovery on the following leg of the voyage. These arrays are designed to measure currents, temperature, salinity, and turbidity.

Swath mapping was carried out during sampling activity within regions. Due to a problem with the sound velocity probe, sound velocity data was manually fed into the system. With this solution we obtained reasonable swath coverage, although it is relatively noisy.

11.6.2 BIOLOGICAL SAMPLING

CMR in collaboration with DEH/NOO had three principal objectives: (1) to characterise the benthic communities in the Gulf of Carpentaria; (2) measure key benthic ecosystem processes; and (3) measure the effects of trawling on the benthic communities and these ecosystem processes. These objectives satisfy long-term needs to: measure the biodiversity of the Gulf; understand the nutrient dynamics that underpin it and the prawn fishery it supports; and be able to assess the impacts of trawling. All these objectives contribute to both ensuring sustainability of the prawn fishery and also designing Marine Protected Areas (MPAs) in shallow tropical waters of northern Australia.

11.6.2.1 CHARACTERISE BENTHIC COMMUNITIES

The benthic community was characterised through a combination of geological and biological sampling using a box corer, benthic sled and a modified prawn trawl at 127 stations in three Regions (Mornington Island, Vanderlins Islands, and Groote Eylandt) (Table 51 has a full listing of the gear deployments (Operation Numbers) for all stations). The Regions were chosen to maximise the degree of environmental variation present in the Gulf of Carpentaria. The selection and delineation of the regions was done by analysing previous environmental and fisheries data. For the benthic characterisation objective the benthic sled was deployed 124 times. Live animals were sorted from the catches on-board and preserved by either freezing or storing in ethanol or formalin. These samples await further analysis by taxonomic experts from the Queensland and Northern Territory Museums. The trawl was deployed 155 times. Most catches were sorted at sea as time permitted. Of the sorted samples fish and invertebrates were identified to lowest taxonomic classification possible and enumerated. Over 75,000 fish from 258 species were identified and over 20,000 of those were weighed and measured. All invertebrates and unidentified fish await taxonomic investigation before further processing. The box corer was deployed 368 times for benthic community analysis as well as geological and biogeochemical analyses (see next sections). Benthic invertebrates were sieved from the sediment samples and preserved for further analysis.

11.6.2.2 CHARACTERISE KEY ECOSYSTEM PROCESSES

The principal ecosystem processes that were examined on this cruise were nitrogen cycling (nitrogen fixation and denitrification) and measurements of sediment oxygen demand. For nitrogen cycle studies a total of 166 samples were incubated on board (Table 49) and await further gas chromatograph and mass spectrometer analysis.

Sediment oxygen demand was measured during 12 h incubations on board. Preliminary analyses show high variability within and between regions. The Mornington Island Region appeared to have both the coarsest sediments and the lowest sediment oxygen demand (Figure 76). Both the Vanderlins and Groote Eylandt regions had finer sediments and higher oxygen demands indicating higher biological activity.

11.6.2.3 QUANTIFY THE EFFECTS OF COMMERCIAL TRAWLING

To examine the effects of commercial trawling the 42 sample locations within each geographic region were allocated randomly across three levels of trawl intensity. Locations were adjusted where necessary to eliminate extreme clumping or avoid untrawlable ground. The trawl intensity was determined by analysing AFMA's Vessel Monitoring System (VMS) data for the past 4 years (Figure 77 is an example of sample allocation for the Mornington Region). Sampling was also divided equally between daytime (0700 to 1700 hrs) and nighttime (1900 to 0500 hrs) to both capture day/night variability and avoid animals that were active during dawn and dusk. Dawn and dusk periods were also used for additional trawl samples to examine feeding behaviour of the trawled fishes. The same samples collected for the benthic characterisation objective will be used to address this objective with the added refinement of incorporating the trawl intensity classification on the data.

11.7 VOYAGE NARRATIVE

The R/V *Southern Surveyor* left Cairns at approximately1330 hrs and off-loaded the pilot an hour later. We steamed north, inside the Great Barrier reef with 17 knot south-easterly winds and a gentle following sea. The first 3 d were spent steaming to our first sampling station in the south-eastern Gulf of Carpentaria. During the steam sampling equipment, instrument arrays and laboratories were set up. This was accompanied by a safety induction and vessel tour, a fire drill, an induction by the Cruise Manager, and a series of Job Hazard Analysis (JHA) meetings and Tool Box meetings for new crew members.

We arrived at our first sampling station around 0630 on 26/2/05. Here, adjacent to the reef GA had found on its last cruise to the Gulf (SS04/2003, May 2003), we deployed our soft sediment sampling gear for the first time. Hydraulic pump failures prevented us from completing all the planned sampling so we departed around 1900 hrs to the "Bruce" and ADCP mooring stations west of Mornington Island.

"Bruce" was deployed 0700 hrs 27/2/05 and the ADCP ("Angus", son of Bruce) around 1140 hrs. Intermittent hydraulic problems prevented the full suite of samples from taken near these two mooring sites. At 1820 hrs we started our first series of samples in our Regional surveys (Station 7, Mornington Region). By day 6 (28/2/05) the hydraulic problems had been resolved and the ship was getting into a rhythm of CTD profiling, box corer sampling, dredging, trawling and sample handling.

We were starting to achieve seven stations per day which was our target. With the exception of two small intense squalls the weather up until now was incredibly calm. And the Cruise Leader had a surprise Birthday Party!

Day 7, Tuesday 1 March 2005 will be remembered as the attack of the heart urchins (*Maretia planulata*). Approximately 3 to 5 t were caught in the trawl, which couldn't be brought on board; the cod-end was torn and it took ca. 3 h to replace the net. Their abundance, based on numbers in the box corer samples, was estimated at ca. 100/m^2. These high abundances were a feature of a whole suite of stations north-west of Mornington Island where trawl durations and station locations had to be modified. Wayne Rochester's adaptive sampling protocol was coming into its own. We also started sending the web-based *"Ship to shore"* articles about activities on board to be mounted on the CMR and National Facility web pages. We completed sampling in Region 1 by 1800 hrs on Day 12 (6/3/05) and started sampling in Region 2 at ca. 2000 hrs. Weather was still incredibly kind – you couldn't see the horizon except for the point of the sea-sky reflection – and the Captain was predicting we'd pay for it later (How prescient he was).

The Mornington Region is characterised by a lot of untrawlable hard bottom (Figure 77) and is characterised by a lot of soft sediments and sponges. The adaptive sampling protocol associated with depth sounder interpretation was used extensively to find suitable sampling sites within each of the three trawl intensities. At 0300 hrs on Day 17 (11/3/05) the Captain decided to take evasive action as Cyclone Ingrid had crossed Cape York and moved into the Gulf and was predicted to move south-west. We suspended sampling and steamed for our third sampling region, north of Groote Eylandt.

Sampling re-commenced at 1600 hrs. Winds were increasing (25 to 30 kts) and Ingrid was tracking north-westerly across the Gulf. We ceased sampling about 0200 when winds reached 40 to 50 kts as gear could not longer be safely deployed. Winds eased by dawn and sampling resumed by 0700 hrs. Seas were mixed, and there was a lot of sediment in the water column. It was too rough to weigh animals on the electronic balances, so we started boxing and freezing the fish catches for later analysis. We continued to defer sample processing for the next 2 d to make up for lost time and were able to make up all the time lost from steaming and suspending sampling by completing 10 to 11 stations per day. Later that day the fumes of the Sewage Treatment Plant started to infuse the ship, especially below decks. A series of not very effective remedial engineering and operational actions were tried, but were not very effective. The smell remained for the rest of the voyage.

Sampling in the Groote Region finished on Day 21 (15/3/05) and after a 15 h steam re-commenced at 1200 hrs Day 22 at the unfinished stations near the Vanderlins. The weather continued to be rainy and squally but sampling went on routinely. "Kenny the Kingfisher" landed exhausted on deck and immediately became the ship's mascot. He/she lived in a box in the Wet lab and was hand-fed round the clock. Kenny was released 3 d later off Weipa. Sampling was completed ca 0100 on Day 26 (20/3/05) and we commenced the 30 h steam to Weipa. A farewell barbeque, and a Customs flyover were the highlights, while we broke down the sampling equipment and cleaned up the laboratories and accommodation.

The pilot came on board at the Weipa fairway buoy at 0830 on Day 27 (21/3/05) and we were docked at Evans Landing Weipa at 1000 hrs. Offloading was hampered by fueling operations at Evans Landing and another ship on the Humbug Wharf. The ship was moved to Humbug at 1700 hrs and offloading completed ca. 1100 hrs on day 28 (22/3/05).

11.8 SUMMARY

11.8.1 SCIENTIFIC ACHIEVEMENTS

All the objectives of the cruise were achieved – see earlier Results Section for more detail. The adaptive sampling protocol allowed us to sample all the planned sites in three Regions at three trawl intensities evenly distributed over day and night samples. Additional dawn/dusk trawl samples were taken for gut-content analysis. The combined sampling between GA and CMR centred around the CTD and box corer was also very successful. Samples await further laboratory analysis.

11.8.2 SUITABILITY OF SHIP AND EQUIPMENT

The RV *Southern Surveyor* has potentially an impressive sampling capacity. However, during the early stages of the voyage continuing problems with the hydraulic systems on the vessel were extremely frustrating to both the crew and scientific party. In some cases the faults were not just an inconvenience they were potentially dangerous and cruise limiting. In discussions with crew it seems that some of these problems have dogged the ship since the major refit, indeed may have been caused by it. The trouble shooting ability and implementation of running repairs was commendable, but the hydraulic system needs a thorough going over.

Some CTD systems and equipment requested in the early Voyage Proposal were not in place. Luckily, instruments were on board and could be installed underway. It appears that the comprehensive equipment lists that were filled in at the proposal stage were not passed to Ship Support staff, which relies on the web-based Voyage Plan. Equally the Voyage Plan prepared by us was not as comprehensive as the earlier equipment lists. There is either a breakdown in communication and/or unnecessary and incomplete duplication of information between these two documents.

On this cruise there was an unusual amount of turnover of ships officers and crew. While all professional mariners, the RV *Southern Surveyor* is a specialised vessel with very specialised equipment and procedures. A great deal of time was spent familiarizing and training crew for what are otherwise routine duties, which of course slowed the early stages of sample and data acquisition.

11.8.3 OVERALL IMPRESSIONS

The RV Southern Surveyor continues to be a vital asset to Australia’s sea-going marine research. It is manned by a professional, helpful and accommodating maritime crew. There is a growing frustration however, that the asset is not being maintained to the level required. Makeshift running repairs will always be required, but seem to be the order of the day rather than thorough maintenance and replacement schedules that minimise the frequency of breakdowns.

However in spite of these frustrations my lasting impression is an excellent crew, very professional support staff, and a capable, but aging, ship. The cruise was demanding for ship, crew and scientists; we had some gear and weather problems (Cyclone Ingrid) but met all our Objectives. Thanks and well done to all.

11.9 PERSONNEL

Scientific crew

Name	Organisation	Role
Pamela Brodie	Southern Surveyor	Computer/data manager, Cruise Manager
Neale Johnston	Southern Surveyor	Hydrochemist
Lindsay McDonald	Southern Surveyor	Electronics Technician
Rodrigo Bustamante	CMR	Principal Investigator, benthic infauna
Peter Rothlisberg	CMR	Chief Scientist, benthic biogeochemistry
Michele Burford	Griffith University	Primary productivity, benthic biogeochemistry
Vicki Passlow	Geoscience Australia	Sediment sampling
Alix Post	Geoscience Australia	Sediment sampling
John Salini	CMR	Fish biology & taxonomy
Ted Wassenberg	CMR	Invertebrate biology & taxonomy
Tom Okey	CMR	Benthic infauna
Wayne Rochester	CMR	Sampling design & data analysis
Gary Fry	CMR	Fish biology & taxonomy
Mark Tonks	CMR	Invertebrate biology & taxonomy

Ship's crew

Ian Taylor	*Master*
Samantha Durnian	Chief Officer
Robert Ferries	Second Officer, Fishing Master
John Morton	Chief Engineer
Jim Hickie	First Engineer
Chris Heap	Second Engineer
Malcolm McDougall	Bosun
Graham McDougall	Day work Integrated Rating (IR)
Mark McRae	IR
Patrick Chamberlain	IR
Tony van Rooy	IR
Phillip French	Greaser
Charmaine Aylett	Chief Stewart
Andy Goss	Chief Cook
Adam Edwards	Second Cook

11.10 ACKNOWLEDGEMENTS:

- Ian Taylor (Master) and crew, especially John Morton (Chief Engineer) and staff for battling and beating the early hydraulic problems
- Doug Chetwynd (CMR) for early data base development under limited time frames and Wayne Rochester (CMR) for keeping it alive and modifying it underway
- Ship Support crew (Pamela Brodie, Lindsay MacDonald and Neale Johnston) were extremely professional and unflappable, all needs were met and problems solved often on the run
- Ted Wassenberg and John Salini for logistical planning and watch leading. Their experience and professionalism ensured a successful outcome.
- Vicki Passlow and Alix Post (GA) would also like to acknowledge the assistance of the crew in setting up and deploying BRUCE and the ADCP, as well as in the sampling program.

Peter C Rothlisberg

Chief Scientist.

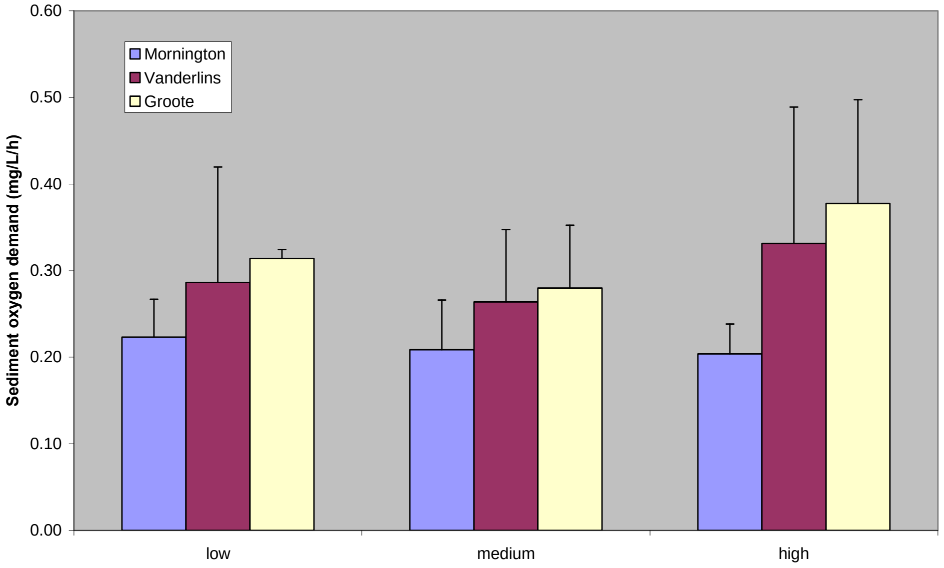


Figure 76. Measurements of sediment oxygen demand at three Regions and three Levels of trawl intensity.

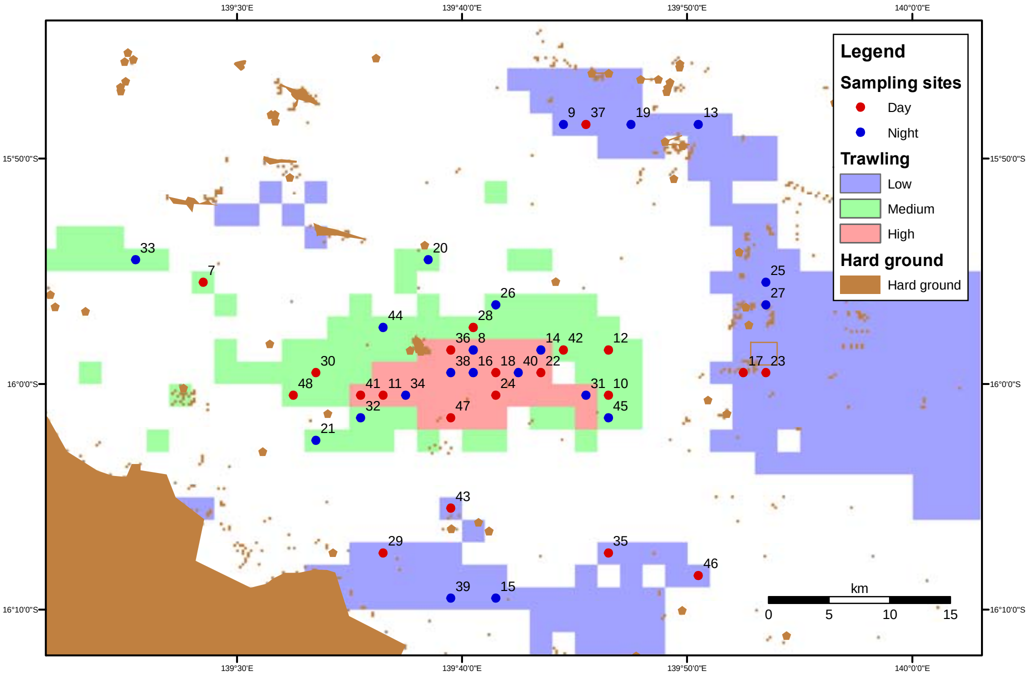


Figure 77. Map of sampling locations in Mornington Region showing randomallocation of day and night stations with respect to high, medium, and low trawling intensity.

Table 49. The number of nitrogen fixation, denitrification and sediment oxygen samples that were incubated on board.

Parameter	Region	High	Medium	Low	Total
N fixation – water column	Mornington	0	0	12	12
	Vanderlins	12	0	0	12
	Groote	0	0	0	0
Total					**24**
N fixation – sediment	Mornington	5	0	5	10
	Vanderlins	4	4	0	8
	Groote	2	2	4	8
Total					**26**
15N denitrification	Mornington	12	12	20	44
	Vanderlins	12	12	12	36
	Groote	12	12	12	36
Total					**116**
Sediment oxygen demand	Mornington	8	8	6	22
	Vanderlins	8	8	8	24
	Groote	8	8	8	24
Total					**70**

Table 50. The number of benthic sampling stations stratified by Region and Trawl Intensity. Equal numbers of daytime (7) and nighttime (7) samples were undertaken in each strata*.

Region	*Trawl intensity*	*Number of stations*
Mornington	High	14
	Medium	14
	Low	14
Vanderlins	High	14
	Medium	12*
	Low	14
Groote	High	14
	Medium	14
	Low	14

*two daylight samples were selectively dropped from the design (medium intensity stratum at the Vanderlin Islands) in order to complete the cruise on schedule

Table 51. Overall operation numbers for each gear type deployments and each sampling stations (sites).

Station	Sled	Trawl	Dawn	Dusk	CTD	Box core	Failed deployments #s (type)
1	3	5					4 (trawl), 6 (CTD)
3	9						
5					10	11, 12	
6						19, 20	
7	21	22			23	24, 25, 28	26 (grab), 27 (grab)
8	29	30			31	32, 33, 35, 36	34 (grab)
9	37	38			39	40, 42, 43	41 (grab)
10	47	48	46		49	50, 51, 52	44 (sled)
11	53	54			55	56, 57, 58	
12	59	60			61	62, 63	
13	64	70		69	65	66, 67, 68	
14	71	72			73	74, 75, 77	76 (grab)
15	78	79			80	82, 83, 84	81 (grab)
16	85	86	87			89, 90	
17	91	92			93	95, 96, 97	94 (grab)
18	98	99			100	101, 102	
19	104	108			105	106, 107	103 (trawl)
20	109	110			111	112, 113	
21	114	115	116		117	118, 119, 120, 121, 123	122 (grab)
22	124	125			126	128, 129, 130, 131, 132	127 (grab)
23	133	134			135	136, 141, 142	137 (grab), 138 (grab), 139 (grab), 140 (grab)
24	143	144		150	145	146, 147, 148	149 (grab)
25	151	152			153	155, 156, 157	154 (grab)
26	158	159			160	161, 162, 164	163 (grab)
27	165	166	171		167	169, 170	168 (grab)
28	174	173			175	176, 177, 179	172 (sled), 178 (grab)
29	180	181			182	183, 184	
30	185	186			187	188, 189	
31	190	193		191	192	194, 195, 196	
32	197	198			199	200, 201, 202	
33	203	204			205	206, 207	
34	208	209	210		211	213, 214, 215, 216, 217	212 (grab)
35	218	219			220	221, 222, 223	
36	226	225			227	228, 229	224 (sled)
37	230	231		232	233	234, 235, 236	
38	237	243			239	240, 241, 242	238 (trawl)
39	244	245			246	247, 249, 250	248 (grab)
40	251	252	256		253	254, 255	
41	257	258			259	260, 261	
42	262	263			264	265, 266, 267	
43	268	269			270	275, 276, 277	271 (grab), 272 (grab), 273 (grab), 274 (grab)
44	282	284		283	278	279, 280, 281	
45	285	286			287	288, 289, 290	
46	292	293	291		294	295, 296, 298, 301, 302	297 (grab), 299 (grab), 300 (grab)
47	303	304			305	306, 307, 308	
48	309	310			311	312, 313	
49	314	315			316	317, 319, 321, 322	318 (grab), 320 (grab)
50	323	324			325	327, 328, 329	326 (grab)
51	332	330	331		333	334, 335, 336, 337, 338, 339, 340, 341	
52	342	343			344	345, 346, 347	

53	348	349			350	351, 352	
54	353	356		355	354	357, 358, 359	
55	360	361			362	363, 364, 365	
56	366	367			368	369, 370, 371	
57	377	378	372		373	374, 375, 376	
58	379	380			381	382, 383, 384	
59	385	386			387	388, 389	
60	391	390		392	393	394, 395, 396	
61	397	398			399	400, 401, 402	
62	403	404	412		405	406, 407, 409	408 (grab), 410 (grab), 411 (grab)
63	413	414			415	416, 417, 418	
64	419	420			421	423, 424	422 (grab)
65	425	426			427	428, 429	
66	431	436		432	430	433, 434, 435	
67	437	438			439	440, 441, 442	
68	443	444			445	446, 447	
69	450	449	448		451	452, 456, 457, 458	453 (grab), 454 (grab), 455 (grab)
70	459	460			461	462, 463, 464	
71	465	466			467	473, 474	468 (grab), 469 (grab), 470 (grab), 471 (grab), 472 (grab)
72	479	480			475	476, 477, 478	
73	481	482			483	484, 485, 486	
74	487	488			489	490, 491	
75	493	492		498	494	495, 496, 497	
76	499	500			501	502, 503, 504	
77	505	506			528	529, 530, 531	
78	507	508			509	510, 511	
79	512	513			514	515, 516	
80	517	518			519	520, 521	
81	522	523			524	525, 526, 527	
82	532	533	555		534	535, 536, 554	537 (grab)
83	538	539			540	541, 542, 543	
84	544	545			546	547, 548	
85	549	550			551	552, 553	
86	557	556			558	559, 560, 561	
87	562	563			564	565, 566	
88	567	568			569	570, 571	
89	572	573			574	575, 576	
90	580	582		581	577	578, 579	
91	583	584			585	586, 587, 588	
92	589	590			591	593, 596, 597	592 (grab), 594 (grab), 595 (grab)
93	598	599			600	601, 602	
94	609	608			603	604, 605, 606, 607	
95	610	611			612	613, 614, 615	
96	616	617			618	619, 620	
97	622	621		624	623	625, 626, 627, 629	628 (grab)
98	630	631			632	633, 634, 635	
99	636	637			638	639, 640, 641	
100	642	643			644	645, 646	
101	651	652	647		648	649, 650	
102	653	654			655	657, 658, 659	656 (grab)
103	660	661			662	663, 664, 665	
104	666	667			668	669, 670	
105	672	674		673	671	675, 676, 677, 678	
106	679	680			681	682, 684, 685	683 (grab)
107	686	687			688	690, 691, 692	689 (grab)

108	693	694			695	696, 697	
109	704	705	703		698	699, 700, 701, 702	
110	707	708			709	710, 711, 712, 713	706 (trawl)
111	714	715			716	717, 718	
112	724	725			719	720, 721, 722, 723	
113	726	727			728	729, 730	
114	731	732			733	734, 735, 736, 737	
115	738	739			740	741, 742	
116	744	743		745	746	747, 748, 749, 750	
117	751	752			753	754, 755, 756, 757, 758, 759, 760, 761	
118	762	763			764	765, 766, 767	
119	768	774			770	771, 772, 773	
120	779	780	775		776	777, 778	
121	781	782			783	786, 787, 788, 789, 790, 793, 794	784 (grab), 785 (grab), 791 (grab), 792 (grab)
122	795	797			798	799, 800, 801, 802	796 (trawl)
123	804	805		803	806	807, 808, 809, 811	810 (grab)
124	812	813			814	815, 816, 817, 818, 819, 820, 821, 822, 823, 824, 825	
125	826	827			828	829, 830	
126	839	840	831		832	833, 834, 835, 836, 837, 838	
127	841	842			843	844, 846, 847	845 (grab)
128	848	849			850	851, 853	852 (grab)
129	858	859		857	854	855, 856	
130	860	861			862	863, 864, 865	
Totals	**126**	**125**	**16**	**15**	**124**	**380**	**66**

APPENDIX 12 ECOSYSTEM CHARACTERISATION OF THE NORTH MARINE REGION AND SPATIAL PREDICTIONS OF ITS TROPHIC FUNCTIONAL GROUP

The contents of this appendix have been extracted from sections of the report by Rochester et al. (2007) and its digital complementary sections available at CMAR's http://www.marine.csiro.au/datacentre/ext_docs/mbp_north/characterisation/index.html .
The work included below was conducted largely by team members of this project, but working on a parallel and complementary project funded by DEWHA to support the Marine Bioregional Planning process -in this case the North Marine Region (NMR), that includes most of the Northern Prawn Fisheries and research focus of the project.

12.1 OVERVIEW

This region comprises that part of the Australian Exclusive Economic Zone from the WA–NT border (as defined by the Petroleum (Submerged Lands) Act 1967) to, but excluding, Torres Strait, stopping at the western boundary of the combined Torres Strait Protected Zone and the 'outside but near' area (Figure 78).

The NMR includes all, or part, of four provincial bioregions: all of 'Timor transition', most of 'Northern IMCRA province' and parts of 'Northwest IMCRA transition' and 'Northeast IMCRA transition' (DEH 2005, Heap et al. 2005). It covers much of the ecologically defined North Large Marine Domain (Rothlisberg et al. 2005). Most of the region is on the continental shelf.

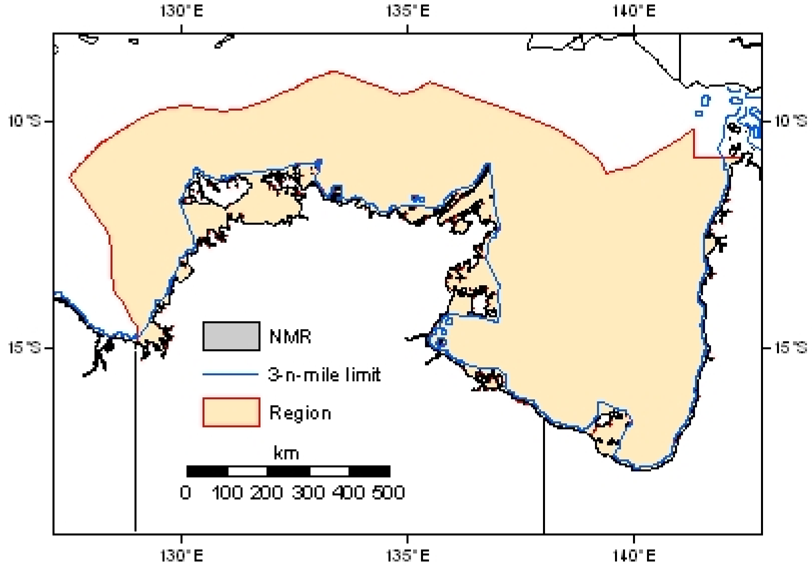


Figure 78. Depiction of the North Marine Region (NMR)

12.2 GEOMORPHOLOGY

The shelf areas are divided among three geomorphic provinces: Gulf of Carpentaria, Arafura Shelf and Sahul Shelf (Harris et al. 2005). All three of these provinces were drowned less than 18,000 years ago (Harris et al. 2005). The continental shelf is continuous between Australia and Papua New Guinea, and formed an emergent land bridge during the last ice age (Harris et al. 2005). The relative scarcity of features in the east of the NMR contrasts with the more complex patterns of banks and valleys in the west and slopes and canyons in the north (Figure 79). Much of the region is characterised by relatively flat terrain and soft sediment. The sediments tend to be sandy in coastal areas, though muddier at some river mouths, and muddy in places such as the north-western Gulf of Carpentaria, shelf areas of the western Arafura Sea, and offshore on the Sahul Shelf.

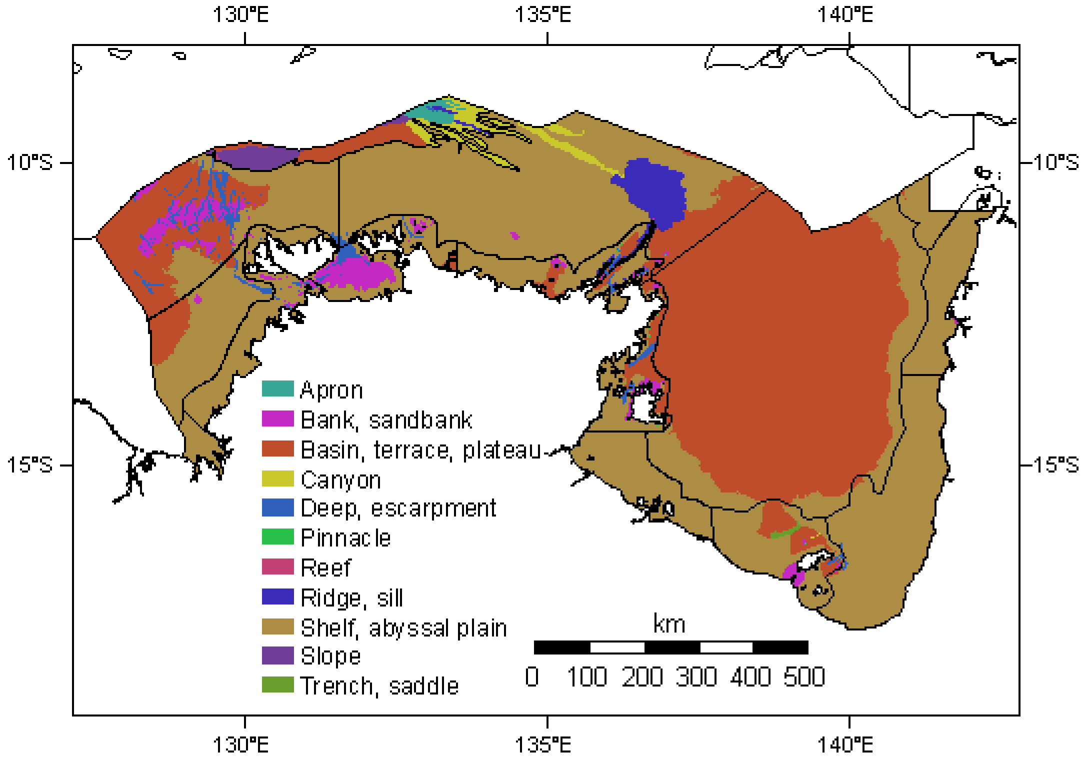


Figure 79. Geomorphic units of the North Marine Region (Harris et al. 2005)

12.3 CLIMATE

The climate features a summer monsoon. From October to March the winds are predominantly northerly or north-westerly and variable in intensity. From April to September the winds are stronger and more consistent south-easterly trade winds (Figure 80). Rainfall is highly seasonal, falling mostly in the summer monsoon, and is generated by monsoonal thunderstorms and tropical cyclones (Linacre and Hobbs 1977). Year-to-year variation of rainfall, and resulting river inflow, is high due to the El Niño Southern Oscillation and the erratic behaviour of cyclones (Hamilton and Gehrke 2005). Air temperatures are warm and show limited seasonal variation (Linacre and Hobbs 1977).

Cyclones affect most regions with a frequency of one per 1–2 years, mostly from December to April. The highest frequencies are in the eastern Gulf of Carpentaria (Thackway and Cresswell 1998, National Oceans Office 2003). Cyclones are the main cause of seabed disturbance, sediment movement and storm surges (National Oceans Office 2003). They can cause localised mixing of the normally stratified summer water column (P. C. Rothlisberg pers. comm.), and can cause intense rainfall and run-off. In shallow areas (e.g. 30 m) they measurably disrupt benthic species (Haywood et al. 2005).

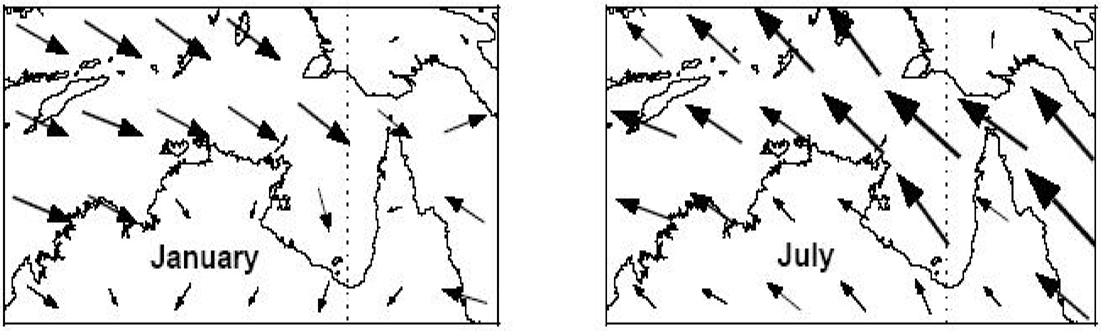


Figure 80. Seasonal wind vectors at 10 m above sea level in summer (January) and winter (July). Source: Rothlisberg et al. 2005.

12.4 WATER PROPERTIES

During the summer wet season (November–March), sea-surface temperatures are high (28–31°C) across the NMR, except near Torres Strait. The water is vertically mixed in shallow regions (e.g. coastal Gulf of Carpentaria, Torres Strait and Joseph Bonaparte Gulf), and stratified in deeper regions due to surface heating. In winter, temperatures are typically 5°C lower, and the water tends to be vertically mixed everywhere by tides, trade winds and surface cooling (Rothlisberg et al. 2005).

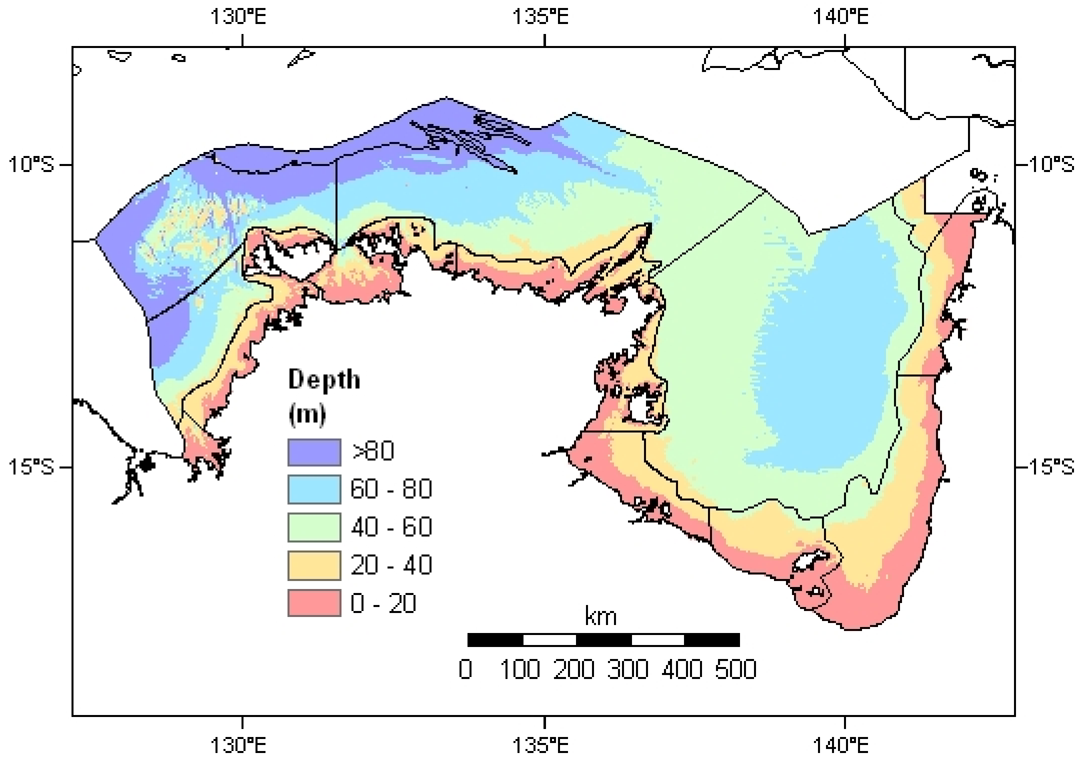
In coastal areas, salinity is affected by evaporation, rainfall and river inflow. Low salinity due to river inflow occurs on the west and east of the Gulf of Carpentaria. High salinity due to greater relative dominance of evaporation occurs in the southern Gulf of Carpentaria and Joseph Bonaparte Gulf (Rothlisberg et al. 2005). Offshore, salinity is affected by ocean currents, such as the Indonesian Throughflow, which inputs low salinity water to the north-west of the NMR (Rothlisberg et al. 2005).

Near the surface, dissolved oxygen remains near equilibrium with the atmosphere, and the concentration depends mainly on temperature (increasing as temperature decreases). Moving down through the water column, oxygen tends to be depleted, but the actual concentration may be affected by ocean currents and vertical mixing (Rothlisberg et al. 2005).Nitrate and phosphate concentrations are low throughout the NMR (Rothlisberg et al. 2005). Bottom concentrations are also low in shallow regions, but higher in deeper regions (See characterisation maps below). Silicate concentrations tend to reflect local riverine inputs, and are relatively high in coastal Gulf of Carpentaria (Rothlisberg et al. 2005). Bottom concentrations again tend to be higher in deeper regions (See characterisation maps below).

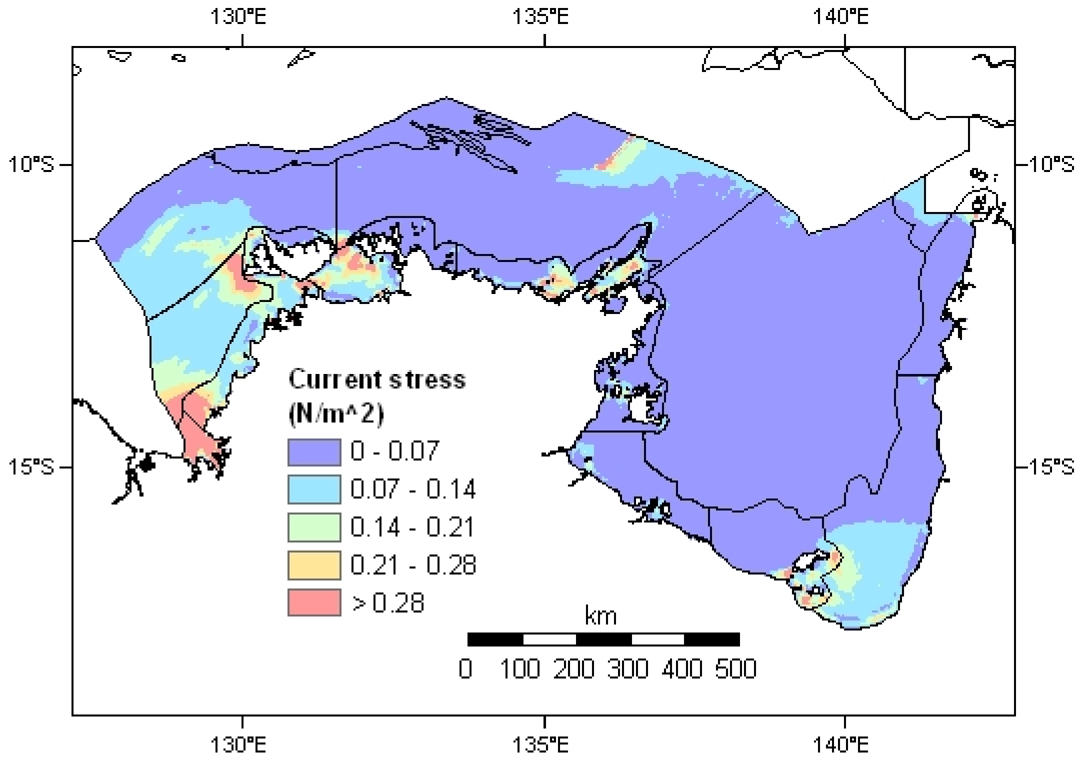
12.5 ENVIRONMENTAL ATTRIBUTE MAPS

This section presents characterisation maps of benthic environmental attributes for the NMR. The distribution of attribute values in each meso-scale bioregion is presented. The data sources were as follows: bathymetry, 'Australian bathymetry and topography grid, June 2005', Geoscience Australia (Webster and Petkovic 2005); current stress, 'Hydrodynamics model of the Northern Prawn Fishery', CSIRO (Hill et al. 2002); sediment, 'Sedimentary features of the EEZ (July 2004)', Geoscience Australia (Passlow et al. 2004); hydrography, 'CSIRO Atlas of Regional Seas', CSIRO (Ridgway et al. 2002, Hill et al. 2002).

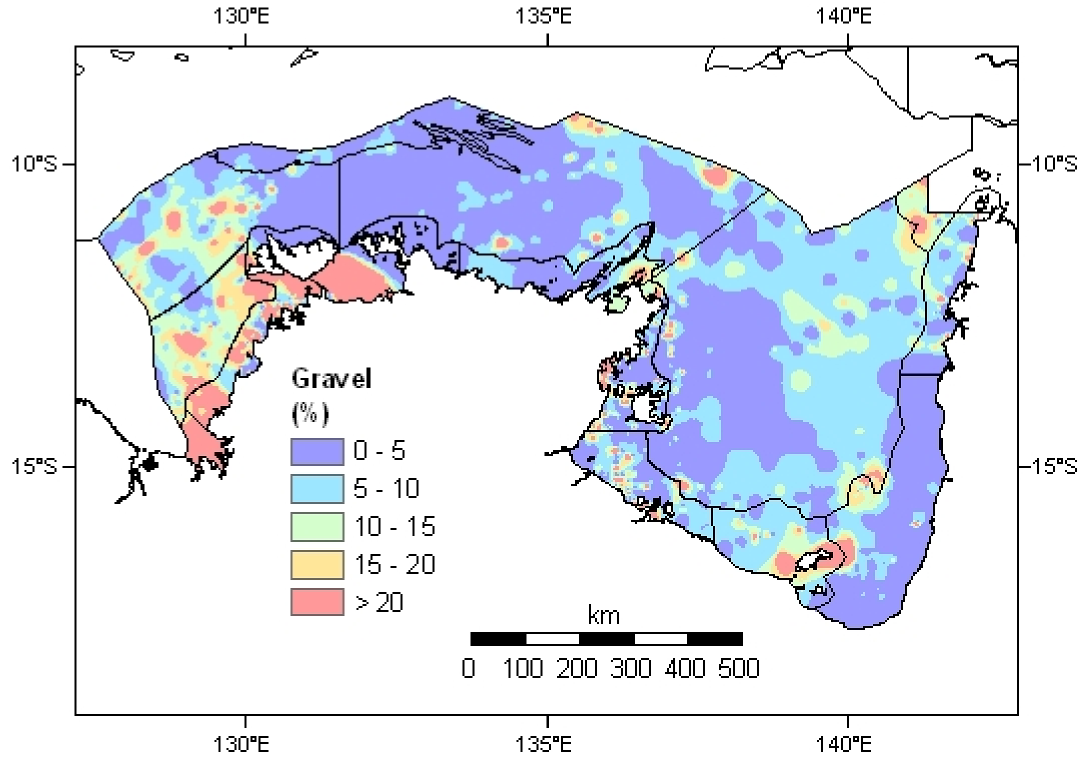
12.5.1 BATHYMETRY (M)



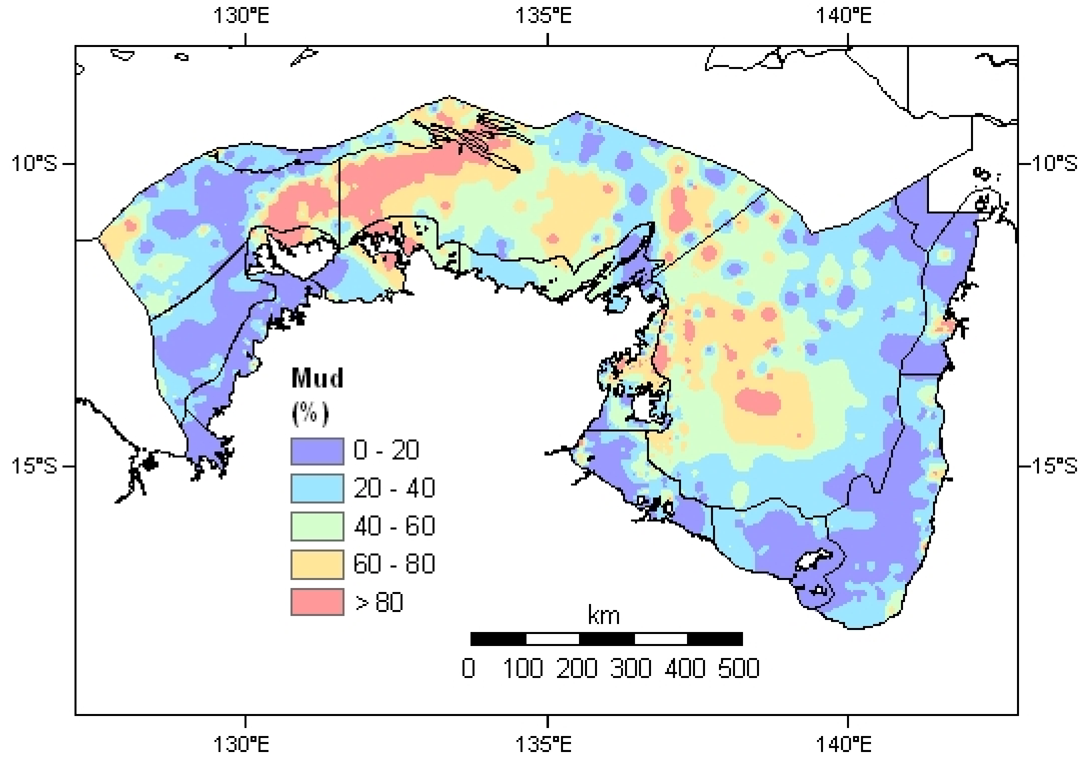
12.5.2 MEDIAN BOTTOM STRESS (N/M^2)



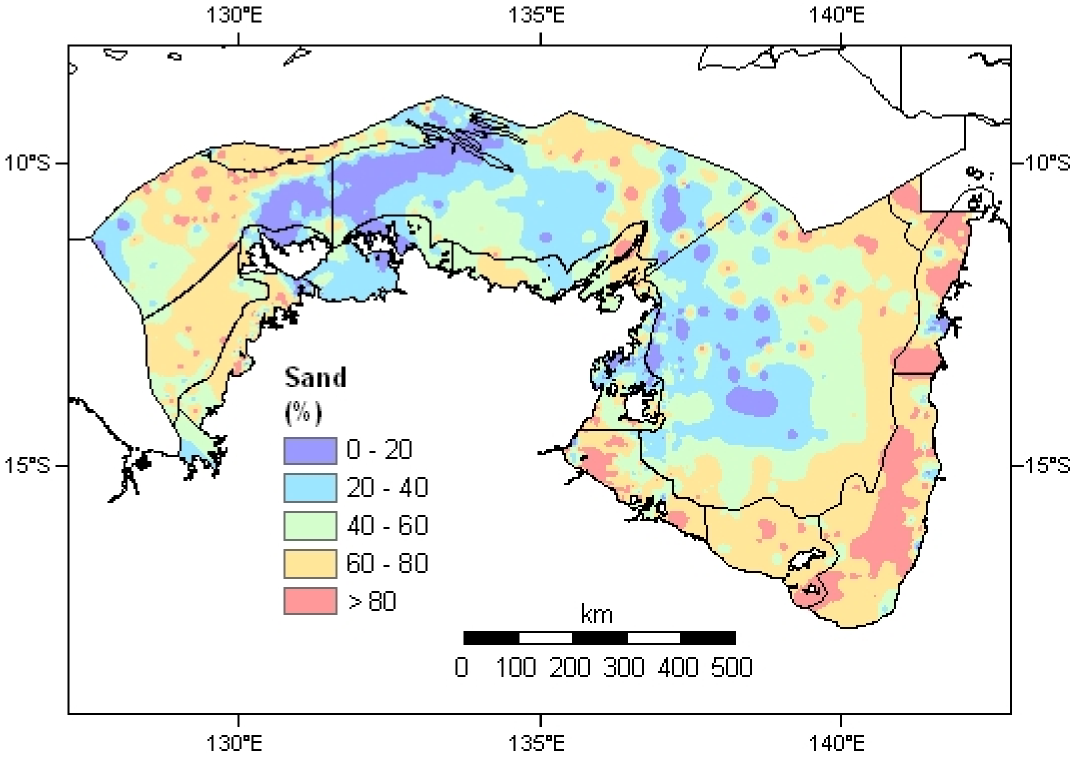
12.5.3 SEDIMENT GRAVEL PERCENTAGE



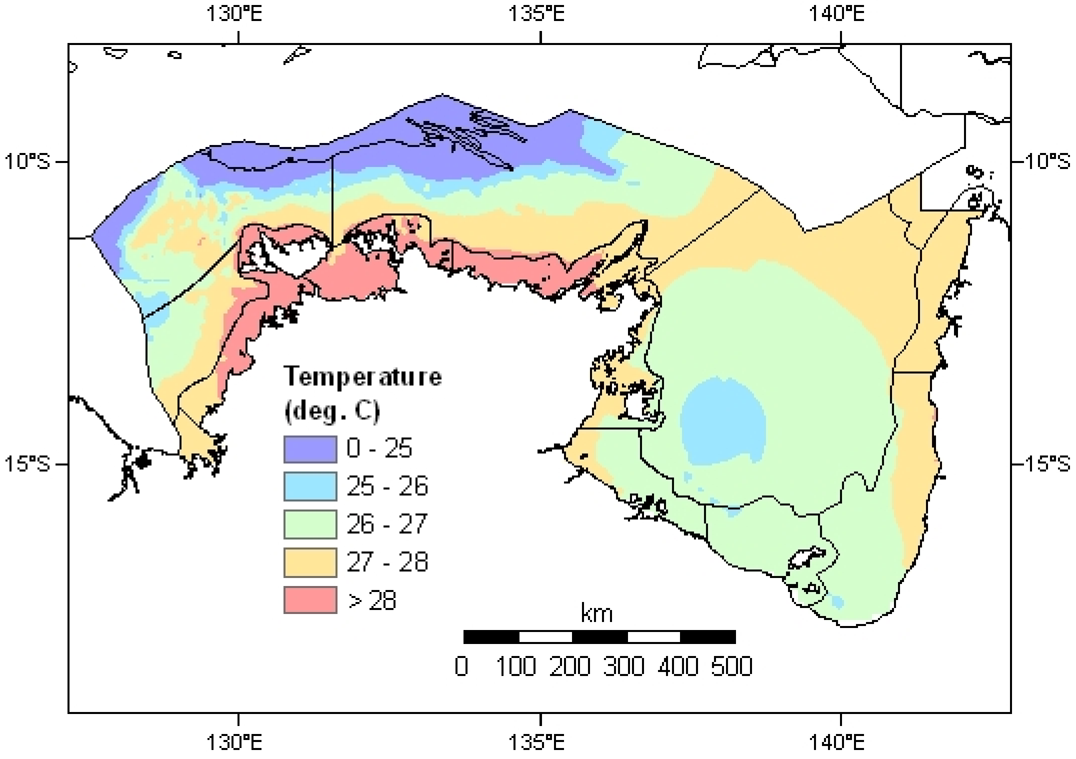
12.5.4 SEDIMENT MUD PERCENTAGE



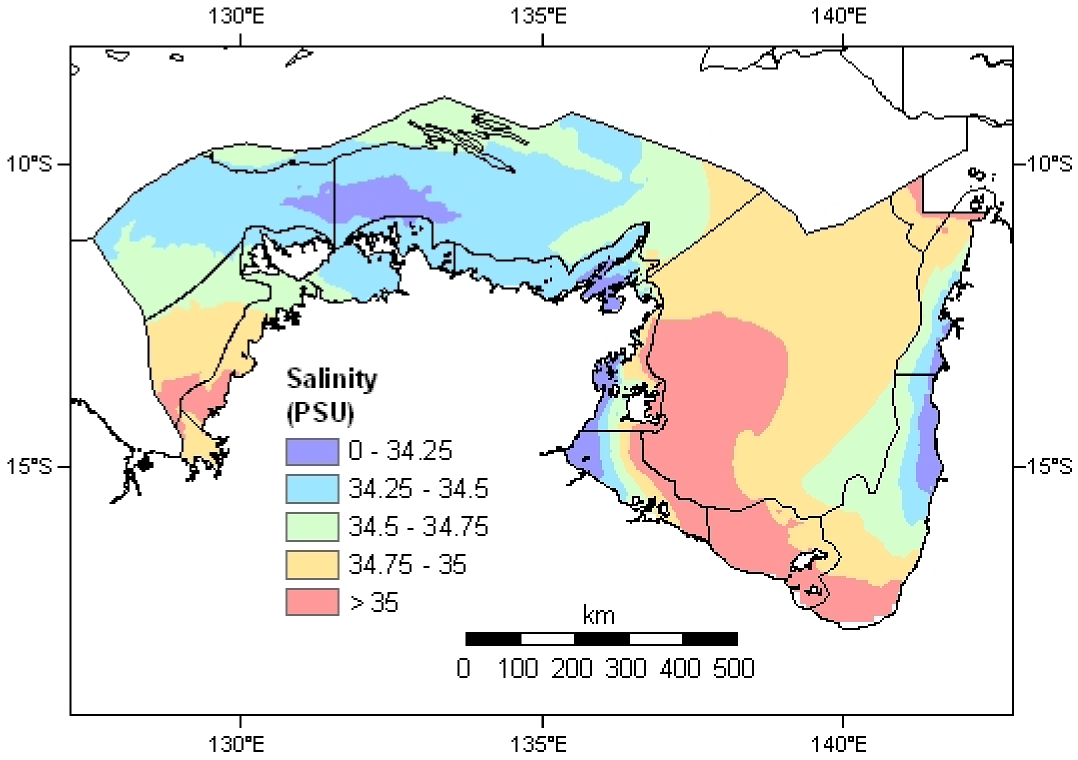
12.5.5 SEDIMENT SAND PERCENTAGE



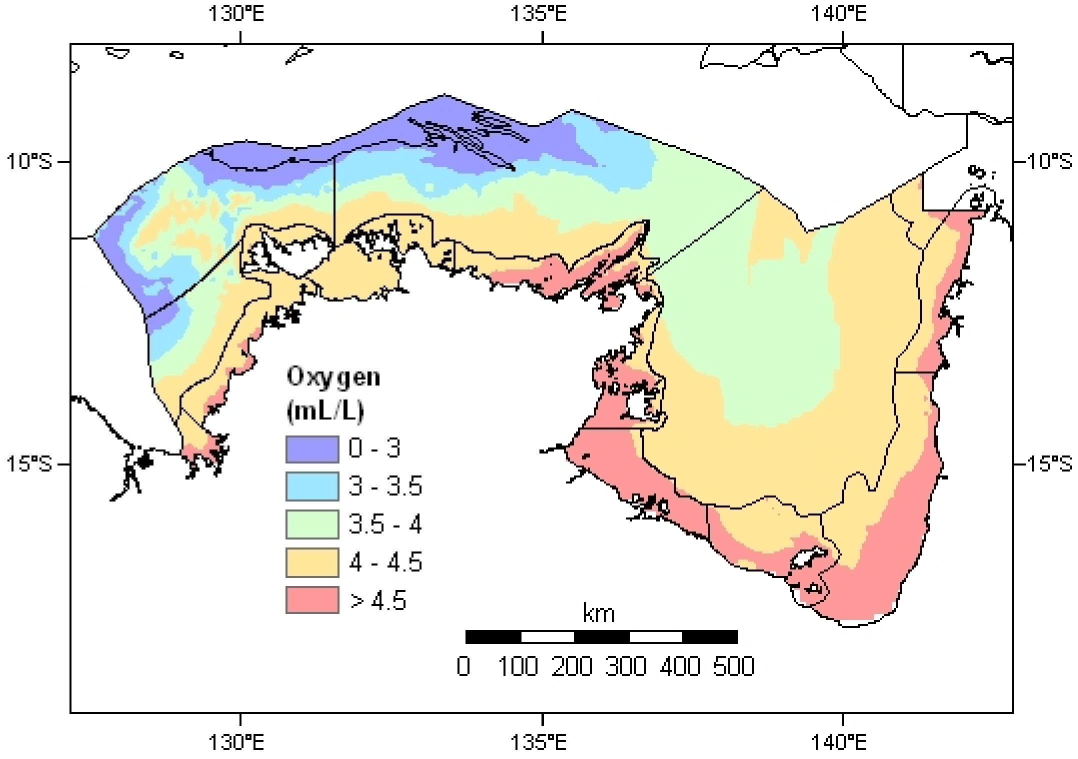
12.5.6 AVERAGE BOTTOM TEMPERATURE (°C)



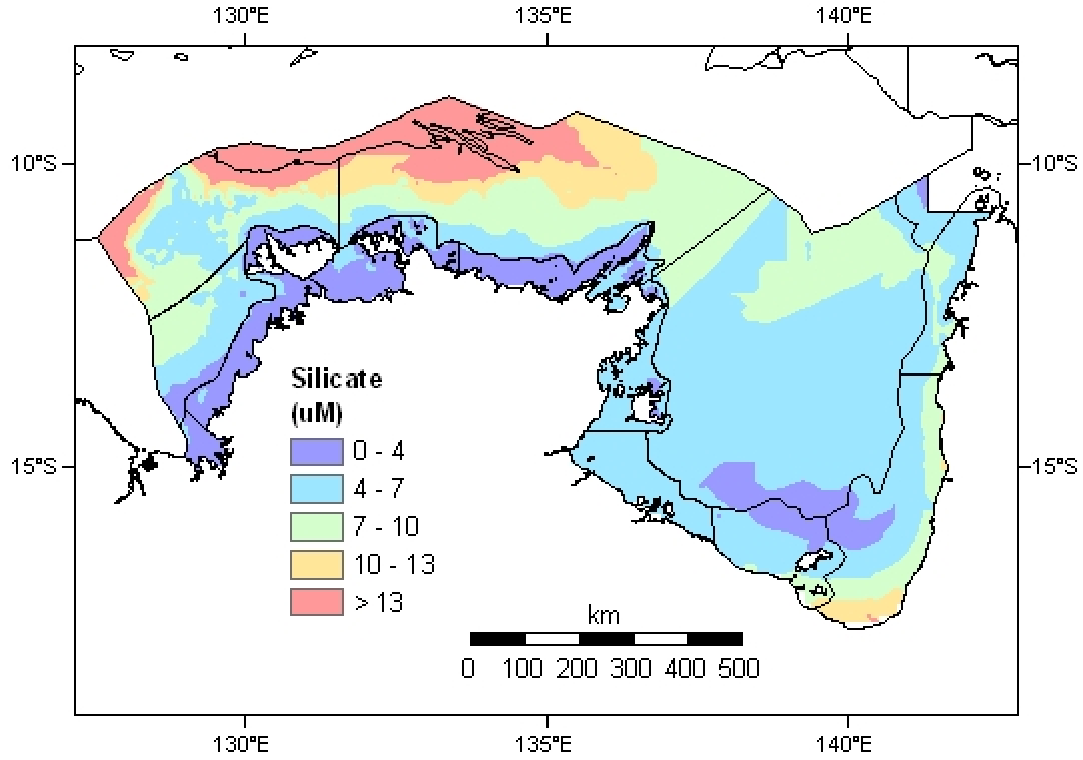
12.5.7 AVERAGE BOTTOM SALINITY (PSU)



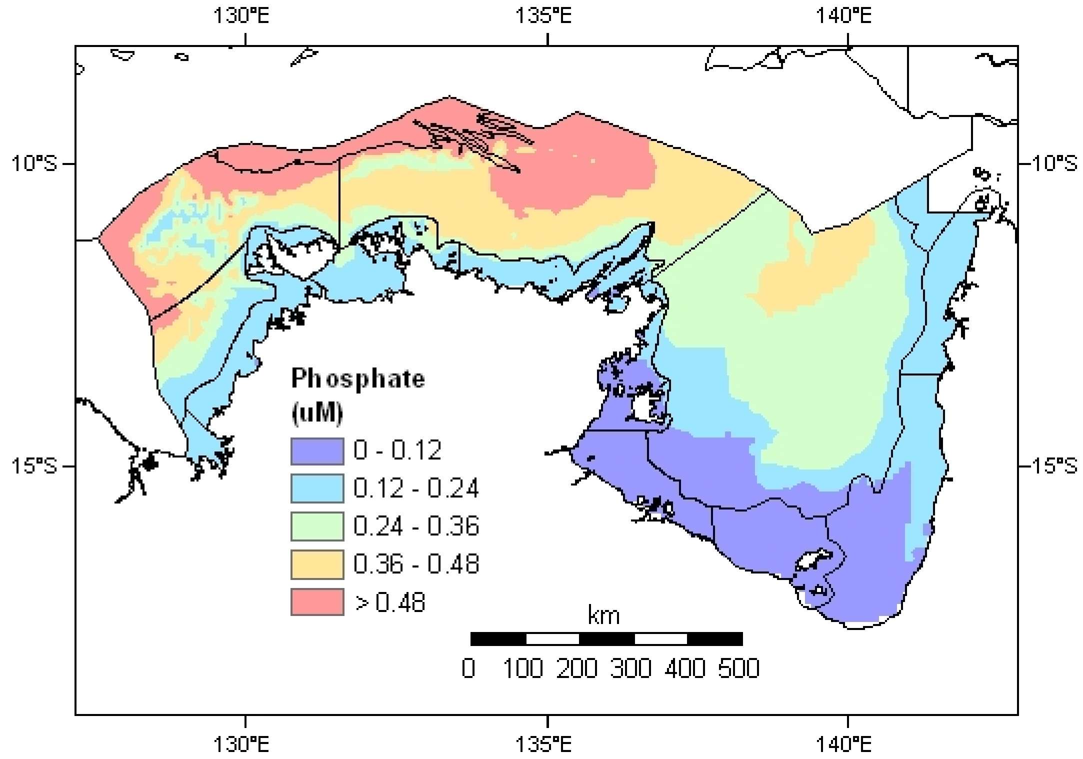
12.5.8 AVERAGE BOTTOM OXYGEN (ML/L)



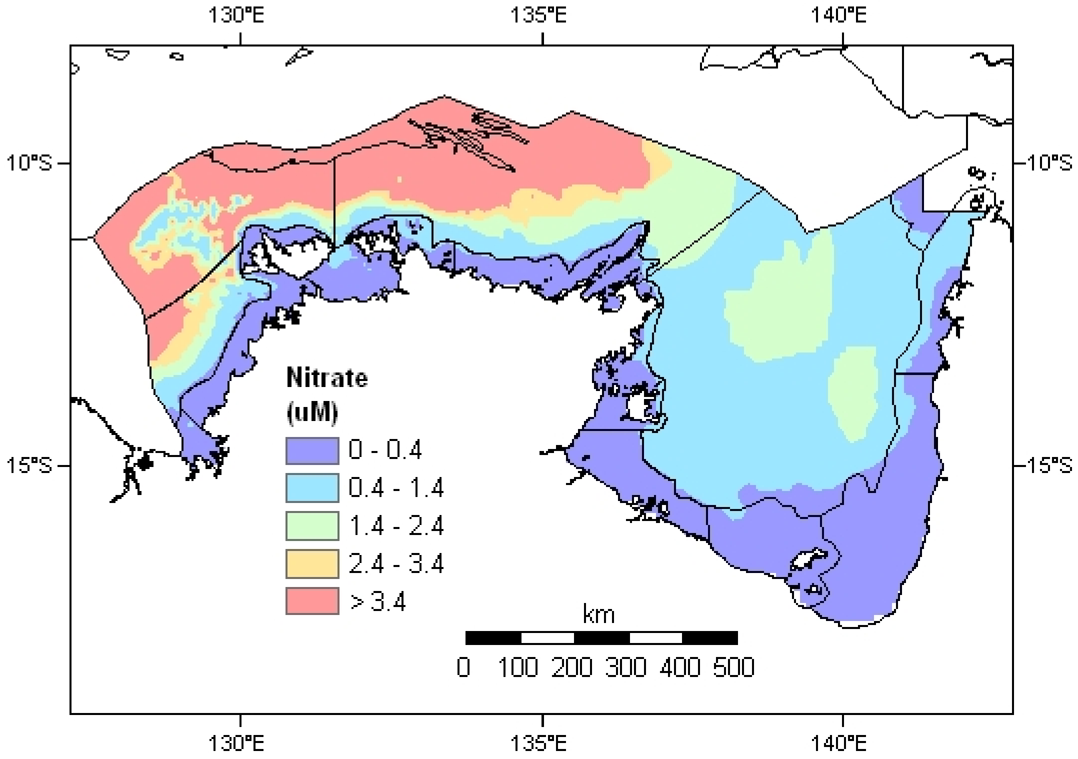
12.5.9 AVERAGE BOTTOM SILICATE MM



12.5.10 AVERAGE BOTTOM PHOSPHATE MM



12.5.11 AVERAGE BOTTOM NITRATE MM



12.6 OCEAN CURRENTS

The region is less influenced by ocean currents than other Australian marine regions because it is mostly a shallow sea that separates the tropical waters of the Indian and Pacific oceans (CSIRO 2001). The currents at any given instant are mostly due to tidal flows. However, the net flows that occur over time are important because they drive the longer term transport patterns (Rothlisberg et al. 2005).

Currents affecting the NMR include the Indonesian Throughflow, the South Equatorial Current and the Gulf of Carpentaria gyre (Figure 81). Although most of the region is shallow, deep channels in the north-west (see system 'Northwest IMCRA transition') carry water from the Indonesian Throughflow, which comprises a series of ocean currents that flow from the tropical Western Pacific Ocean through the Indonesian Seas into the Indian Ocean (Rothlisberg et al. 2005, CSIRO 2001, Thackway and Cresswell 1998). The water from these currents tends to be warm and of low salinity. The clockwise gyre in the Gulf of Carpentaria (system 'Gulf of Carpentaria') during the summer monsoon is the net flow of the tides (Forbes and Church 1983). Tidal currents moving forward and backward through Torres Straight affect the north-east of the NMR (Saint-Cast and Condie 2006). Net flows through Torres Strait are small (Wolanski et al. 1988), but circulation modelling predicts a net westward flow in winter, driven by the trade winds, and a net eastward flow in summer, driven by the monsoon winds (Saint-Cast and Condie 2006).

Currents within the NMR vary within and among years. Seasonal variation within years is caused by factors such as seasonal patterns of winds, water density (affected by temperature and salinity) and atmospheric pressure (Forbes and Church 1983, Rothlisberg et al. 2005). Variation among years is caused by climate variability (again affecting winds and water density) and variation in ocean currents, for example due to the El Niño Southern Oscillation (Phillips and Wijffels 2005).

A model of the hydrodynamics of the NMR has been developed and can be used to provide visual and statistical summaries of tides and circulation patterns and to provide modelled data for modelling and analysis (Hill et al. 2002, Rothlisberg et al. 2005).

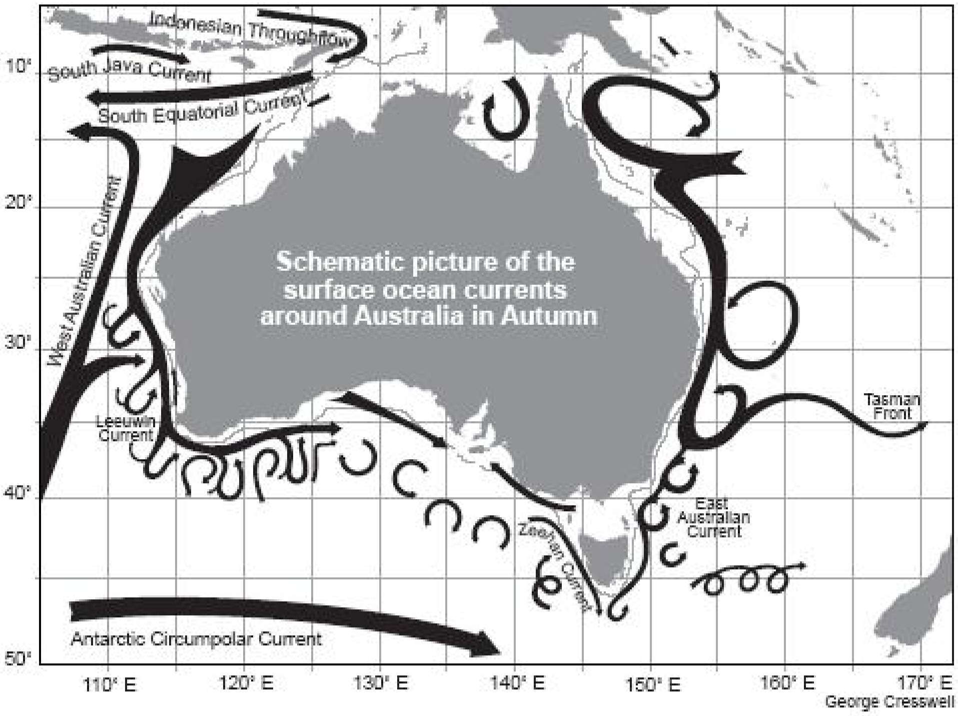


Figure 81. Schematic diagram of the surface ocean currents around Australia. Key currents for the NMR are the Indonesian Throughflow, the South Equatorial Current and the clockwise net flow of the Gulf of Carpentaria. Source: CSIRO 2001.

12.7 HABITATS

Perhaps the most characteristic habitats of the region are soft sediments on relatively flat terrain. The sediments are not uniform, but vary in factors such as the amounts of mud and sand and the availability of hard substrate for attachment by sessile organisms, and these factors affect the biological assemblages found within and on the sediments (Long et al. 1995).

Seagrasses provide key habitats in the NMR. They stabilise coastal sediments and trap and recycle nutrients. They provide nursery grounds for commercially harvested fish and prawns. They provide feeding grounds for dugongs and green turtles (Kirkman 1997).

Other marine habitats include geomorphic features scattered around the region (e.g. reefs, shoals and valleys; see above), and of course the water column. Islands provide important nesting sites for seabirds (National Oceans Office 2004a).

The coasts adjacent to the NMR provide a number of key habitats. Mangroves provide habitat for waterbirds and support many commercially and recreationally important fish and crustacean species for parts of their life cycles. They buffer the coast from large tidal movements, storm surges and flooding. They are used directly by Aboriginal people, and support other species that are also used by Aboriginal people (National Oceans Office 2004a). Beaches provide nesting sites for sea turtles (National Oceans Office 2004a), and coastal parts of rivers provide nesting sites for saltwater crocodiles (Letnic and Connors 2006, Read et al. 2004). Coastal wetlands provide feeding and nesting sites for waterbirds (National Oceans Office 2004a).

12.8 BIODIVERSITY

The fauna of the NMR is diverse, but the level of endemism is low because the majority of species are widespread through the Indo–Pacific region. In contrast, the relatively isolated southern Australian marine fauna has lower diversity and high endemism (Mummery and Hardy 1994).

12.8.1 MAMMALS AND REPTILES

Charismatic species include dugongs, sea turtles, crocodiles, dolphins and whales. The dugong population is globally significant, and dugongs are of cultural significance to Aboriginal people (National Oceans Office 2004a). All six species of sea turtles in the NMR are listed as endangered or vulnerable. The NMR breeding populations of green turtle, hawksbill turtle and flatback turtle are globally significant (National Oceans Office 2004a). Saltwater crocodiles are widespread in coastal areas and breed along the coastline (Letnic and Connors 2006, Read et al. 2004). About 30 species of sea snakes have been recorded in the region (National Oceans Office 2004a, Chapter 3 Rochester et al. 2007t).

12.8.2 BIRDS

The NMR is an important region for seabirds, waterbirds, waders and migratory birds (National Oceans Office 2004a). The region provides nesting sites, feeding sites and, for migratory birds, staging points. Some of the largest waterbird breeding colonies in Australia are in mangroves along the east coast of the Northern Territory (National Oceans Office 2004a).

12.8.3 DEMERSAL FISH

NMR demersal fish include trevallies, giant queenfish, barramundi, grunters, emperors, snappers, blue salmon, king threadfin, black jewfish and groupers. They are variously fished by commercial, recreational and Indigenous fishers (National Oceans Office 2004a).The demersal fish fauna of the NMR has been extensively studied. Research includes work on fish caught by demersal fish and prawn trawls, inshore fish and their diets (particularly in relation to the consumption of commercially harvested prawns) and sustainability assessment (see 'demersal fish').

A recent and comprehensive checklist of fish species for the NMR is not currently available, although the fish fauna in the region is known to be highly diverse. Russell and Houston (1989) reported 474 species from 123 families from the Arafura Sea. Griffiths et al. (2006b) summarised existing trawl data and reported 460 teleost and 56 elasmobranch species. There are likely to be many additional species in this overall list when we take into account non-trawled fishes that have been recorded in shallow inshore habitats and estuaries. These may include species from the 344 species recorded in Albatross Bay and the Embley estuary (Blaber et al. 1990b) and the 179 species from the Groote Eylandt region (Blaber et al. 1992a).

Many species of snappers and a few species of emperors have been recorded in the NMR, but the distributions of individual species are unknown (National Oceans Office 2004a). They form two of the most abundant predatory species groups in the habitats they occupy. Most snappers are demersal, and are often associated with structured habitats such as coral or rocky reefs. Emperors are generally associated with coral or rock reefs or nearby sand, rubble or seagrass habitats. Snappers and emperors are harvested by commercial, recreational and Indigenous fishers, with snappers dominating in all cases (National Oceans Office 2004a).

Groupers form one of the dominant groups of predatory fish on coral reefs and other areas they inhabit. They are caught by commercial and recreational fishers; however, fishing effort outside of Torres Strait may not be high (National Oceans Office 2004a).

There are many coastal fish in the coastal waters and estuaries of the NMR. Those most important to coastal fisheries and communities are barramundi, king threadfin, blue salmon, black jewfish, giant queenfish and grunters (*Pomadasys kaakan* and *Pomadasys argenteus*) (National Oceans Office

2004a). Although the ecological roles of coastal fishes are not well understood, the species are likely to be ecologically important because they include some of the most abundant predatory species in coastal waters (National Oceans Office 2004a). The fish are harvested by commercial, recreational and Indigenous fishers (National Oceans Office 2004a). The giant queenfish is ecologically important and of increasing importance to commercial and recreational fisheries (Griffiths et al. 2005).

Seahorses and pipefishes are captured as bycatch by trawling (National Oceans Office 2004a, Chapter 3 of this report). Some are variously listed under the EPBC Act, under state legislation or by IUCN (National Oceans Office 2004a).

There have been numerous analyses of the spatial variation in fish assemblage structure across northern Australia (Rainer and Munro 1982, Stobutzki et al. 2001a, Stobutzki et al. 2003, Dell et al. in press, Tonks et al. in press). Zhou and Griffiths (in press) synthesised the various studies to establish five bioregions for an ecological risk assessment for demersal trawl teleost bycatch in the Northern Prawn Fishery (Appendix C4). Last et al. (2005) collated data for offshore demersal fishes for Australia for use in bioregionalisation of the continental slope and outer shelf.

The feeding ecology of inshore and estuarine fishes of the Gulf of Carpentaria has been studied in detail, particularly in relation to consumption of commercially important prawns (Brewer et al. 1989, Brewer et al. 1991, Brewer et al. 1995, Salini et al. 1990, Salini et al. 1994, Salini et al. 1998, Haywood et al. 1998) (see system 'NPF prawn fishing grounds'). These studies indicate that the overall fish assemblage consumes a vast variety of prey that cover the entirety of the surrounding ecosystem. Many species have been found to be generalists or opportunistic, such as the larger predatory trevallies and snappers that consume a range of teleosts, crustaceans, and molluscs. In contrast, there are several species that are specialists, including detritivorous mullets and milkfish, and a weasel shark, Hemigaleus microstoma, which almost exclusively consumes octopus (Stevens and McLoughlin 1991, Salini et al. 1992).

12.8.4 PELAGIC FISH

Despite the extensive research undertaken on demersal fishes in northern Australia, there is still a poor understanding of the pelagic fish community and pelagic fish habitats, biology, population status and trophic interactions with other ecosystem components. One exception is Spanish mackerel, which due to its value to commercial and recreational fisheries in northern Australia has been the subject of several studies focusing on age and growth (McPherson 1992, Buckworth 1999), reproductive biology (Mackie et al. 2005), stock structure as determined from tagging (Buckworth et al. in press), otolith microchemistry (Newman et al. in press), genetics (Ovenden and Street in press), parasites (Lester et al. 2001, Moore et al. 2003) and stock assessment (Walters and Buckworth 1997, Buckworth and Clarke 2001).

A recent CSIRO study (Griffiths et al. in prep.) summarised existing pelagic fish datasets from northern Australia, and opportunistically collected additional data on the pelagic fishes caught by commercial vessels in Queensland's N9 shark gillnet fishery in the eastern Gulf of Carpentaria.

A total of 61 pelagic fish species from 16 families have been recorded from the four quantitative surveys conducted in northern Australia. The largest contributions to the pelagic fish assemblage by number of species were from the families Carangidae (23 species), Scombridae (16 species), Elopidae and Istiophoridae (4 species). In terms of number of fish, five species—longtail tuna, grey mackerel, Spanish mackerel, mackerel tuna, black pomfret and spotted mackerel—were by far the most abundant, contributing 92% to the overall catches. It is noteworthy that the next most abundant species, albeit contributing less than 1% to catches, were large predators including Indo–Pacific sailfish, black marlin and cobia.

Mackerels and tunas are pelagic or epipelagic. Mackerels tend to be coastal, whereas tunas tend to be oceanic and to range widely. The longtail tuna is atypical for a tuna because it is coastal rather than oceanic (National Oceans Office 2004a). Mackerels and tunas are harvested by commercial and recreational fishers (National Oceans Office 2004a).

There has apparently been no comprehensive dietary study of the pelagic fish fauna in northern Australia. A preliminary study of the feeding ecology of pelagic fishes in the Gulf of Carpentaria was recently completed by CSIRO, although the majority of data remain in an unpublished report (Griffiths et al. in prep.) and the works conducted in this project (FRDC 050/2005). This preliminary study suggests there are four main functional groups of pelagic fishes in northern Australia: 'billfishes', 'large tunas and mackerels', 'small tunas, mackerels and bonitos', and 'small planktivorous scads and mackerels'. Species within each of these groups were all shown to feed opportunistically as indicated by high prey diversity and high variability in the diet composition among individuals. Although most species primarily fed on pelagic fishes, the diet of many species comprised demersal or benthic prey indicating these fishes utilise the entire water column for feeding.

Based on this work, a more detailed analysis of the diet of longtail tuna has recently been published (Griffiths et al. 2007). The paper describes the spatial, temporal and ontogenetic variability in diet composition, and provides daily ration and annual prey consumption estimates for a number of prey groups, including commercially important prawns, in the Northern Prawn Fishery. This species is ecologically important in the region since it is a large-growing species (up to 36 kg), is highly abundant (1.8 fish per square kilometre in the Gulf of Carpentaria; Griffiths et al. in press) and consumes vast quantities of prey. This study estimated that longtail tuna consumes 148,178 t/y of prey in the Gulf of Carpentaria, primarily small schooling pelagic clupeids and engraulids, and 599 t/y of penaeids; equivalent to 11% of the annual commercial catch.

12.8.5 BENTHOS AND INFAUNA

On the soft sediments of the Gulf of Carpentaria, the diversity of megabenthic invertebrates is moderately high. Characteristic groups include echinoids (e.g. heart urchins and sand dollars), sponges, solitary corals, molluscs (especially bivalves), decapods, bryozoans, sea cucumbers and sessile tunicates (Long et al. 1995, Haywood et al. 2005). In the predominantly sandy sediments in the east and south-east of the Gulf the megabenthos comprises mainly sessile, suspension feeding sponges, solitary corals, sea pens, bivalves and sessile tunicates. In the muddier sediments in the central and western Gulf it comprises mainly deposit-feeding heart urchins and sand dollars. However, sessile suspension-feeders are also found in the central Gulf in places where the substrate is suitable for attachment (Long et al. 1995).

NMR crustaceans include mud crabs, swimmer crabs, lobsters and bugs. Crabs are harvested by commercial, recreational and traditional fishers (National Oceans Office 2004a). Lobsters are probably sparsely distributed outside of Torres Strait (National Oceans Office 2004a). Bugs are mainly caught by the Northern Prawn Fishery as byproduct (National Oceans Office 2004a) (see system 'NPF prawn fishing grounds').

Corals occur both as reefs and in non-reef coral communities. Reefs include patch reefs and fringing reefs, and are sparsely distributed (e.g. the patch reefs of the Gulf of Carpentaria, see 'coral reef' in system 'Karumba–Nassau'). Non-reef coral communities can be extensive and diverse (e.g. see 'coral' in system 'Arnhem Wessel') (National Oceans Office 2004a).

Benthic plants include seagrasses and green, brown and red algae (Kirkman 1997, Haywood et al. 2005).

In the Gulf of Carpentaria, infaunal species are moderately diverse. Species include polychaetes, crustaceans (e.g. decapods, amphipods, tanaids, ostracods and cumaceans), molluscs (mainly bivalves), echinoderms (e.g. brittle stars, heart urchins and sea cucumbers), ribbon worms and peanut worms. Abundance is typical for a tropical shelf region. Abundance and diversity are highest in the sandier sediments of the east and south-east. Species tend to be scavengers and deposit feeders rather than suspension feeders and herbivores (Long and Poiner 1994).

12.8.6 PLANKTON AND PRODUCTIVITY

The phytoplankton of the NMR shows high species diversity (ca 200 species) and is dominated by a nanoplanktonic (2–20 μm) diatom flora, quite different from the oceanic dinoflagellate flora of the adjacent Coral Sea and Indian Ocean (Hallegraeff and Jeffrey 1984). Chlorophyll concentrations and productivity are relatively high (Hallegraeff and Jeffrey 1984). There are no major upwellings in the NMR. However, there appears to be some upwelling in the eastern Arafura Sea and Joseph Bonaparte Gulf (Rochford 1962).

In the Gulf of Carpentaria the standing stock of zooplankton is very high (mean estimate 77 mg/m3 dry weight) and compares favourably with the very high abundances in the seasonal upwelling areas of the north-west shelf of Australia (Rothlisberg and Jackson 1982). The copepod faunal composition is characteristic of warm neritic waters and shows large similarities to that of South-East Asia (88 of the 102 species in common) (Othman et al. 1990).

12.9 DRIVERS OF CHANGE

Human activities within and outside of the NMR have impacts or potential impacts (threats) on the ecosystems of the NMR. Activities and changes having current or potential impacts include the following:

- *fishing.* NMR fishing includes prawn trawling (see system 'NPF prawn fishing grounds'), line fishing, net fishing, fish trawling, recreational fishing and traditional fishing. Impacts include mortality of target and non-target species and, in the case of trawling, possible disruption of the seabed (Hill et al. 2002).
- *illegal, unreported and unregulated (IUU) fishing for sharks.* IUU shark fishing, largely by Indonesian fishing vessels, is common in the NMR. Although the catch rate has not yet been accurately estimated, it may greatly exceed that by regulated Australian fishing vessels (J. P. Salini pers. comm.)
- *ministerial direction on fisheries management.* In December 2005 the Minister for Fisheries, Forestry and Conservation directed the Australian Fisheries Management Authority to address the slow recovery of overfished stocks, to improve stock management and to improve the management of broader environmental impacts. Specific measures were prescribed and included more general use of catch quotas rather than effort controls
- *agriculture*. Impacts from agriculture and land clearing include siltation, chemical and nutrient run-off and water diversion. Threats include acid run-off due to disturbance of acid sulphate soils (Hill et al. 2002).
- *aquaculture.* Species currently farmed include prawns, pearl oysters and barramundi. Potential impacts include pollution and habitat destruction. Threats include introduction of diseases (Hill et al. 2002)
- *mining.* Mining activity affecting the NMR includes extraction of minerals on the land (especially around the Gulf of Carpentaria) and the production of oil and gas in the Timor Sea. Impacts from mining include dredging around ports and siltation of estuaries. Threats include spillage of ore and chemicals, and oil spills from oil wells and fuel transport (Hill et al. 2002)
- *shipping.* There is a considerable amount of shipping among ports in the NMR (e.g. Darwin Harbour) and to and from ports outside of the NMR. Impacts include leaching of toxins from anti-fouling paint. Threats include introduction of marine pests carried on hulls or in ballast water and oil spills (Hill et al. 2002)
- *water resource development.* There are economic and social pressures to develop the relatively rich freshwater resources in tropical Australia, potentially disrupting the flow of river water into the NMR (Hamilton and Gehrke 2005)
- *coastal development.* Darwin is the main focus of coastal development in the NMR. Impacts include habitat loss (e.g. of mangroves) and siltation (Hill et al. 2002)
- *climate change.* Predicted climate change includes changes to air and water temperature, wind, currents and water chemistry. Indicators of predicted climatic changes that have the potential

to stress ecosystems are generally moderate for the NMR relative to those of other Australian regions (Hobday et al. 2006)

Hill et al. (2002) performed a qualitative risk assessment of the threats to the seabed fauna of the managed area of the Northern Prawn Fishery (which covers almost the same area as the NMR). The assessment was based on the likelihood and consequence of each threat. Two threats were scored as being of extreme risk: (a) introduction of a serious marine pest and (b) altered rainfall patterns due to climate change. Five threats were scored as being of high risk: (a) rise in sea level, (b) rise in sea temperature, (c) increased frequency of climate change, (d) changes in water flows in estuaries, and (e) impacts on benthos from prawn trawling (threats a–c being elements of predicted climate change).

12.10 SOCIAL, ECONOMIC AND CULTURAL FEATURES

Social, economic and cultural features are outside of the scope of this review. However, general reviews are provided in Larcombe et al. 2006, National Oceans Office 2004b, National Oceans Office 2005 and OESR 2004.

Component and process summary:

Inputs

Component	**Affected by**	**Affects**	**Attributes**
climate			climate type (monsoonal)
rainfall			seasonal variation (high), year-to-year variation (high)
wind		currents	seasonal pattern (monsoonal)
cyclones		seabed, seagrass, coast	frequency (moderate)
tides		currents	spatial variation (high)
currents	wind, tides		
upwelling			intensity (low)
shark fishing		elasmobranchs	
IUU shark fishing		elasmobranchs	

Habitats

Component	**Affected by**	**Affects**	**Attributes**
water column			nutrient level (low), temperature (warm)
seabed	cyclones		topography (flat)
coral reef			
seagrass	cyclones	fish, prawns, dugongs, sea turtles	
island		sea turtles	
coast	cyclones	birds, sea turtles, crocodiles	
coastal wetland		birds	
mangrove			

Functional groups

Component	Affected by	Affects	Inter-system flow	Attributes
fish	seagrass			
demersal fish				
pelagic fish				
elasmobranchs	shark fishing, IUU shark fishing			
crustaceans				
prawns	seagrass			
sponges				
coral				
cetaceans				
dugongs	seagrass			
birds	coast, coastal wetland			
sea turtles	marine debris, seagrass, island, coast			
crocodiles	coast			
sea snakes				

Pools

Component	Affected by	Affects	Inter-system flow	Attributes
marine debris		sea turtles		

Processes

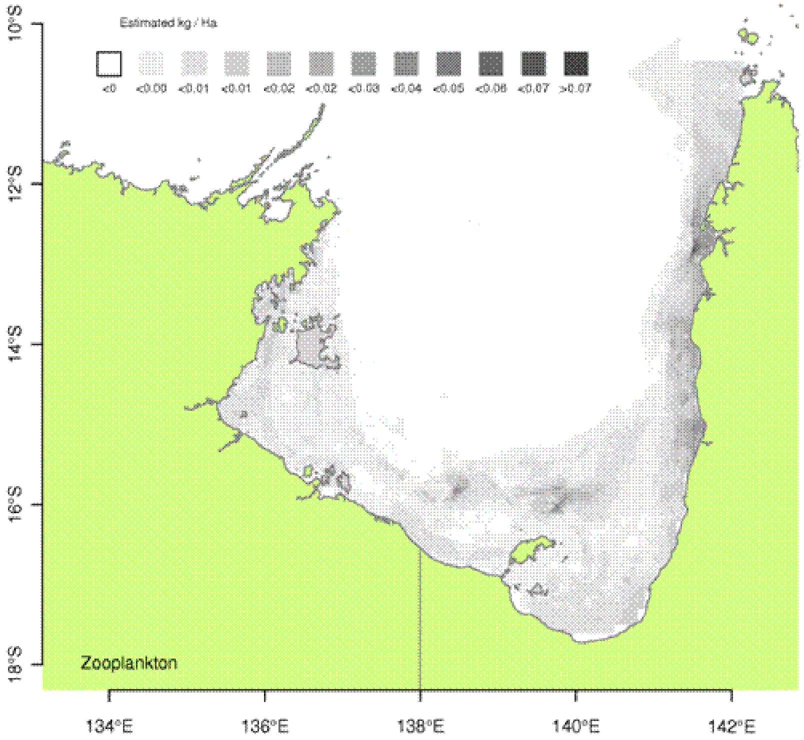
Process	Affecting components	Affected components
Wind inducement of currents	wind	currents
Cyclone disturbance of the seabed	cyclones	seabed
Cyclone disturbance of seagrass	cyclones	seagrass
Cyclone disturbance of the coast	cyclones	coast
Tide inducement of currents	tides	currents
Use of seagrass by fish as habitat	seagrass	fish
Use of seagrass by fish as a nursery	seagrass	fish
Use of seagrass by prawns as a nursery	seagrass	prawns
Use of seagrass by dugongs for feeding	seagrass	dugongs
Use of intertidal mudflats by shorebirds for feeding	coastal wetland	birds
Use of the coast by birds for breeding	coast	birds
Use of seagrass by sea turtles for feeding	seagrass	sea turtles
Use of islands by sea turtles for breeding	island	sea turtles
Use of the coast by sea turtles for breeding	coast	sea turtles
Use of the coast by crocodiles for breeding	coast	crocodiles
Capture of sharks by shark fishing	shark fishing	elasmobranchs
Capture of sharks by IUU shark fishing	IUU shark fishing	elasmobranchs
Entanglement of sea turtles in derelict fishing gear	marine debris	sea turtles

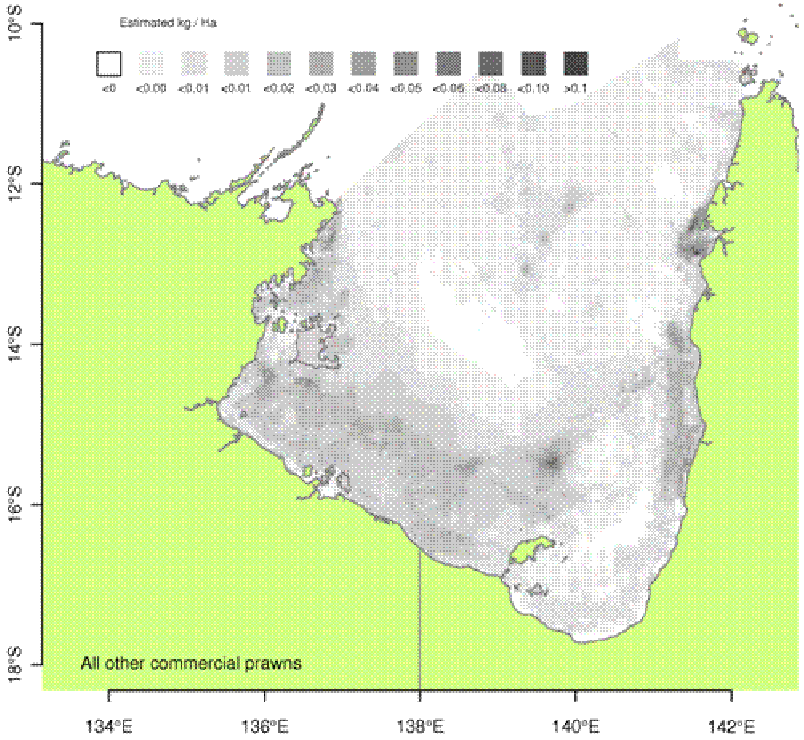
12.11 SPATIAL PREDICTION OF THE MAIN BIOTIC FUNCTIONAL GROUPS

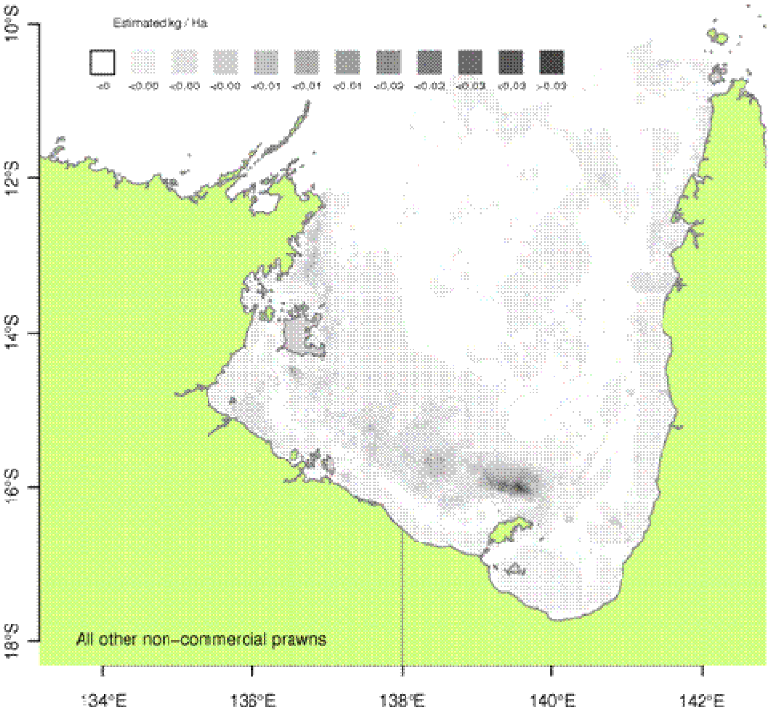
The biotic component of the ecosystem characterisation was a core output of this project and was achieved with the complementary project funded by DEWHA “The North Marine Region marine bioregional plan: Information and analysis for the regional profile“ (Rochester et al. 2007). This project, in conjunction with this FRDC 2005/050, conducted an in-depth characterisation of the Northern Marine Region (NMR) that included most of the Northern Prawn Fishery grounds. This section provides the detailed spatial predictions of the species distribution model (see Appendix 8) and the resulting maps are presented below. These ouputs have been central to this project since its underlying data sets and food web, ERA and spatial MSE models used interpolation maps.

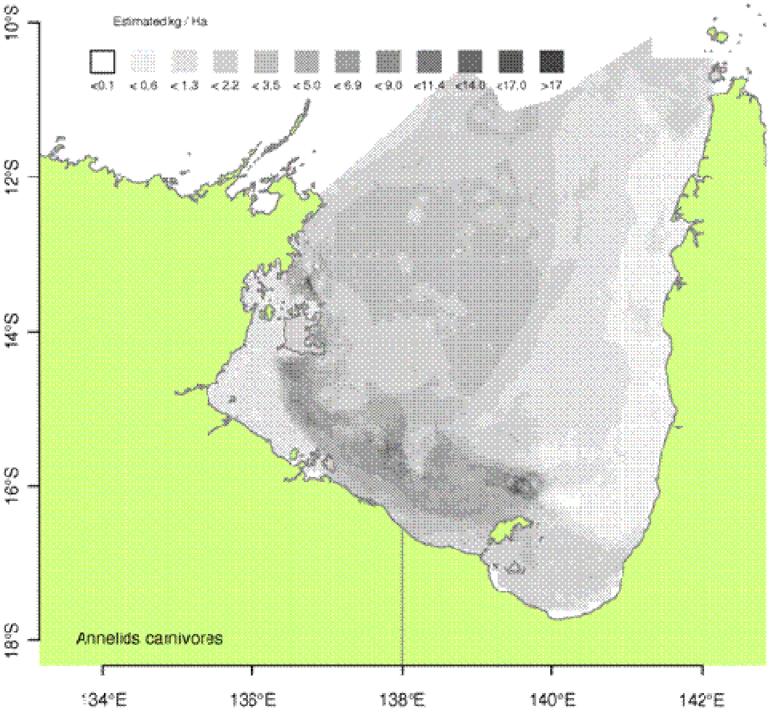
The maps below present the spatial characterization and visualisation of the spatial predictions for all 44 main functional groups expressed in the overall biomass predictions (in kg/ha) of the different trophic functional groups that characterise the benthic ecosystem of the Gulf of Carpentaria. These predictions were the outputs of the species distribution biophysical model and were used in the trophic modelling, in the construction and operation of the food web ecosystem model and, subsequently, by the the ecological risk assessment ERA model.

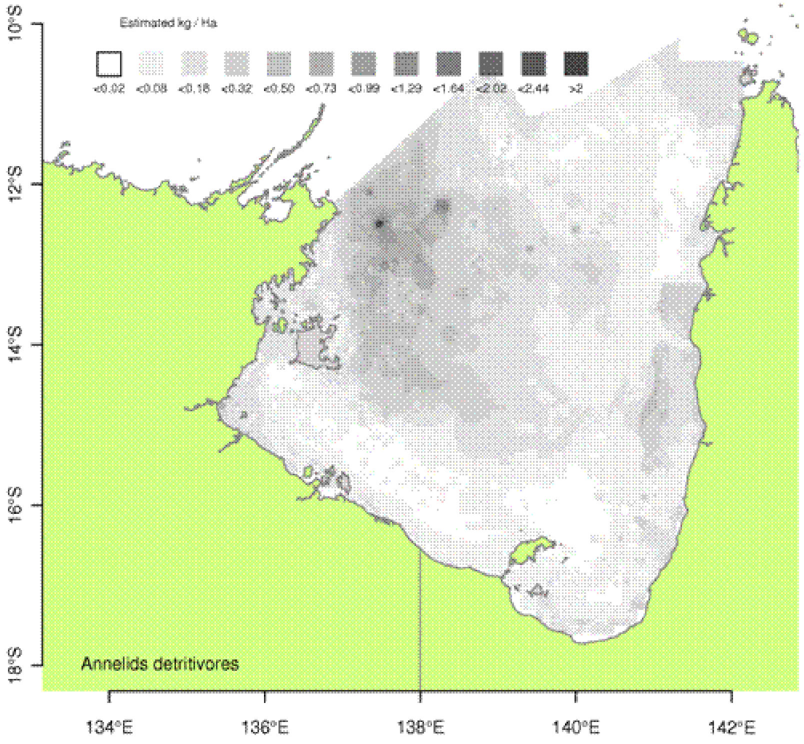
The spatial coverage of these spatial predictions is exactly the same as the area modelled by the overall spatial MSE framework.

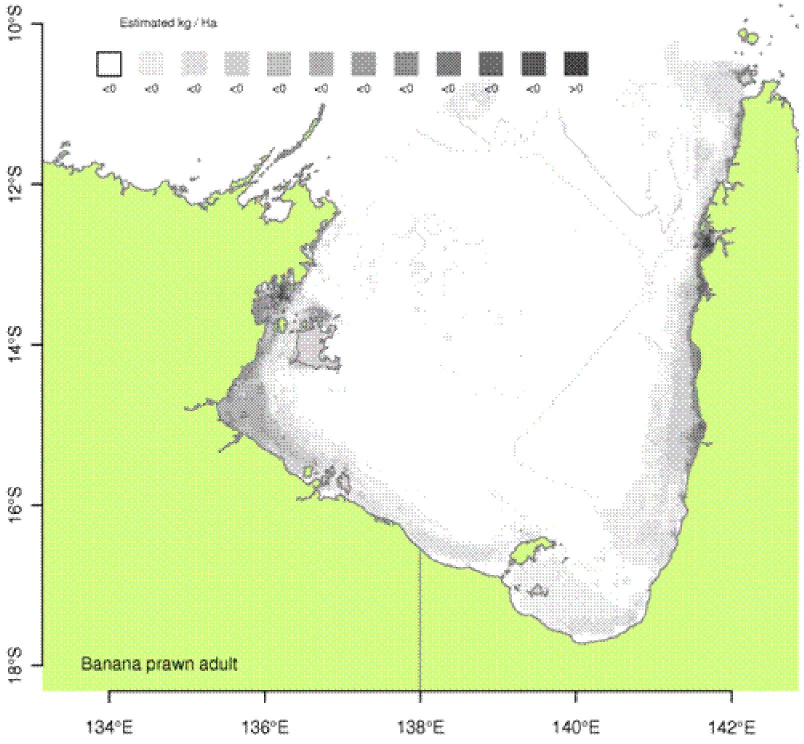

Zooplankton

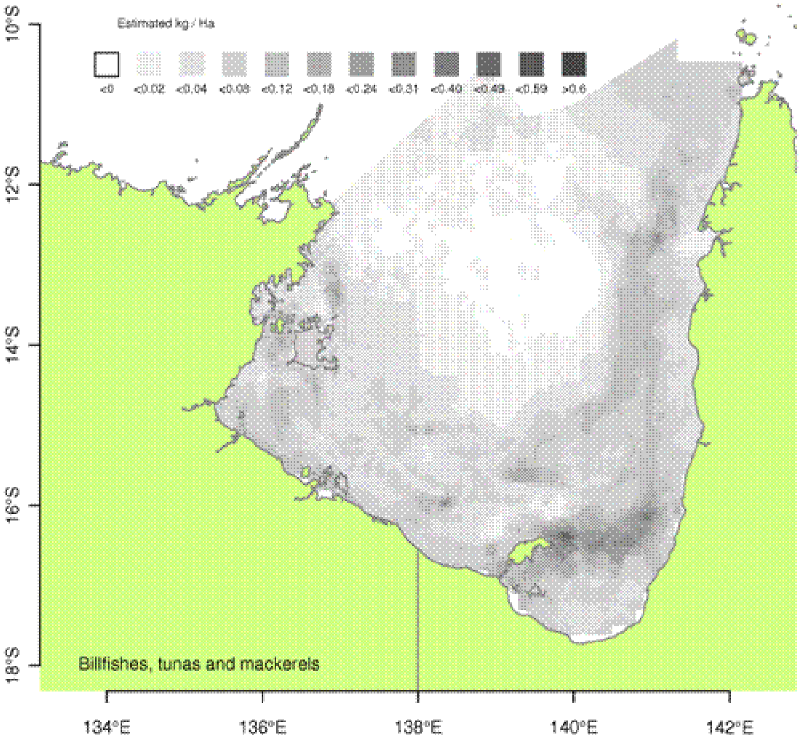

All other commercial prawns

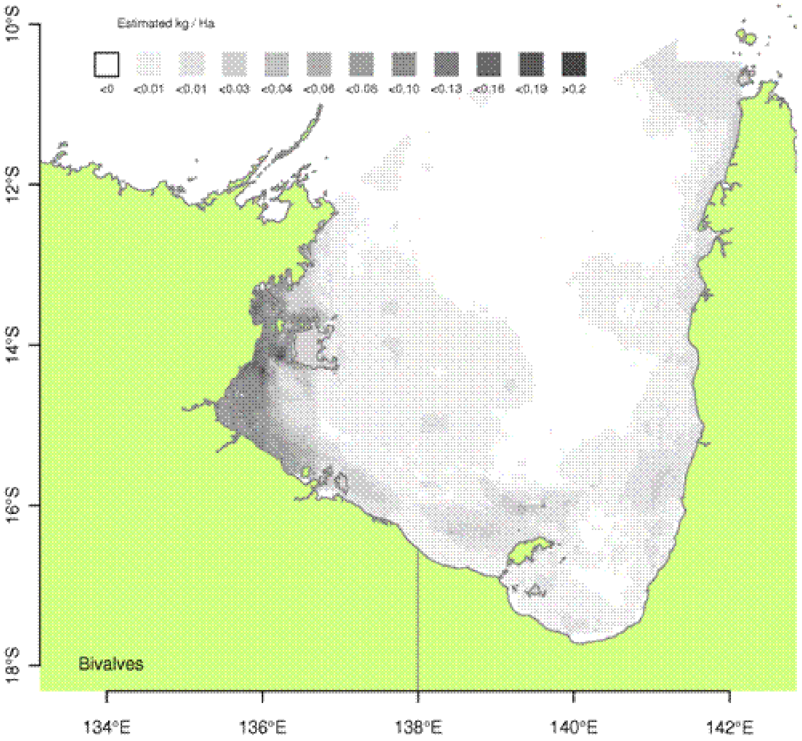

Estimatedkg / Ha
<0
<0.00
<0.00
<0.00
<0.01
<0.01
<0.01
<0.02
<0.02
<0.03
<0.03
>0.03
10°S
12°S
14°S
16°S
18°S
All other non–commercial prawns
134°E
136°E
138°E
140°E
142°E

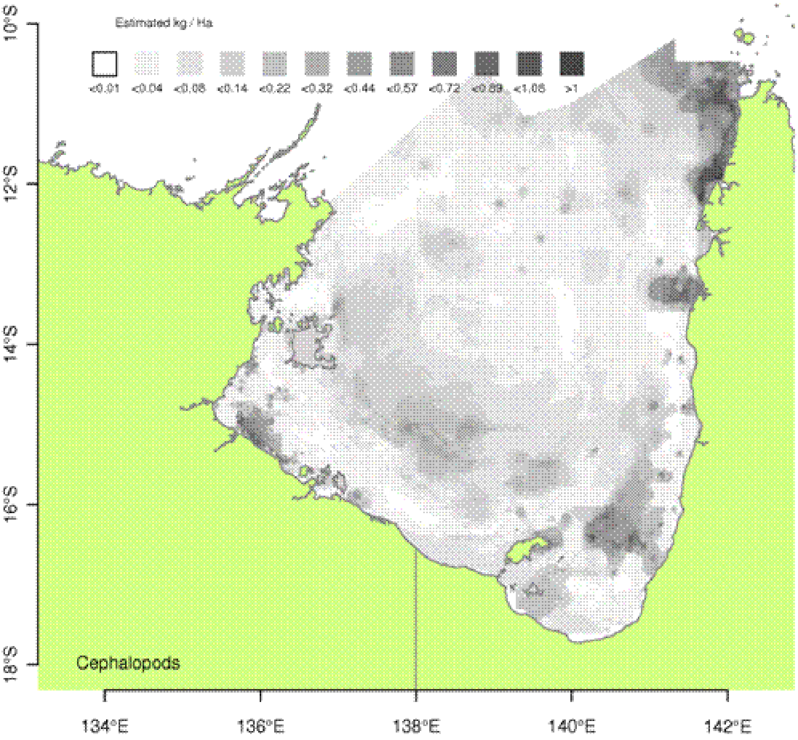

Estimatedkg / Ha
<0.1
< 0.6
< 1.3
< 2.2
< 3.5
< 5.0
< 6.9
< 9.0
<11.4
<14.0
<17.0
>17
10°S
12°S
14°S
16°S
18°S
Annelids carnivores
134°E
136°E
138°E
140°E
142°E

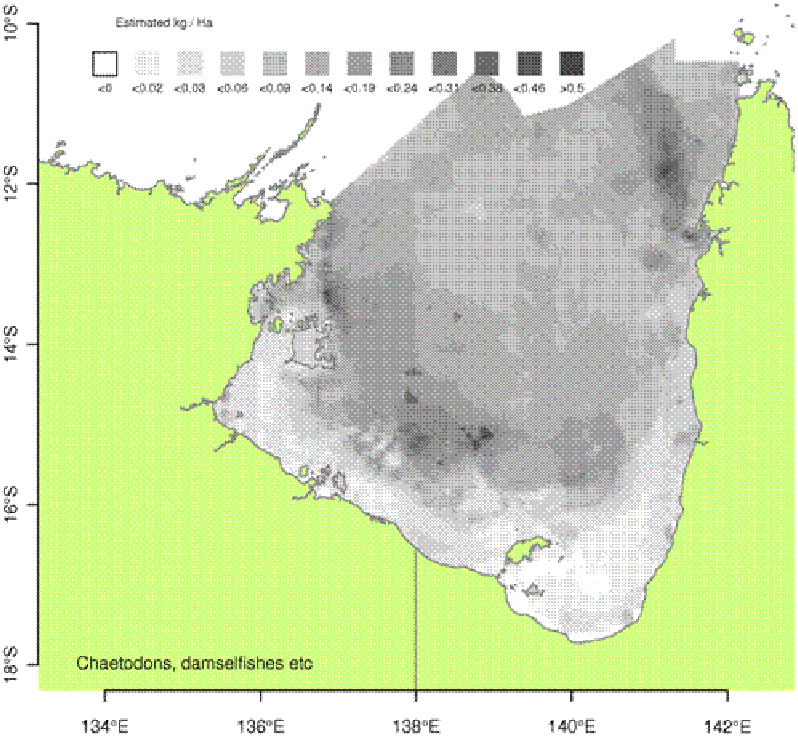

Estimated kg / Ha
<0.02
<0.08
<0.16
<0.32
<0.50
<0.73
<0.99
<1.29
<1.64
<2.02
<2.44
>2
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E
Annelids detritivores

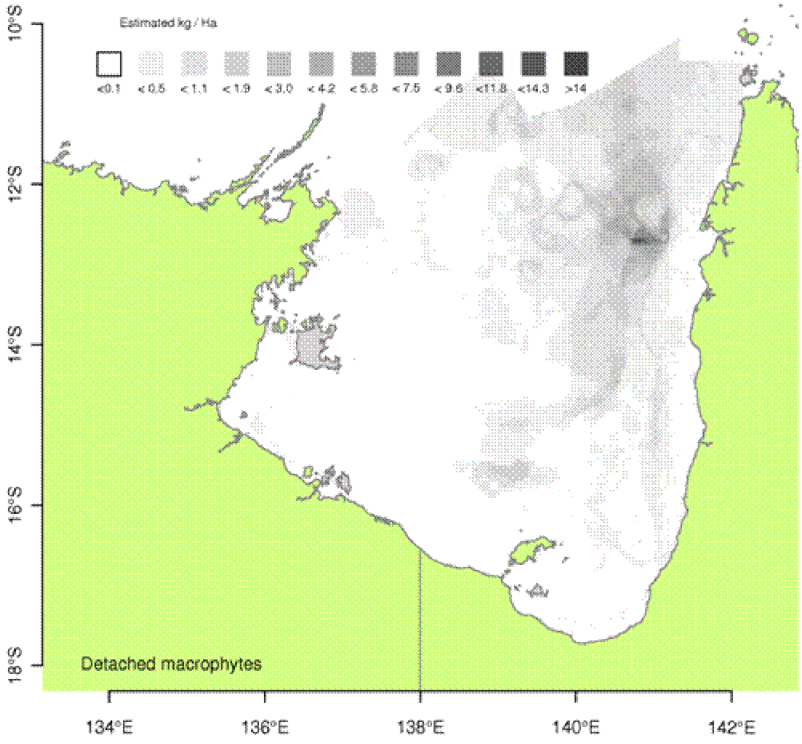

Estimated kg / Ha
<0
<0
<0
<0
<0
<0
<0
<0
<0
<0
<0
>0
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E
Banana prawn adult

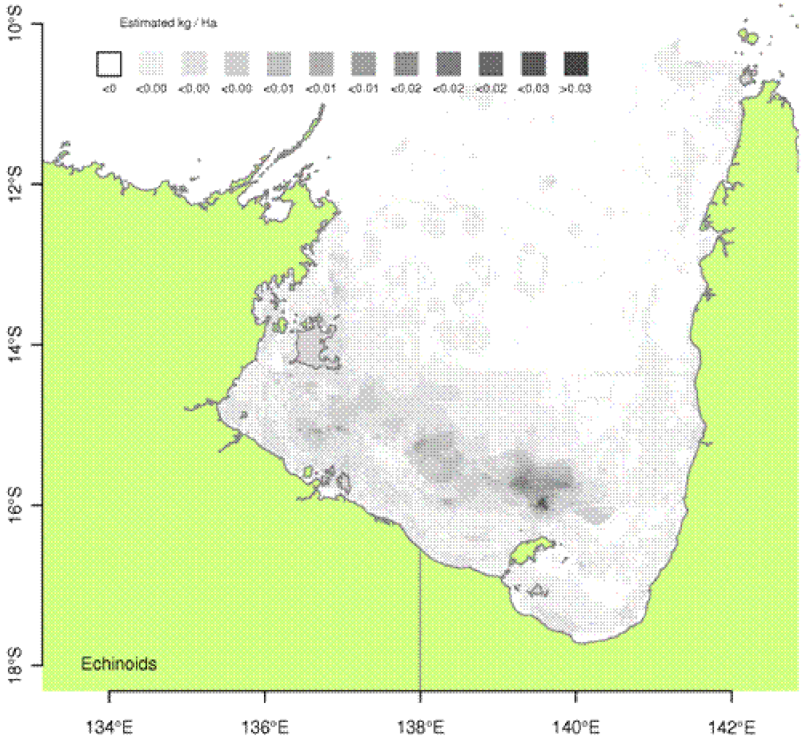

Estimated kg / Ha
<0
<0.02
<0.04
<0.08
<0.12
<0.18
<0.24
<0.31
<0.40
<0.49
<0.59
>0.6
Billfishes, tunas and mackerels
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

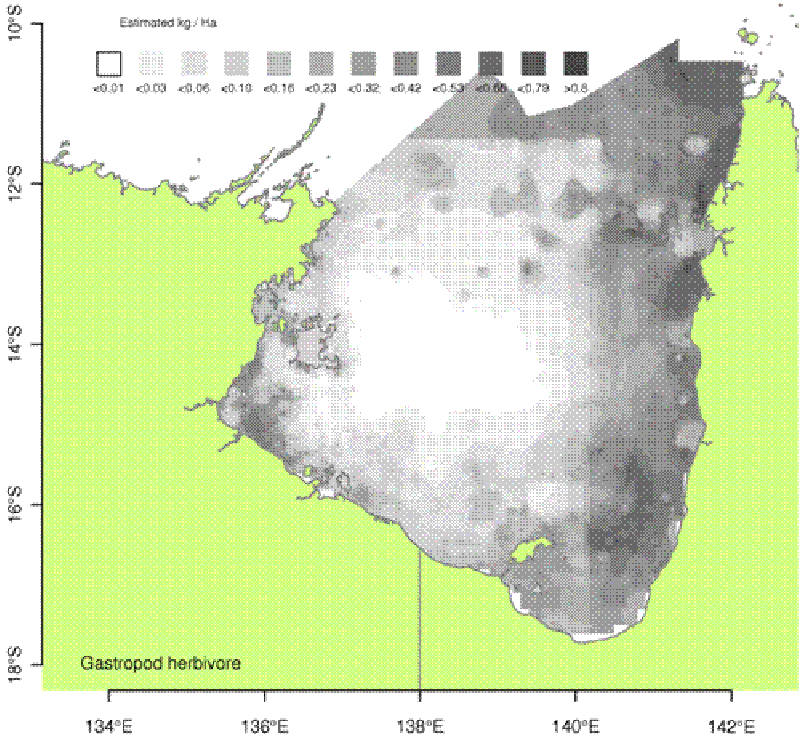

Estimated kg / Ha
<0
<0.01
<0.01
<0.03
<0.04
<0.06
<0.08
<0.10
<0.13
<0.16
<0.19
>0.2
Bivalves
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

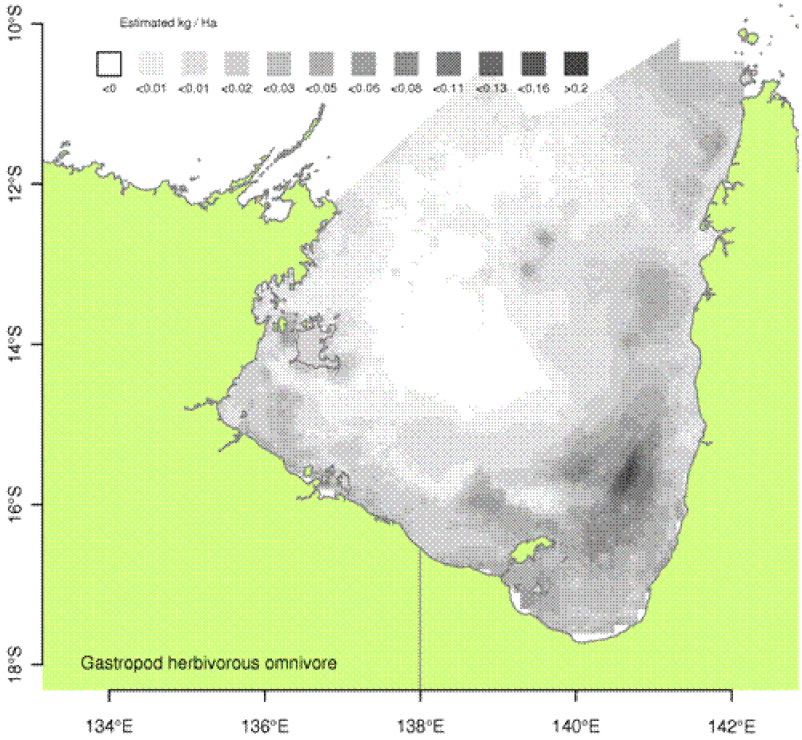

Estimated kg / Ha
<0.01
<0.04
<0.08
<0.14
<0.22
<0.32
<0.44
<0.57
<0.72
<0.89
<1.06
>1
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E
Cephalopods

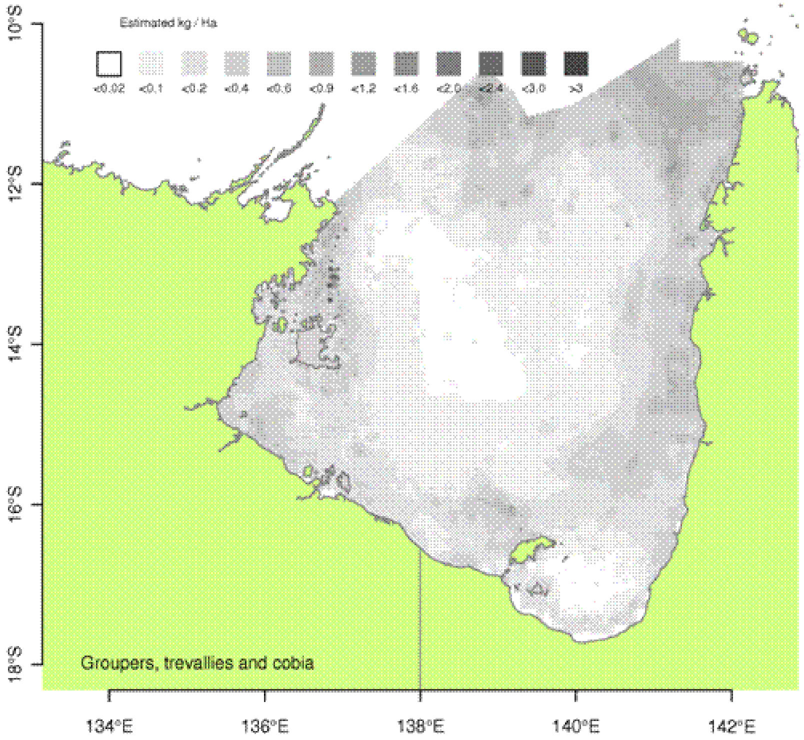

Estimated kg / Ha
<0
<0.02
<0.03
<0.06
<0.09
<0.14
<0.19
<0.24
<0.31
<0.38
<0.46
>0.5
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E
Chaetodons, damselfishes etc

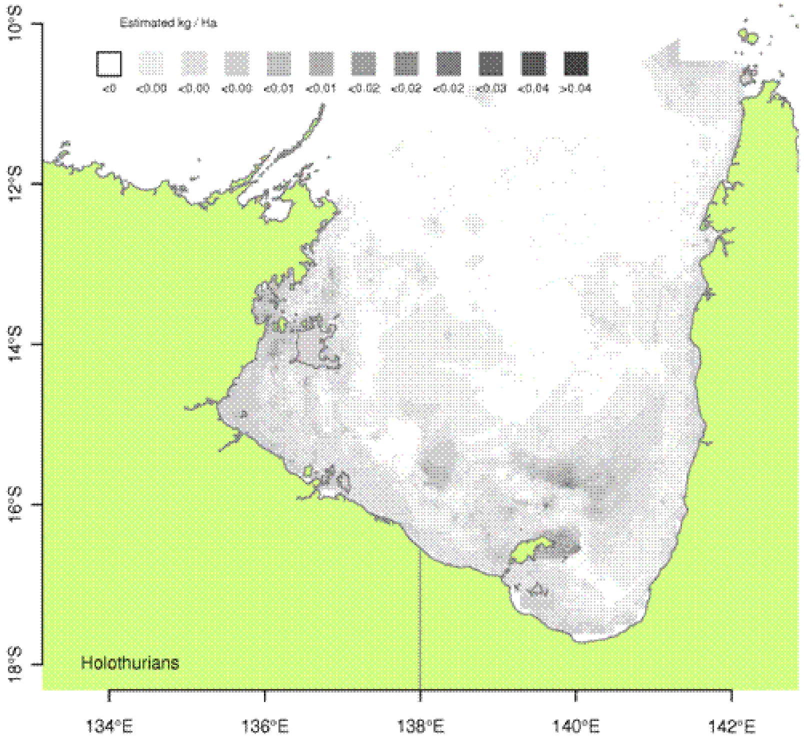

Estimated kg / Ha
<0.1
< 0.5
< 1.1
< 1.9
< 3.0
< 4.2
< 5.8
< 7.5
< 9.6
<11.8
<14.3
>14
10°S
12°S
14°S
16°S
18°S
Detached macrophytes
134°E
136°E
138°E
140°E
142°E

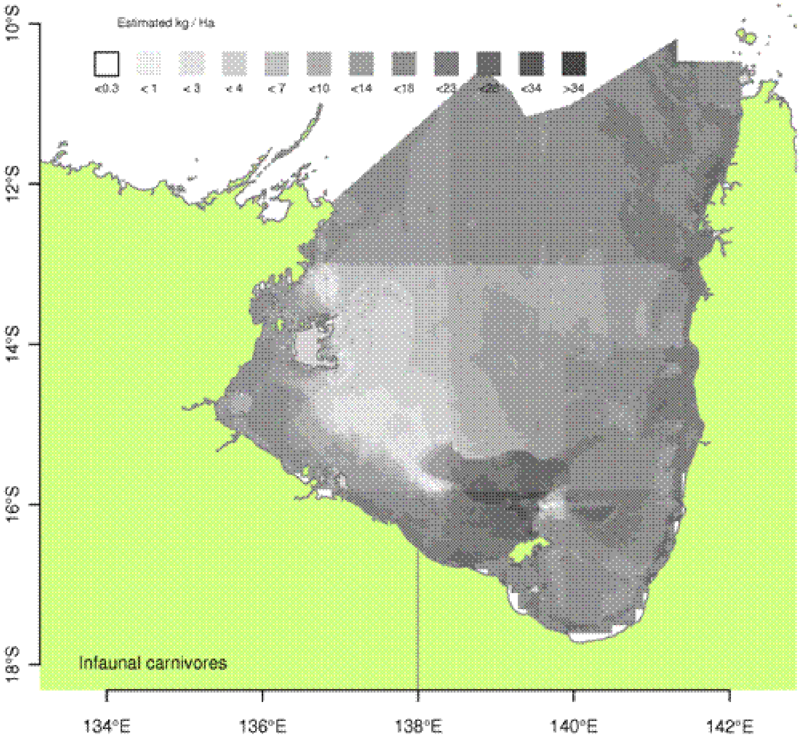

Estimated kg / Ha
<0
<0.00
<0.00
<0.00
<0.01
<0.01
<0.01
<0.02
<0.02
<0.02
<0.03
>0.03
10°S
12°S
14°S
16°S
18°S
Echinoids
134°E
136°E
138°E
140°E
142°E

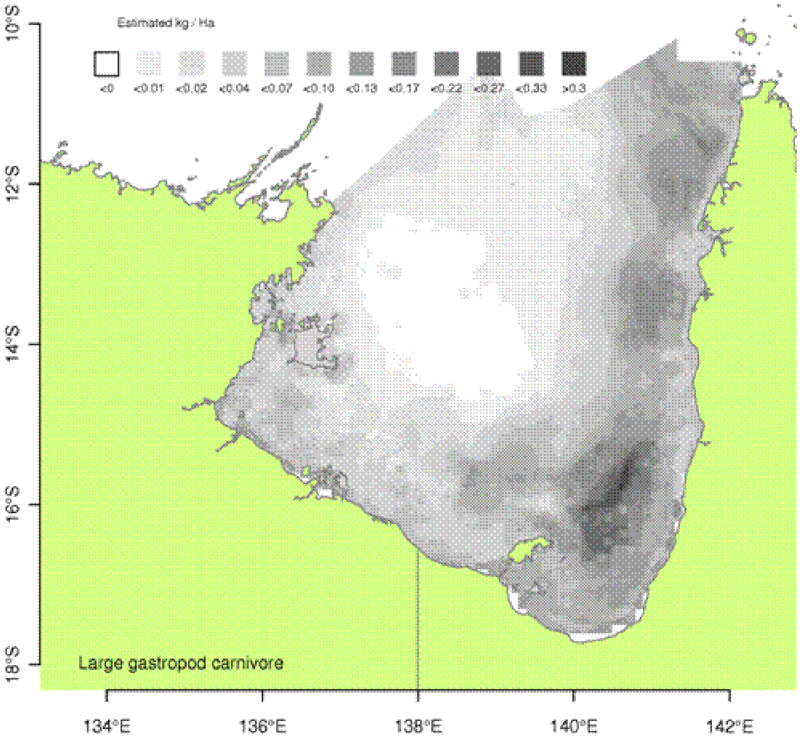

Estimated kg / Ha
<0.01
<0.03
<0.06
<0.10
<0.16
<0.23
<0.32
<0.42
<0.53
<0.66
<0.79
>0.8
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E
Gastropod herbivore

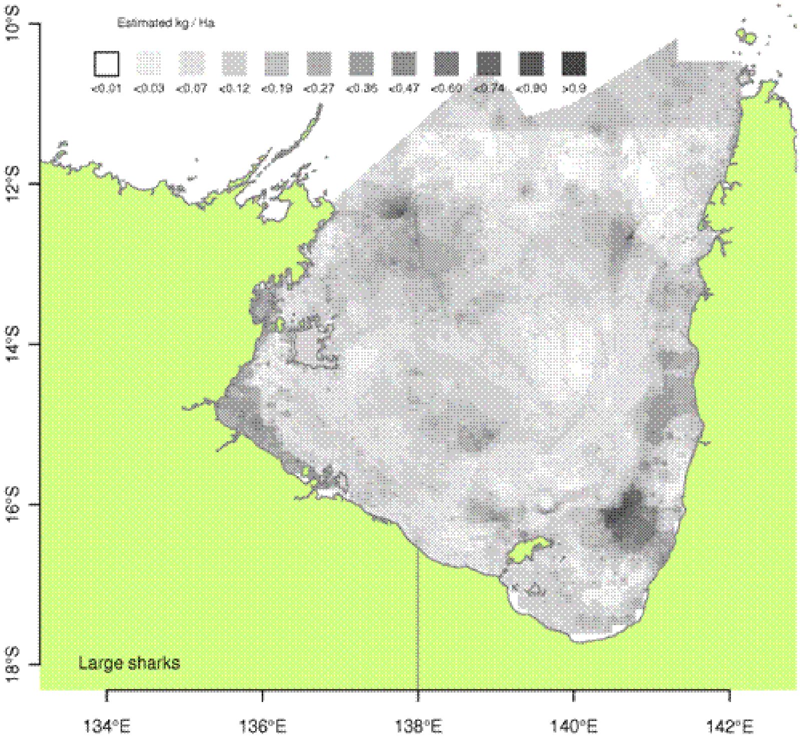

Estimated kg / Ha
<0
<0.01
<0.01
<0.02
<0.03
<0.05
<0.06
<0.08
<0.11
<0.13
<0.16
>0.2
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E
Gastropod herbivorous omnivore

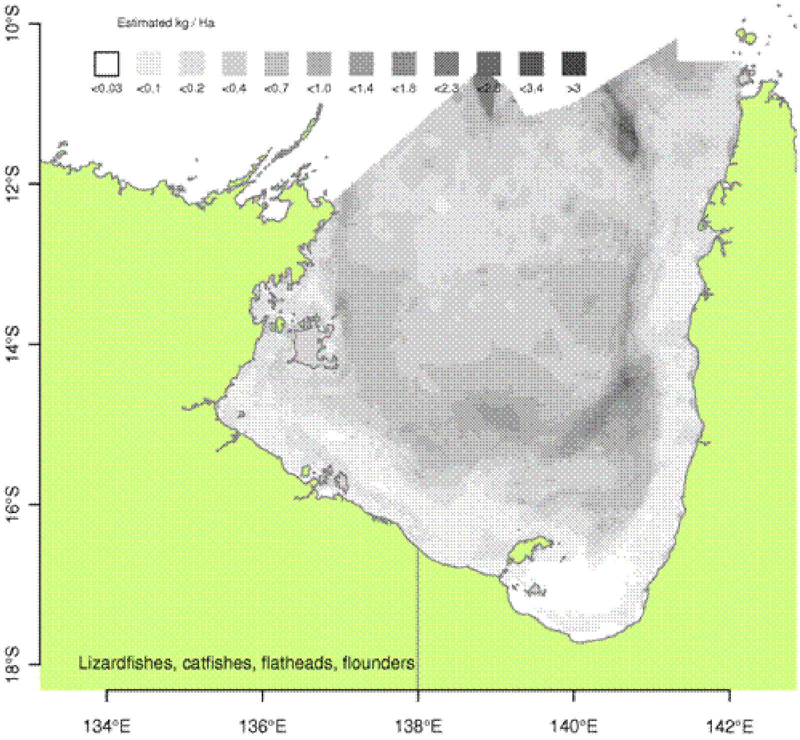

Estimated kg / Ha
<0.02
<0.1
<0.2
<0.4
<0.6
<0.9
<1.2
<1.6
<2.0
<2.4
<3.0
>3
Groupers, trevallies and cobia
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

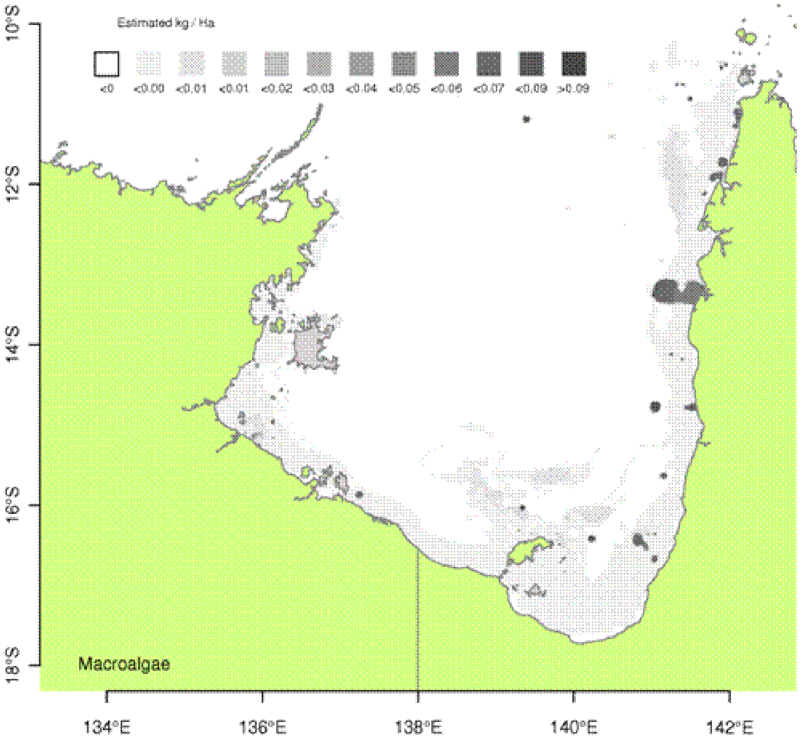

Estimated kg / Ha
<0
<0.00
<0.00
<0.00
<0.01
<0.01
<0.02
<0.02
<0.02
<0.03
<0.04
>0.04
Holothurians
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

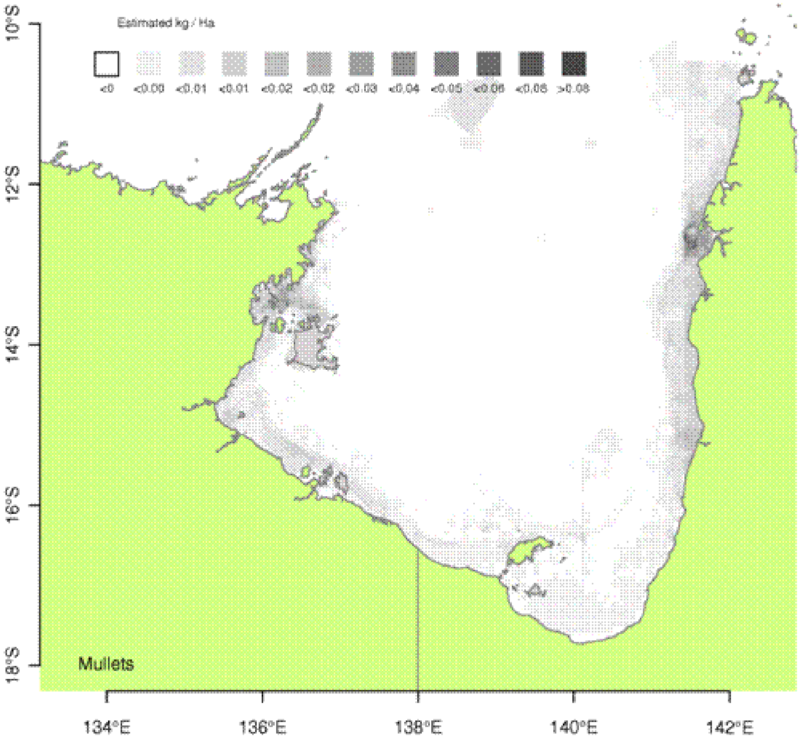

Estimated kg / Ha
<0.3
< 1
< 3
< 4
< 7
<10
<14
<18
<23
<28
<34
>34
Infaunal carnivores
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

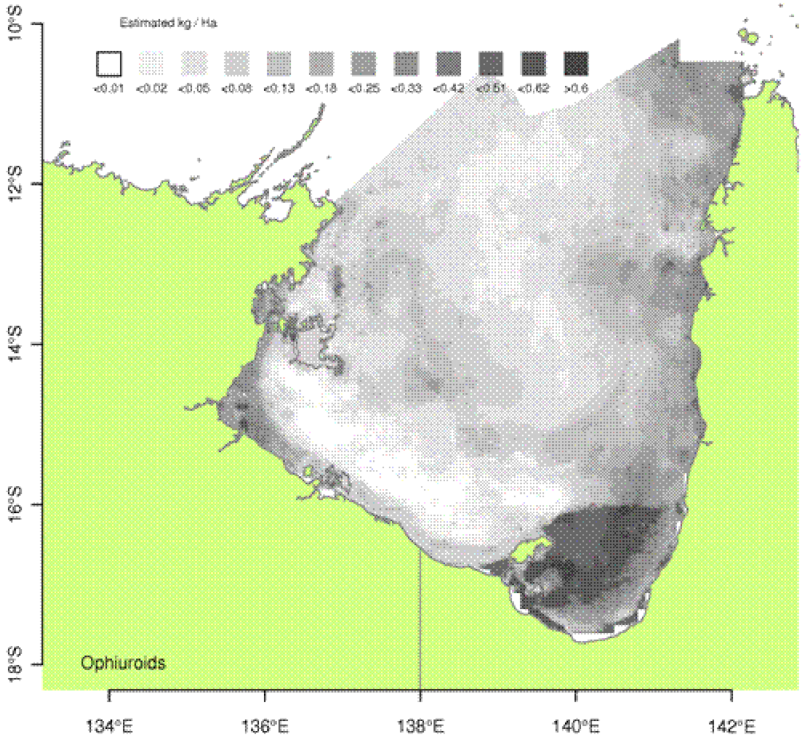

Estimated kg / Ha
<0
<0.01
<0.02
<0.04
<0.07
<0.10
<0.13
<0.17
<0.22
<0.27
<0.33
>0.3
Large gastropod carnivore
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

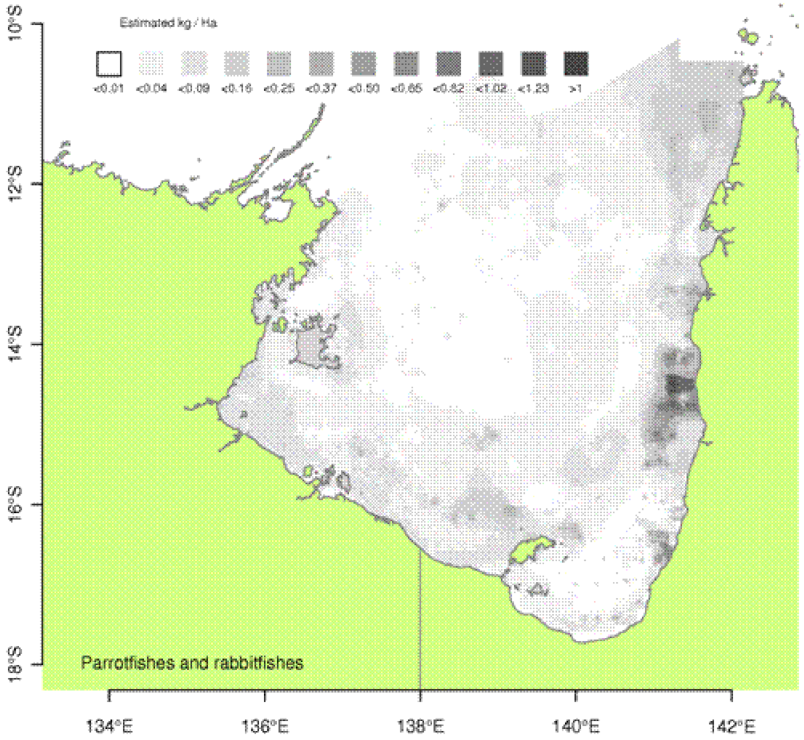

Estimated kg / Ha
<0.01
<0.03
<0.07
<0.12
<0.19
<0.27
<0.35
<0.47
<0.60
<0.74
<0.90
>0.9
Large sharks
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

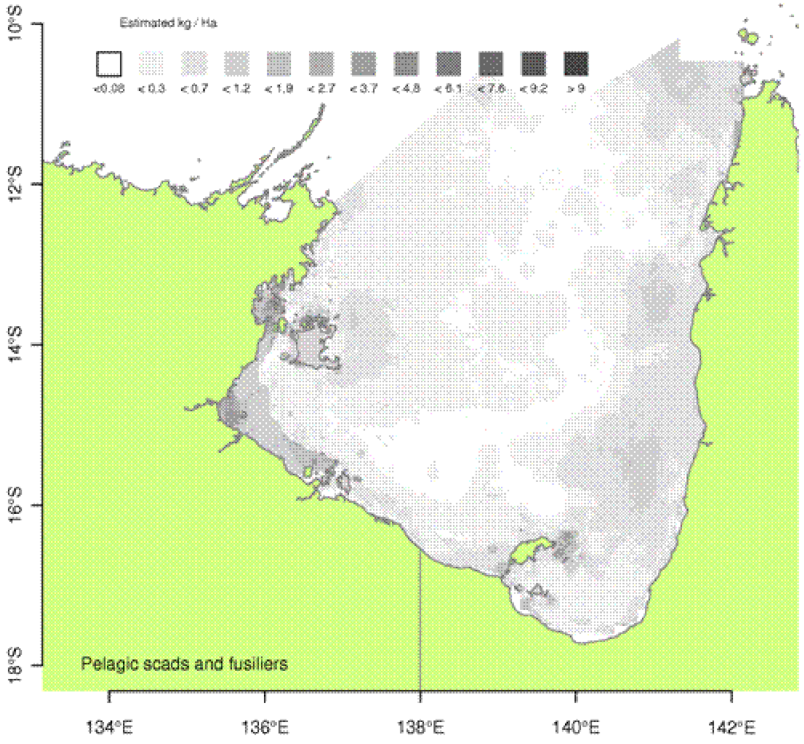

Estimated kg / Ha
<0.03
<0.1
<0.2
<0.4
<0.7
<1.0
<1.4
<1.8
<2.3
<2.8
<3.4
>3
Lizardfishes, catfishes, flatheads, flounders
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

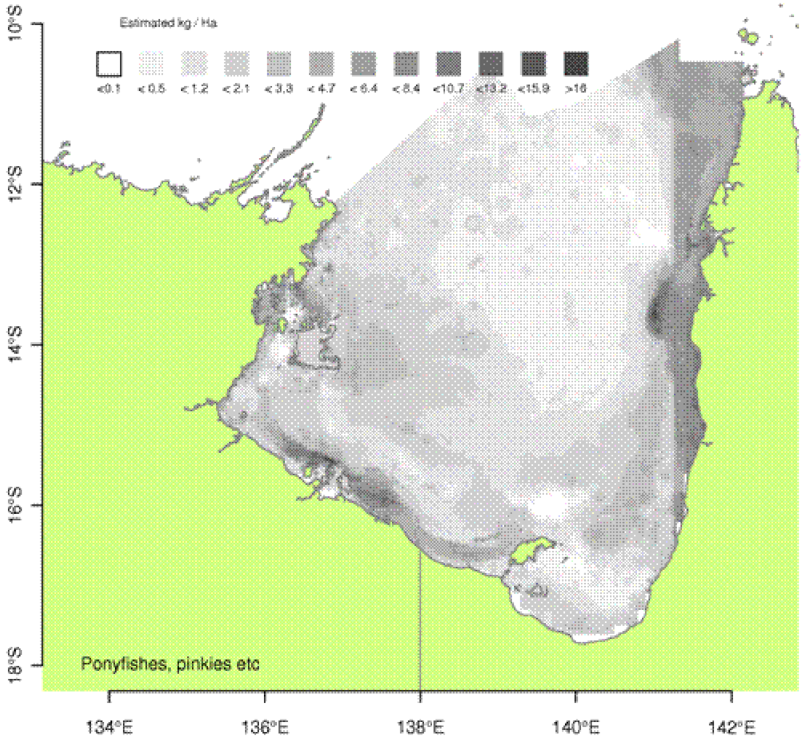

Estimated kg / Ha
<0
<0.00
<0.01
<0.01
<0.02
<0.03
<0.04
<0.05
<0.06
<0.07
<0.09
>0.09
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E
Macroalgae

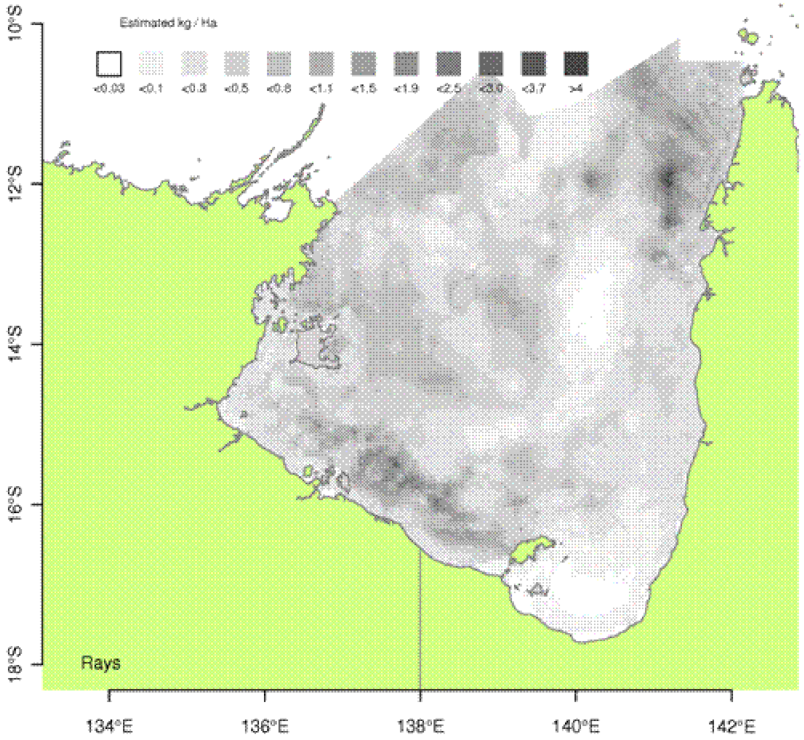

Estimated kg / Ha
<0
<0.00
<0.01
<0.01
<0.02
<0.02
<0.03
<0.04
<0.05
<0.06
<0.08
>0.08
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E
Mullets

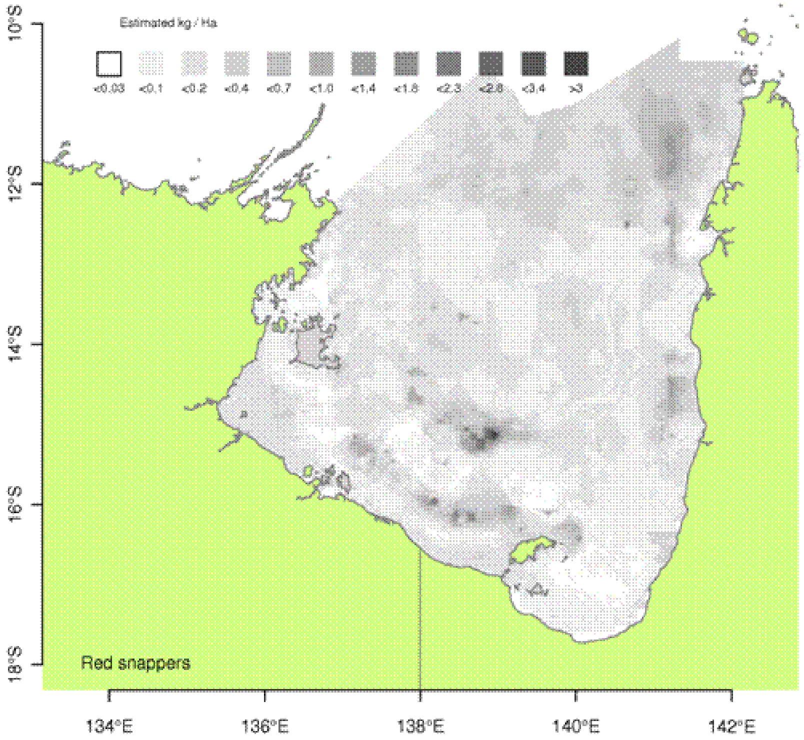

Estimated kg / Ha
<0.01
<0.02
<0.05
<0.08
<0.13
<0.18
<0.25
<0.33
<0.42
<0.51
<0.62
>0.6
Ophiuroids
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

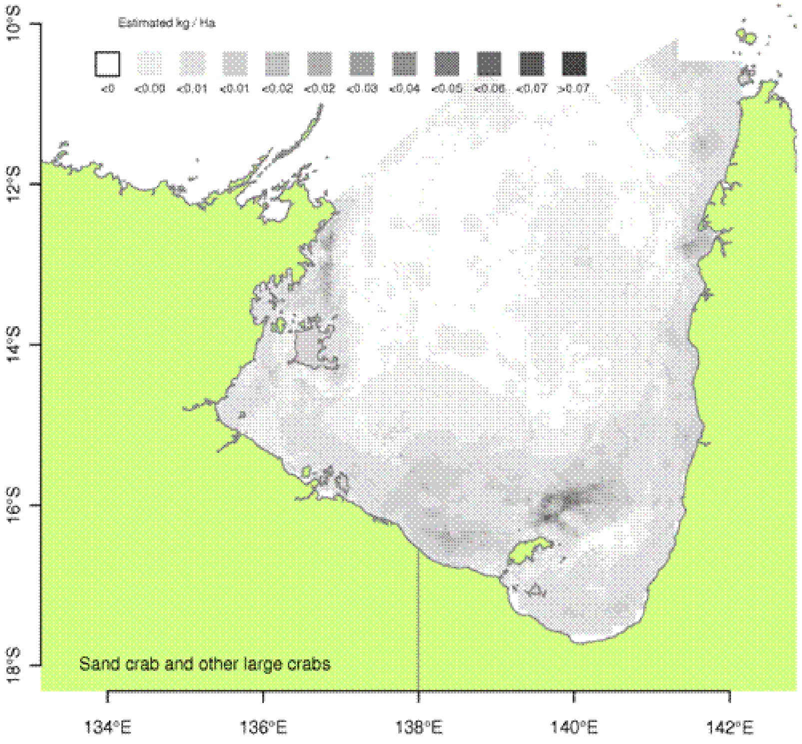

Estimated kg / Ha
<0.01
<0.04
<0.09
<0.16
<0.25
<0.37
<0.50
<0.65
<0.82
<1.02
<1.23
>1
Parrotfishes and rabbitfishes
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

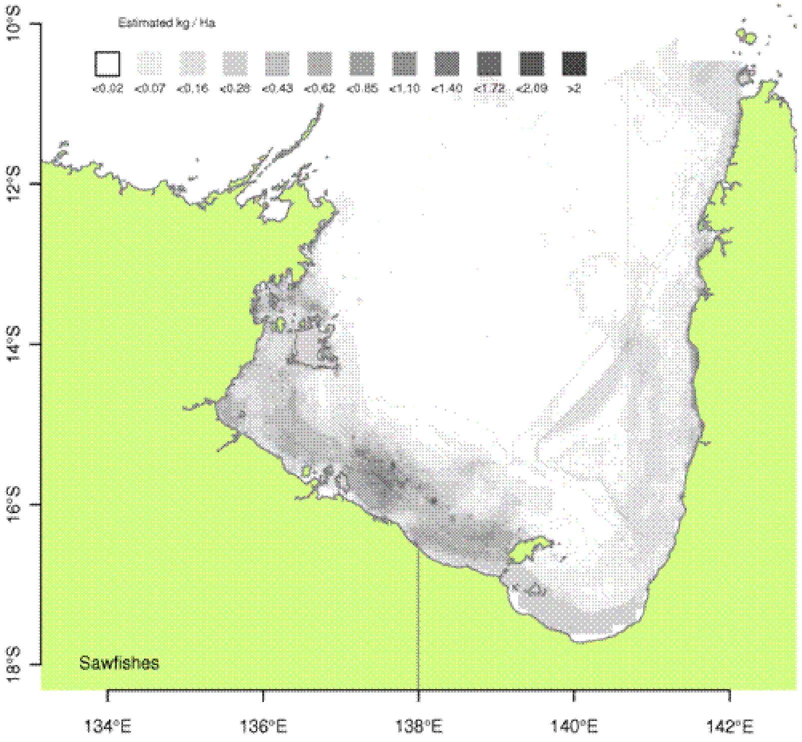

Estimated kg / Ha
<0.08
< 0.3
< 0.7
< 1.2
< 1.9
< 2.7
< 3.7
< 4.8
< 6.1
< 7.6
< 9.2
> 9
Pelagic scads and fusiliers
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

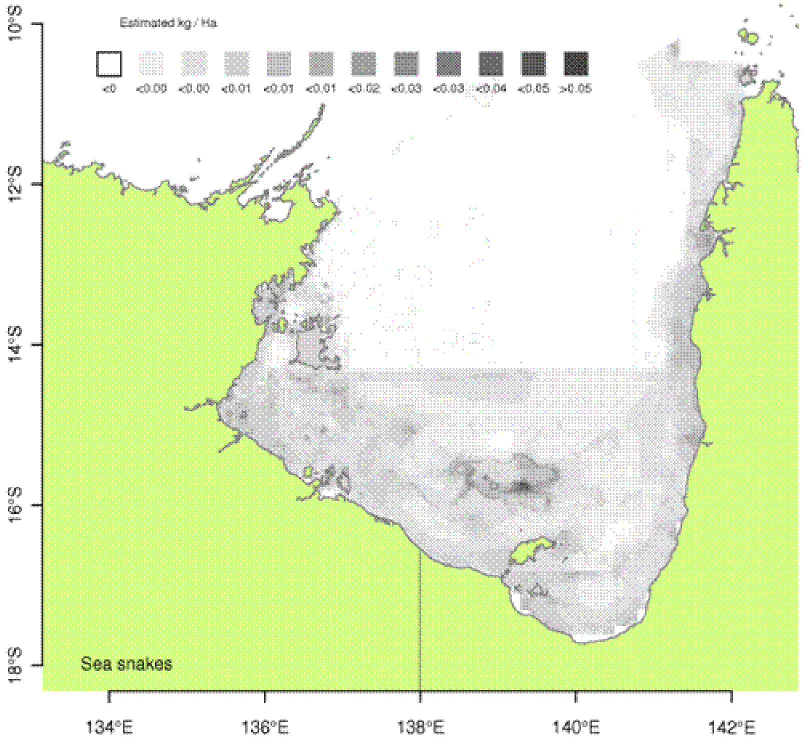

Estimated kg / Ha
<0.1
< 0.5
< 1.2
< 2.1
< 3.3
< 4.7
< 6.4
< 8.4
<10.7
<13.2
<15.9
>16
Ponyfishes, pinkies etc
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E

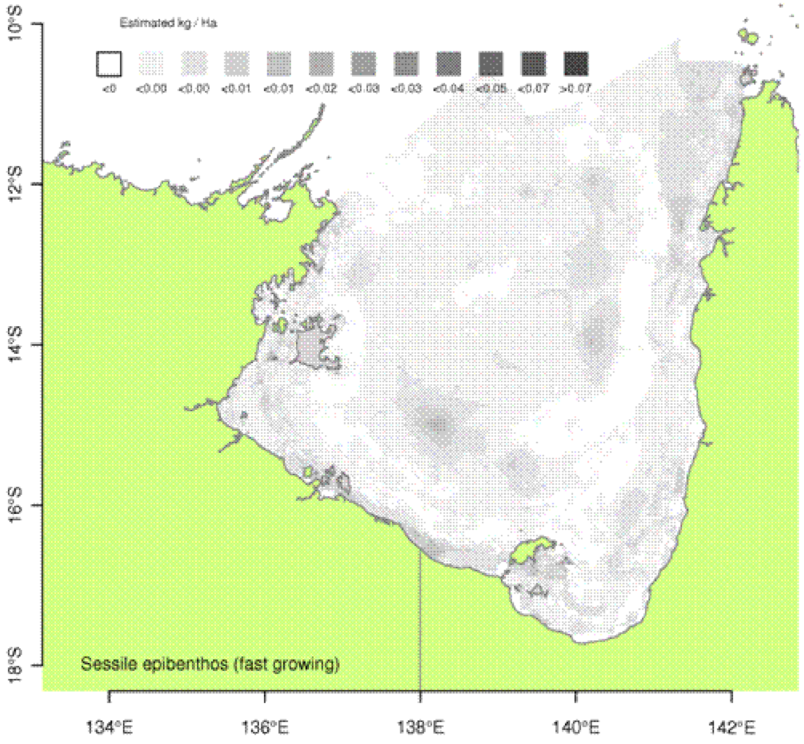

Estimated kg / Ha
<0.03
<0.1
<0.3
<0.5
<0.8
<1.1
<1.5
<1.9
<2.5
<3.0
<3.7
>4
10°S
12°S
14°S
16°S
18°S
Rays
134°E
136°E
138°E
140°E
142°E

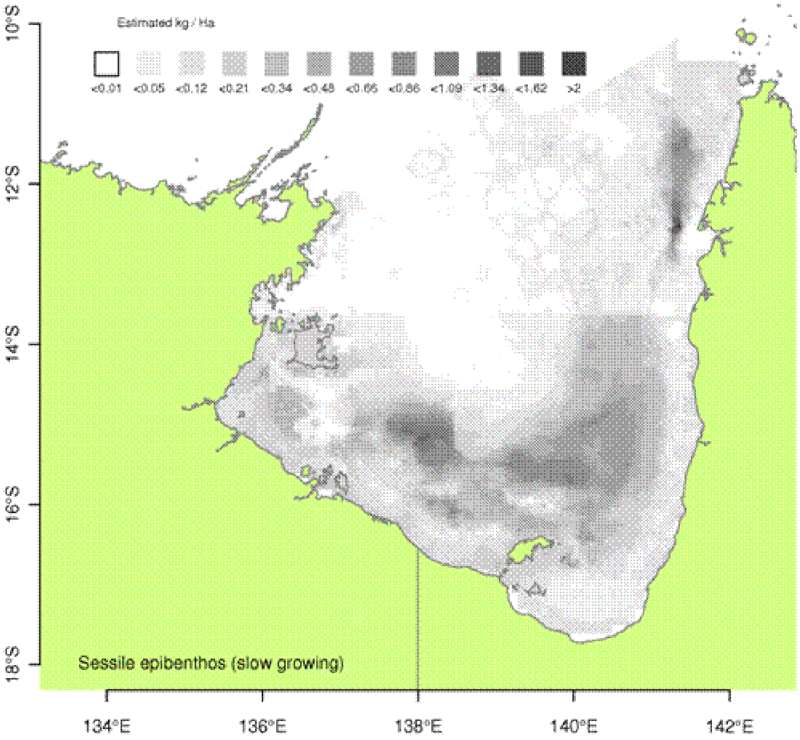

Estimated kg / Ha
<0.03
<0.1
<0.2
<0.4
<0.7
<1.0
<1.4
<1.8
<2.3
<2.8
<3.4
>3
10°S
12°S
14°S
16°S
18°S
Red snappers
134°E
136°E
138°E
140°E
142°E

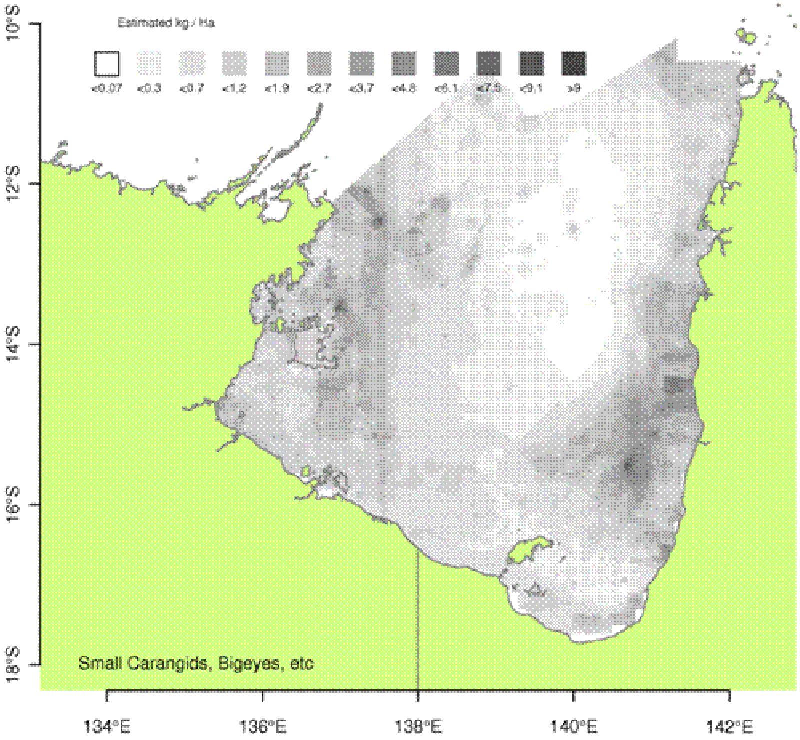

Estimated kg / Ha
<0
<0.00
<0.01
<0.01
<0.02
<0.02
<0.03
<0.04
<0.05
<0.06
<0.07
>0.07
10°S
12°S
14°S
16°S
18°S
Sand crab and other large crabs
134°E
136°E
138°E
140°E
142°E

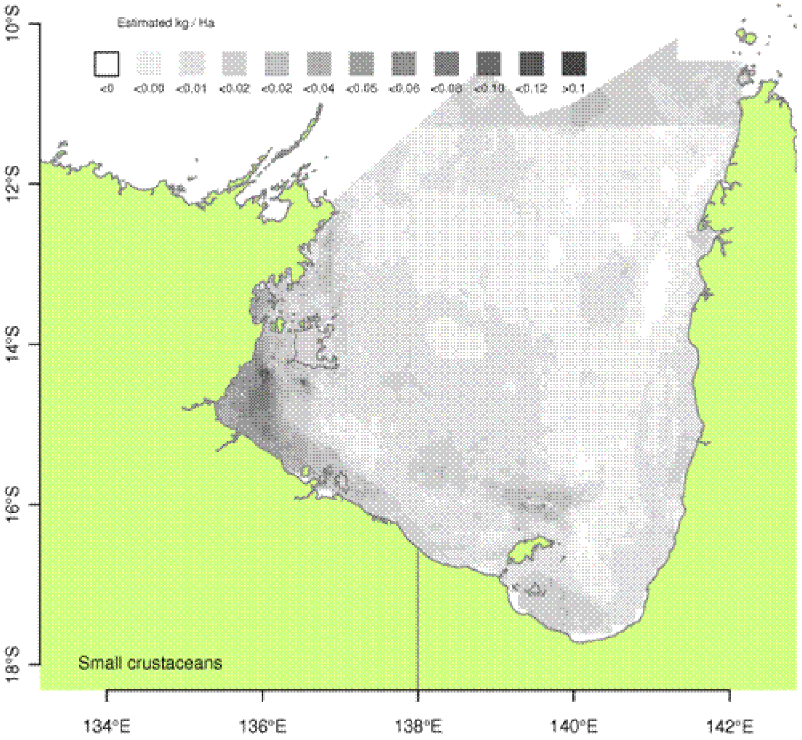

Estimated kg / Ha
<0.02
<0.07
<0.16
<0.28
<0.43
<0.62
<0.85
<1.10
<1.40
<1.72
<2.09
>2
10°S
12°S
14°S
16°S
18°S
Sawfishes
134°E
136°E
138°E
140°E
142°E

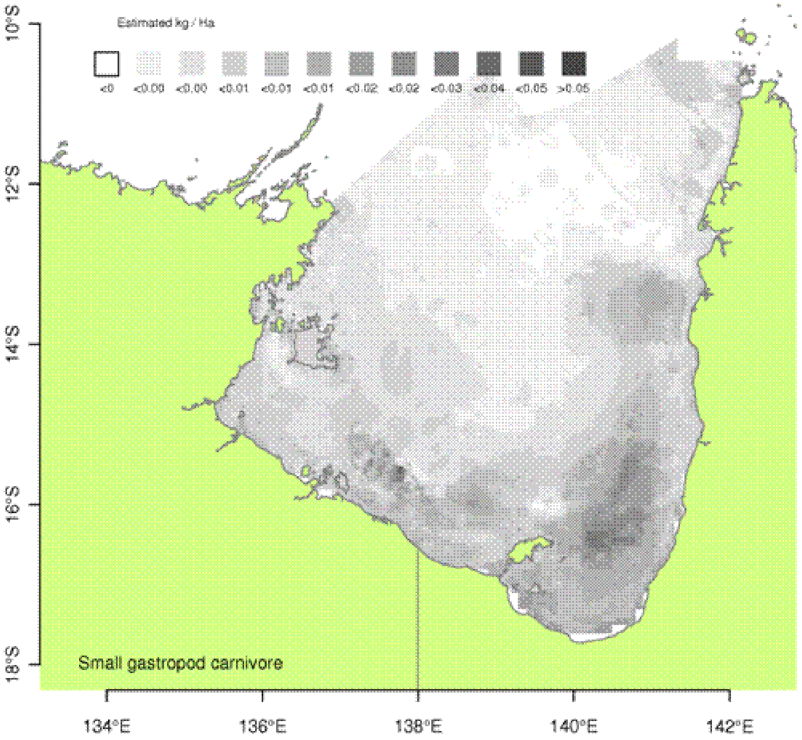

Estimated kg / Ha
<0
<0.00
<0.00
<0.01
<0.01
<0.01
<0.02
<0.03
<0.03
<0.04
<0.05
>0.05
Sea snakes
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E


Estimated kg / Ha
<0
<0.00
<0.00
<0.01
<0.01
<0.02
<0.03
<0.03
<0.04
<0.05
<0.07
>0.07
Sessile epibenthos (fast growing)
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E


Estimated kg / Ha
<0.01
<0.05
<0.12
<0.21
<0.34
<0.48
<0.65
<0.86
<1.09
<1.34
<1.62
>2
Sessile epibenthos (slow growing)
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E


Estimated kg / Ha
<0.07
<0.3
<0.7
<1.2
<1.9
<2.7
<3.7
<4.8
<6.1
<7.5
<9.1
>9
Small Carangids, Bigeyes, etc
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E


Estimated kg / Ha
<0
<0.00
<0.01
<0.02
<0.02
<0.04
<0.05
<0.06
<0.08
<0.10
<0.12
>0.1
10°S
12°S
14°S
16°S
18°S
Small crustaceans
134°E
136°E
138°E
140°E
142°E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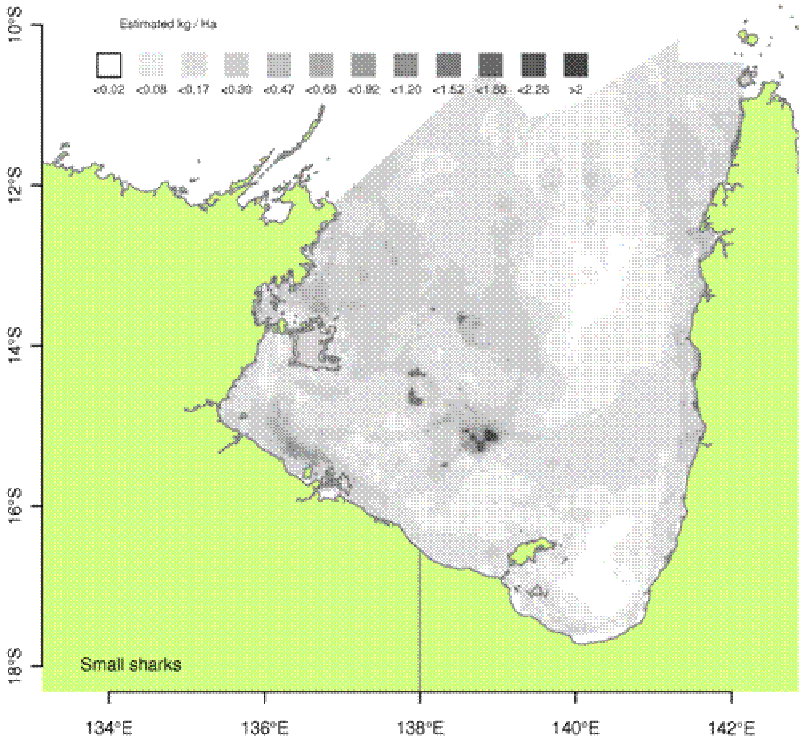
<0.01
<0.01
<0.01
<0.02
<0.02
<0.03
<0.04
<0.05
>0.05
10°S
12°S
14°S
16°S
18°S
Small gastropod carnivore
134°E
136°E
138°E
140°E
142°E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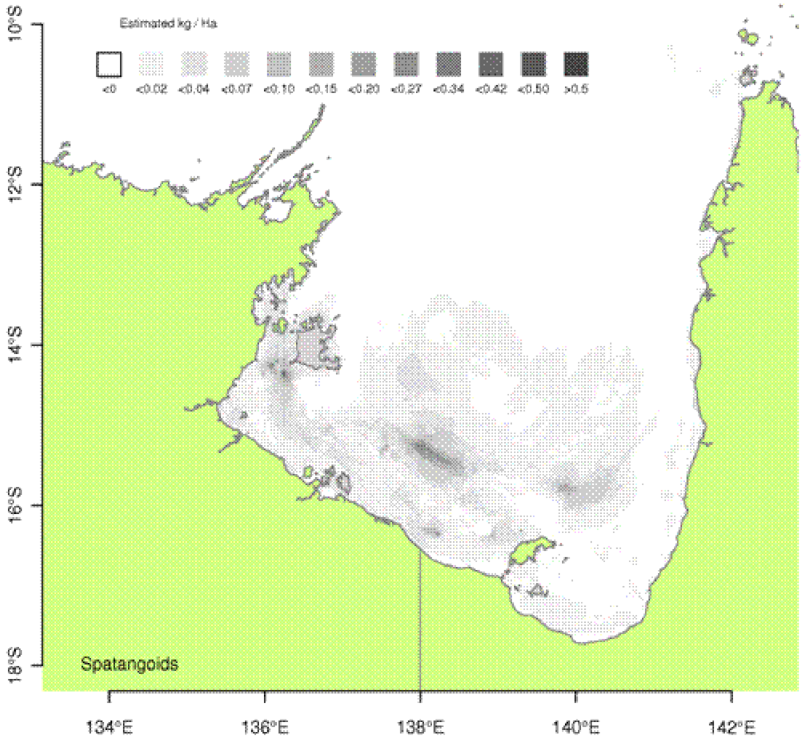
<0.47
<0.68
<0.92
<1.20
<1.52
<1.88
<2.26
>2
Small sharks
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E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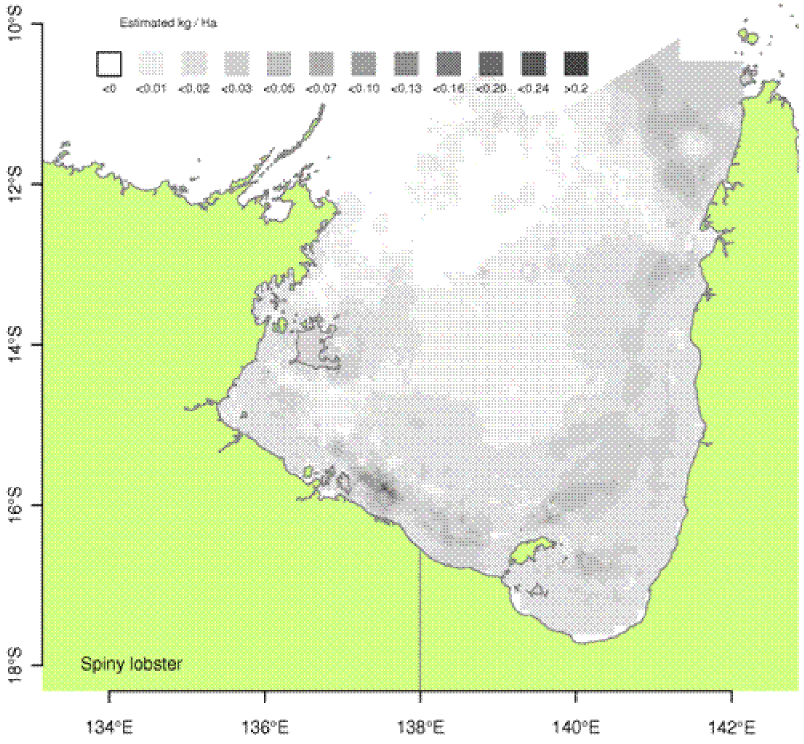
<0.10
<0.15
<0.20
<0.27
<0.34
<0.42
<0.50
>0.5
Spatangoids
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E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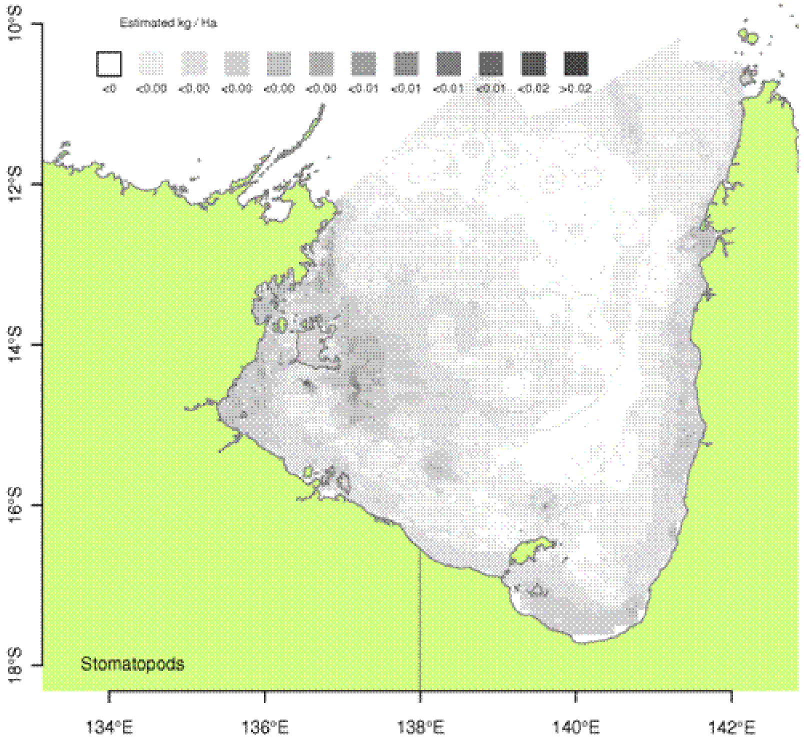
<0.02
<0.03
<0.05
<0.07
<0.10
<0.13
<0.16
<0.20
<0.24
>0.2
Spiny lobster
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E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<0.00
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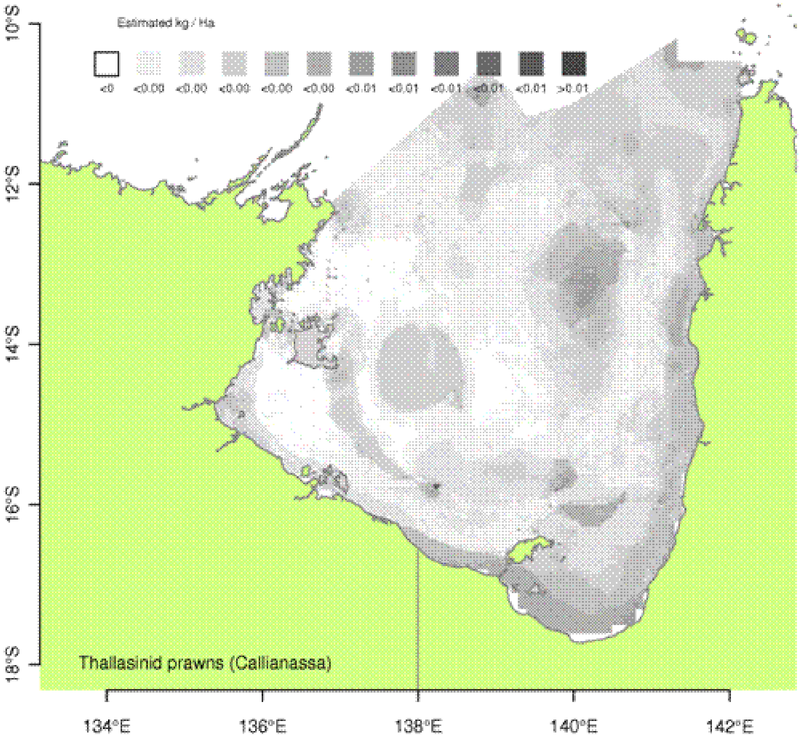
<0.00
<0.01
<0.01
<0.01
<0.01
<0.02
>0.02
Stomatopods
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E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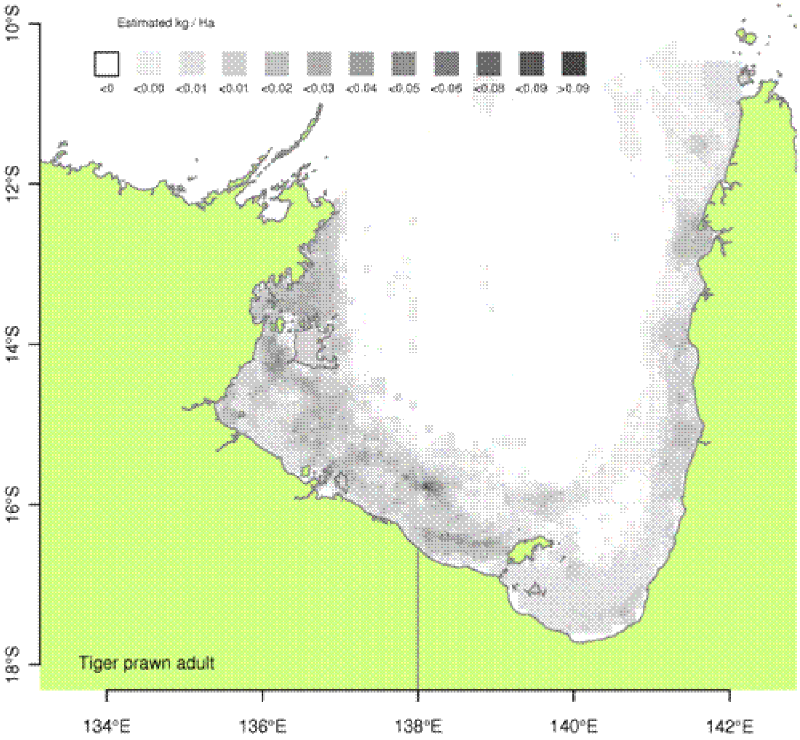
<0.00
<0.01
<0.01
<0.01
<0.01
<0.01
>0.01
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E
Thallasinid prawns (Callianassa)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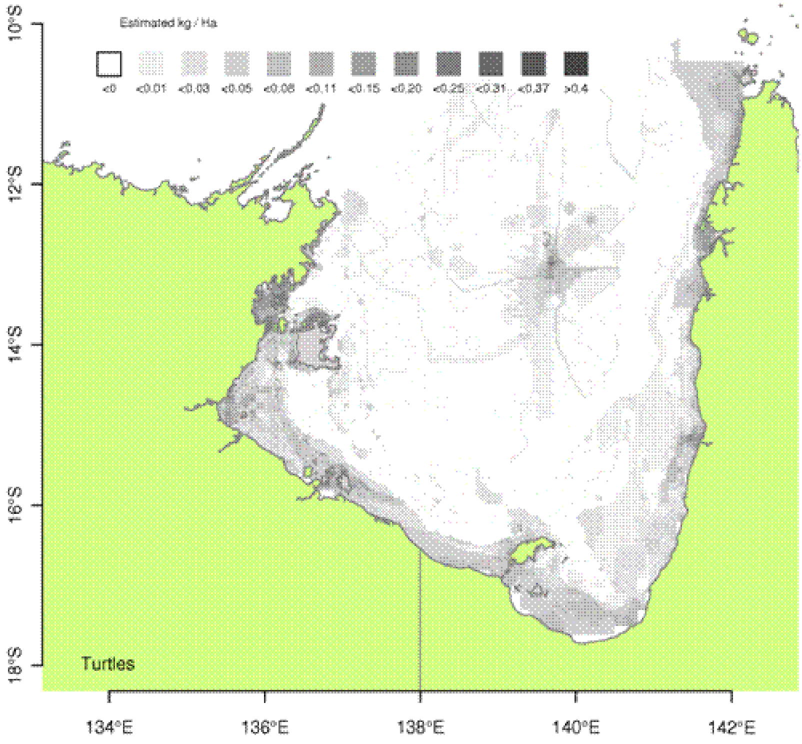
<0.00
<0.01
<0.01
<0.02
<0.03
<0.04
<0.05
<0.06
<0.08
<0.09
>0.09
10°S
12°S
14°S
16°S
18°S
134°E
136°E
138°E
140°E
142°E
Tiger prawn adult


Estimated kg / Ha
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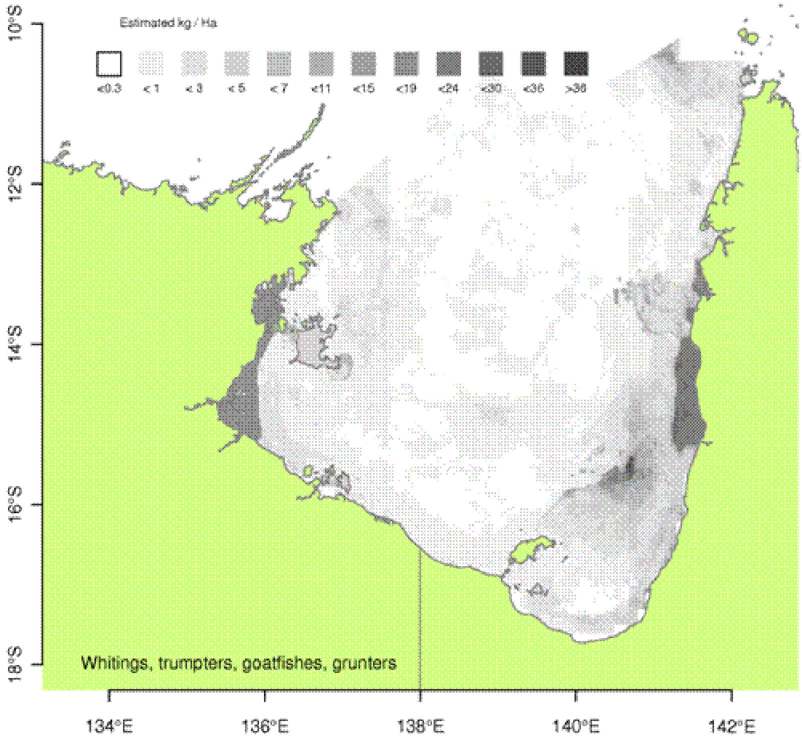
<0.01
<0.03
<0.05
<0.08
<0.11
<0.15
<0.20
<0.25
<0.31
<0.37
>0.4
10°S
12°S
14°S
16°S
18°S
Turtles
134°E
136°E
138°E
140°E
142°E


Estimated kg / Ha
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< 1
< 3
< 5
< 7
<11
<15
<19
<24
<30
<36
>36
10°S
12°S
14°S
16°S
18°S
Whitings, trumpters, goatfishes, grunters
134°E
136°E
138°E
140°E
142°E

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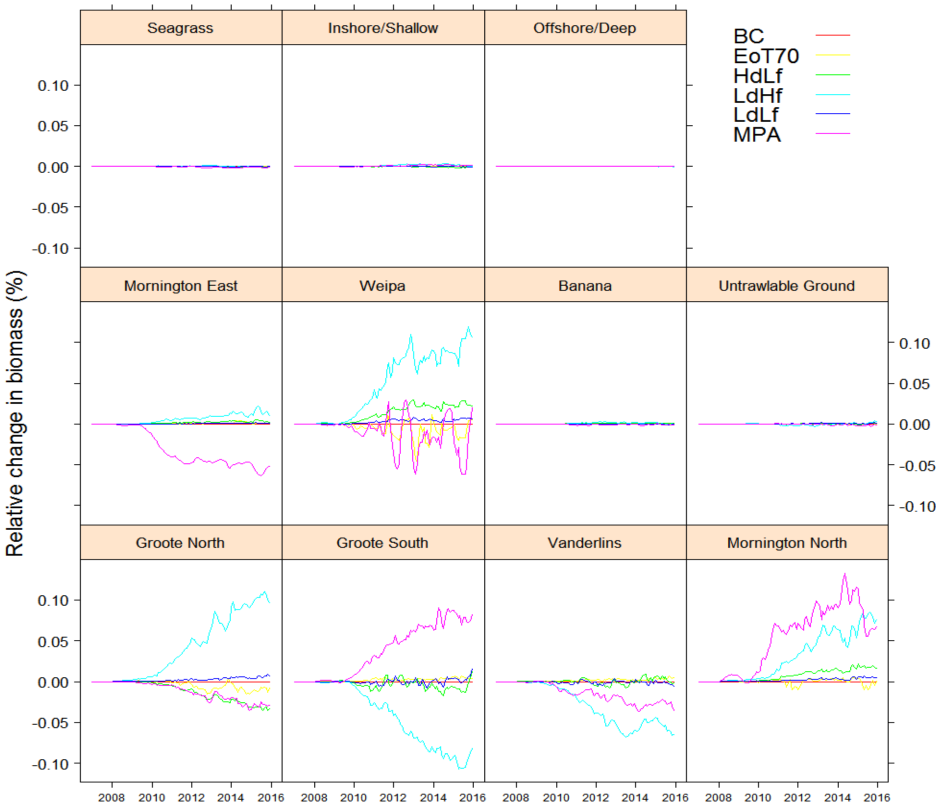
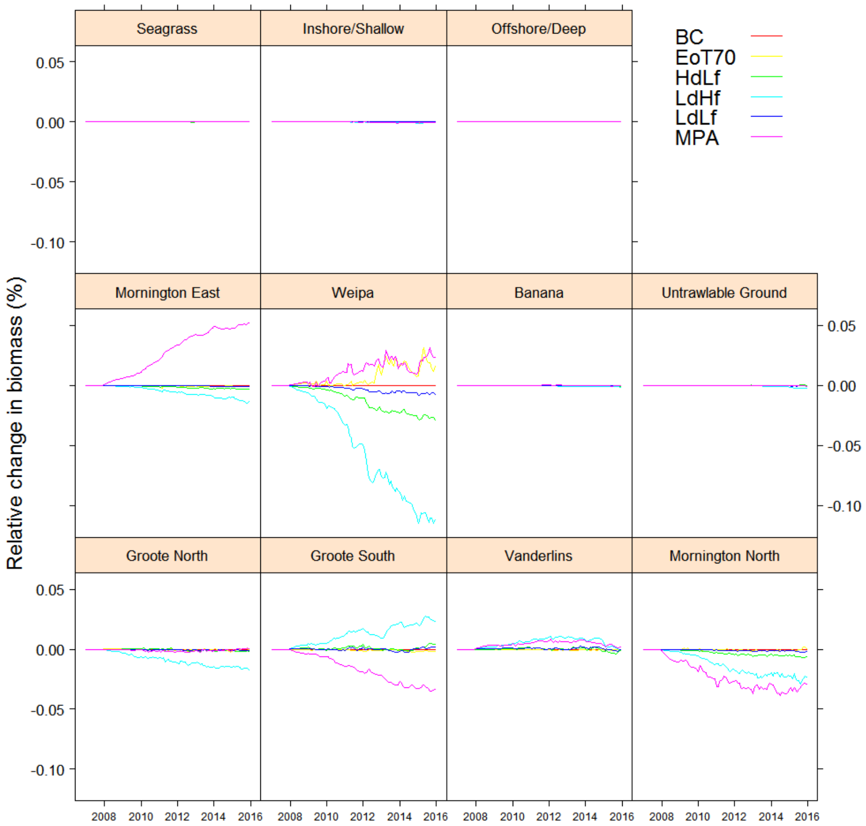
APPENDIX 13 . TEMPORAL (A) AND SPATIAL (B) PREDICTIONS OF RELATIVE BIOMASS IN RELATION TO THE BASE CASE (BC) FOR ALL 53 FUNCTIONAL GROUPS.

13.1 TEMPORAL CHANGES IN THE PREDICTED RELATIVE BIOMASS FOR ALL SIX MODELLED SCENARIOS IN RELATION TO THE BASE CASE (BC)



Annelids



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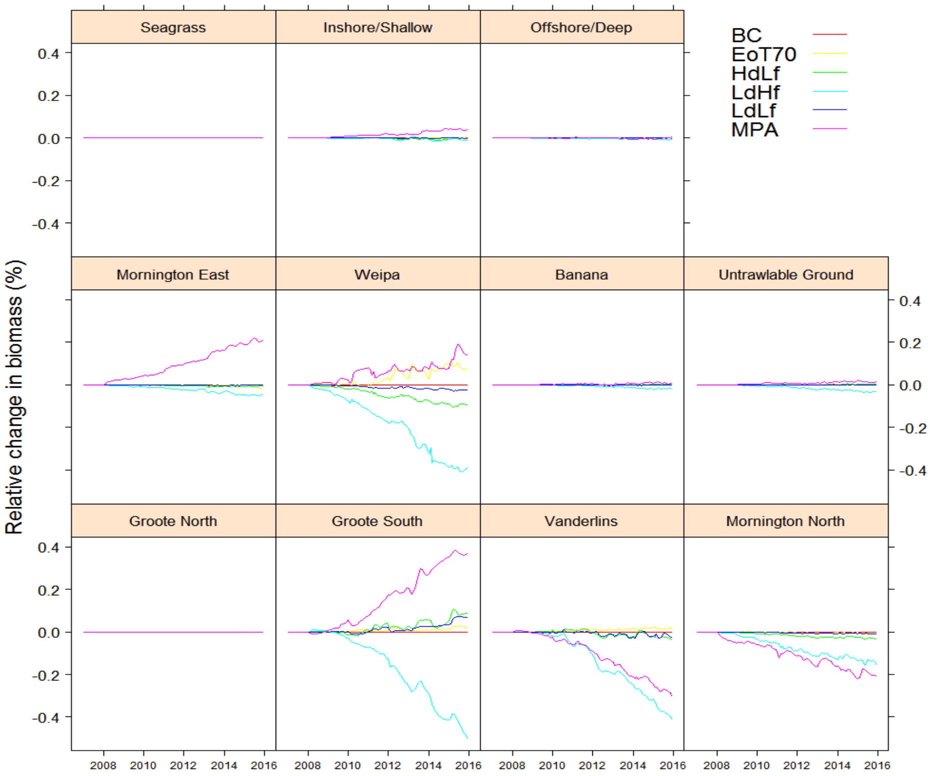
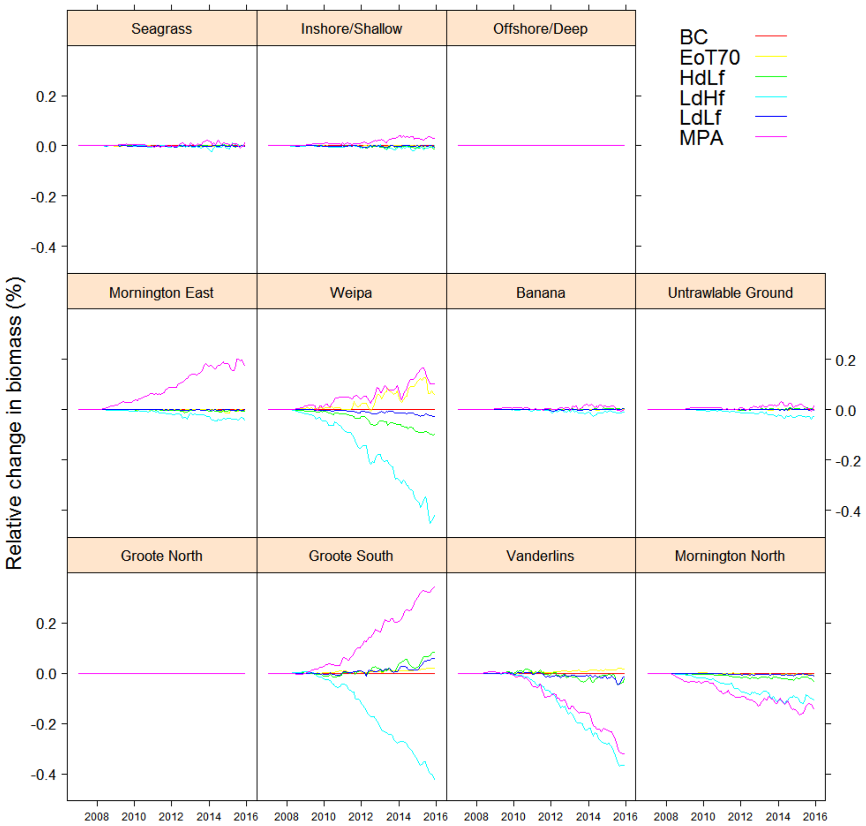
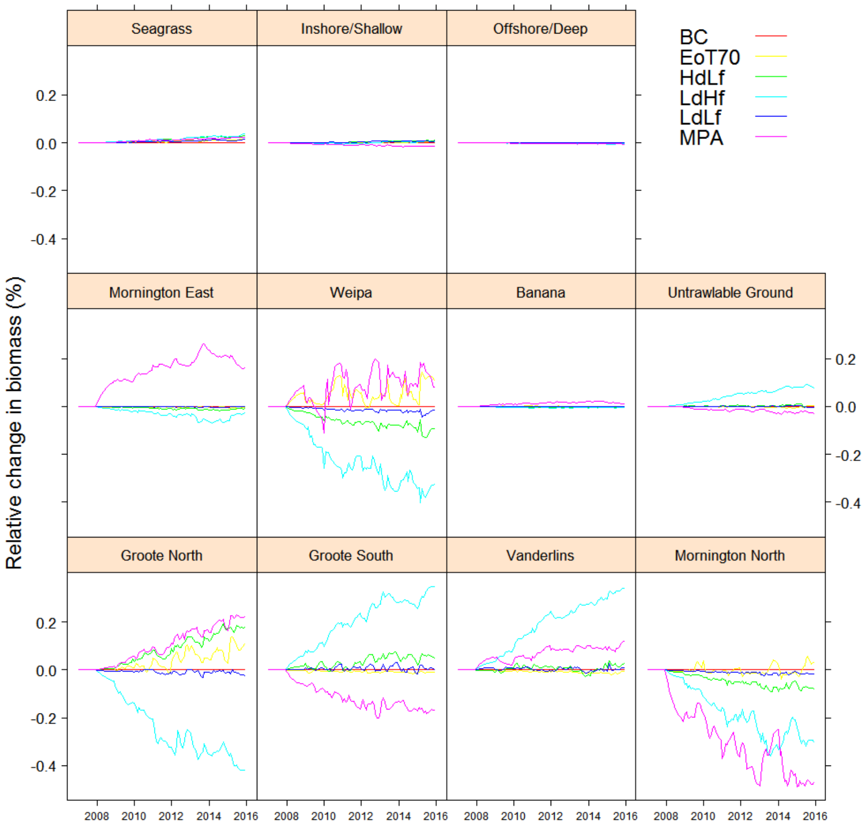


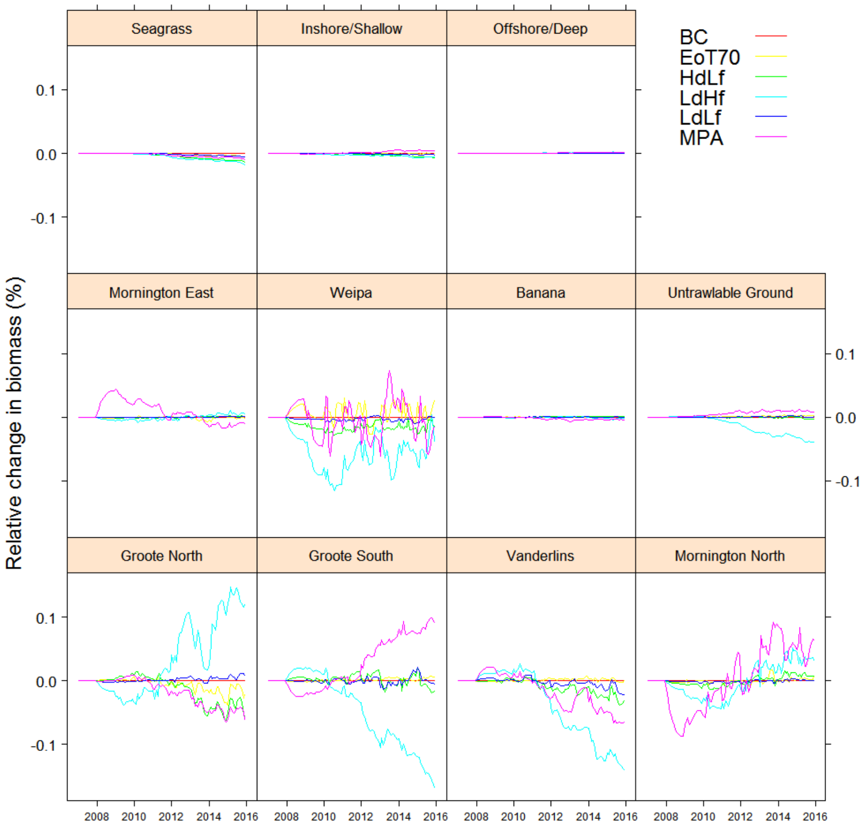
Banana Prawn Adult



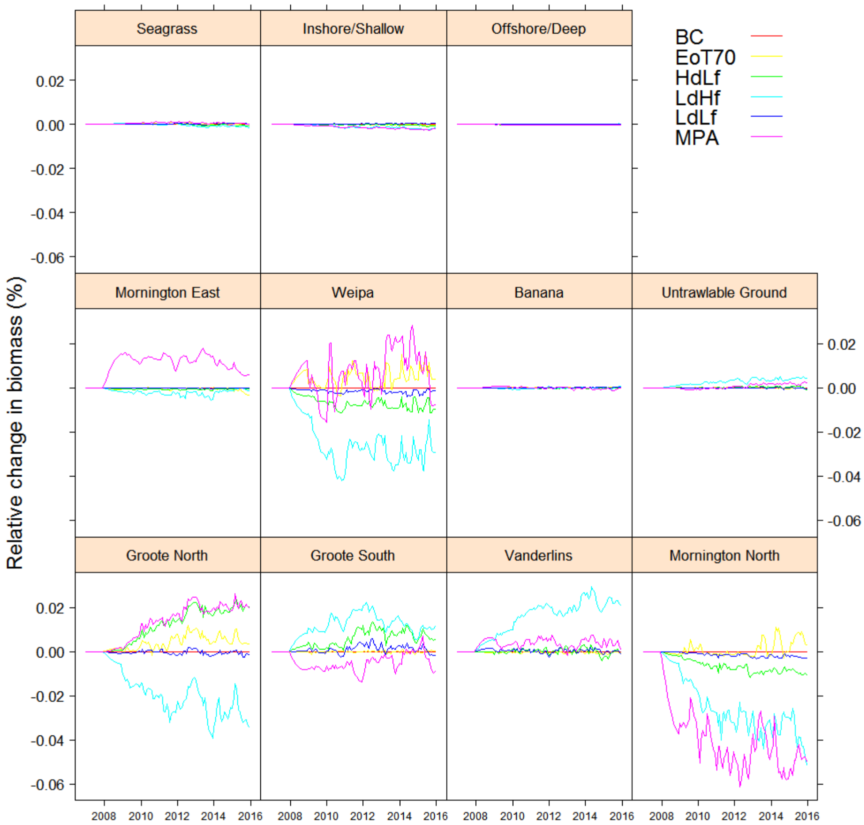
Banana Prawn juvenile



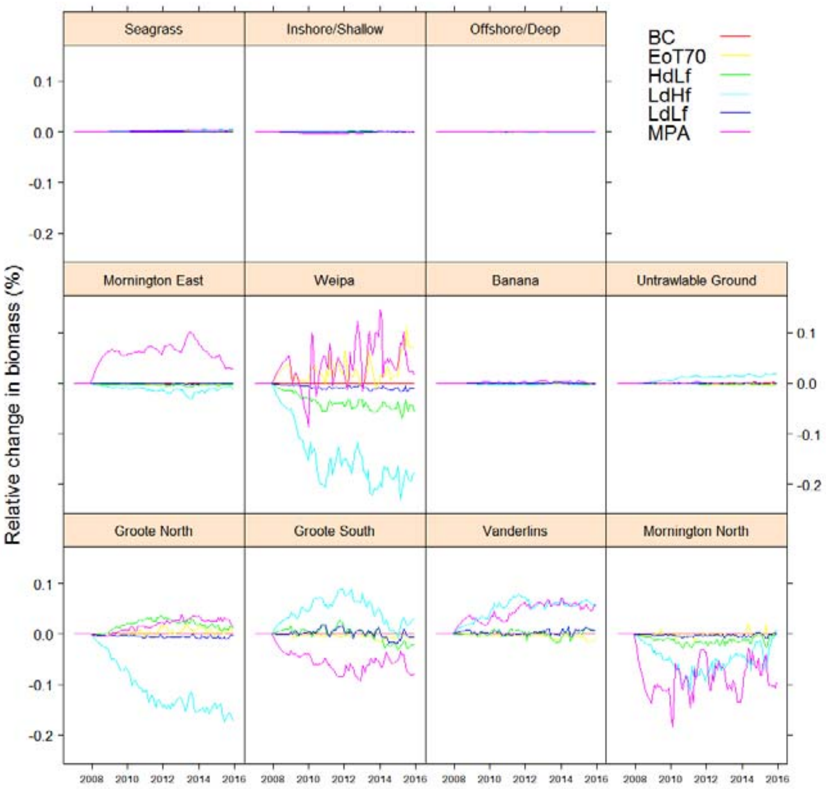
Benthic carnivores Fish



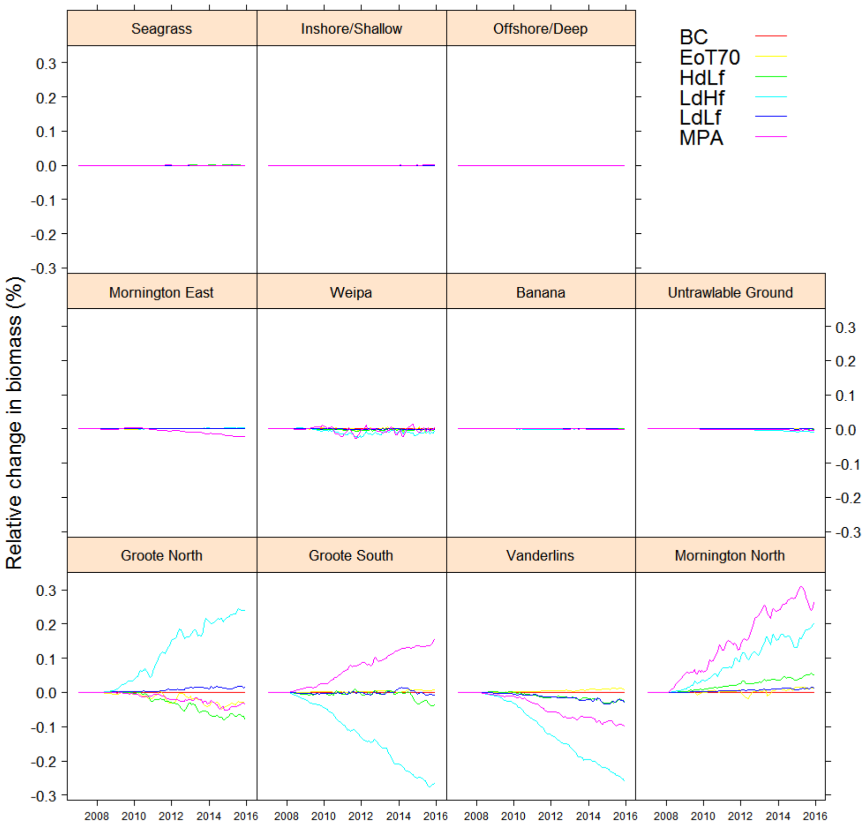
Benthic Invertebrate Feeders Fish



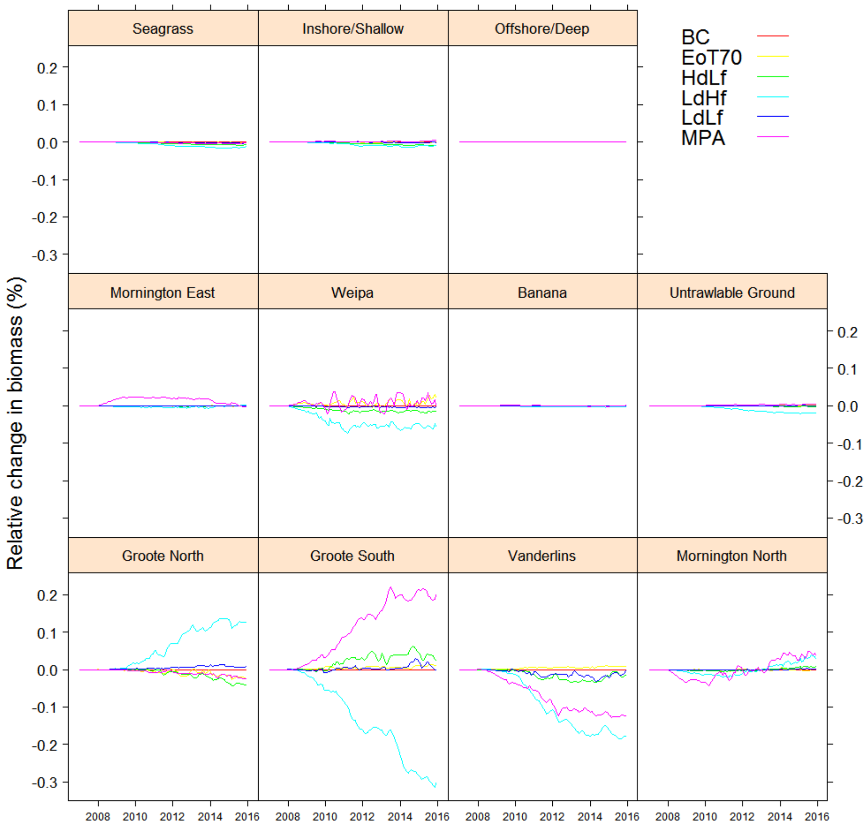
Benthopelagic Carnivores Fish



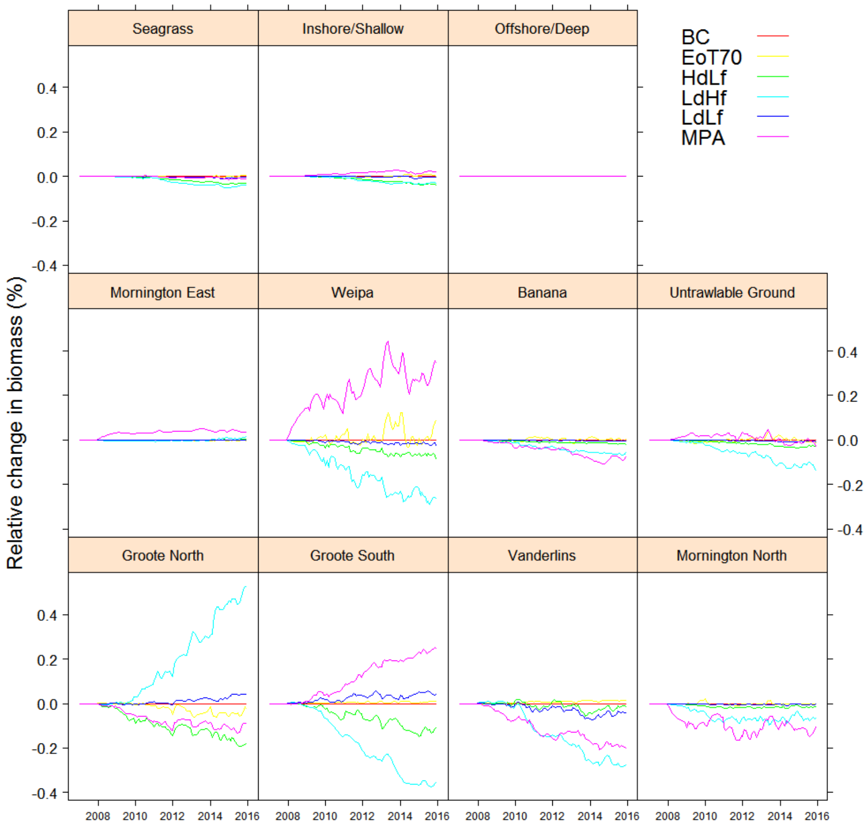
Benthopelagic Invert Feeders Fish



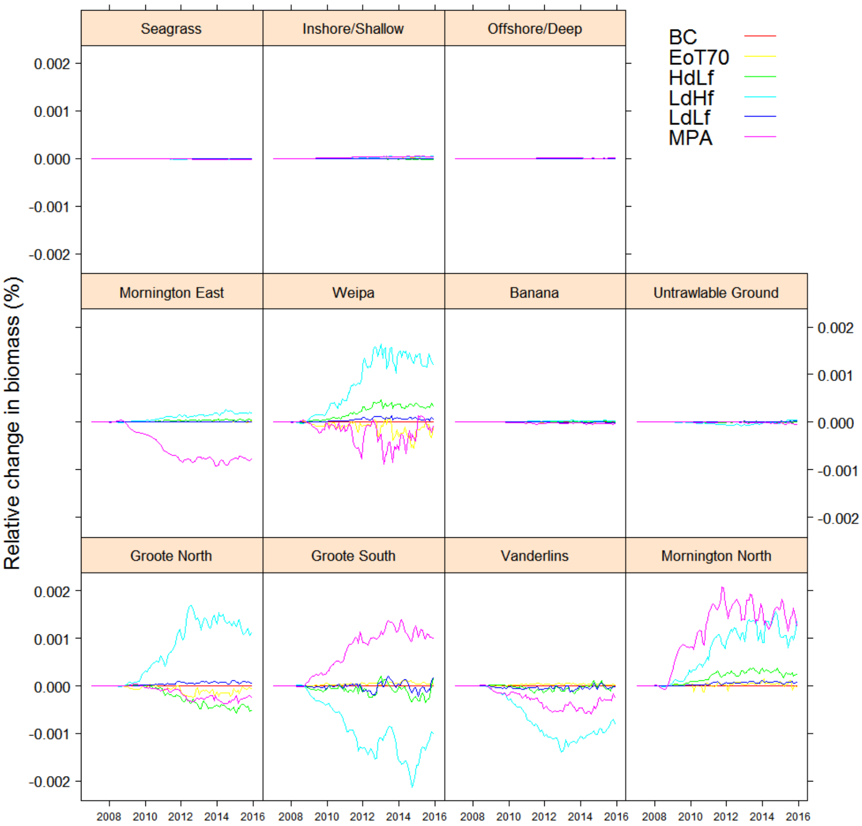
Bivalves



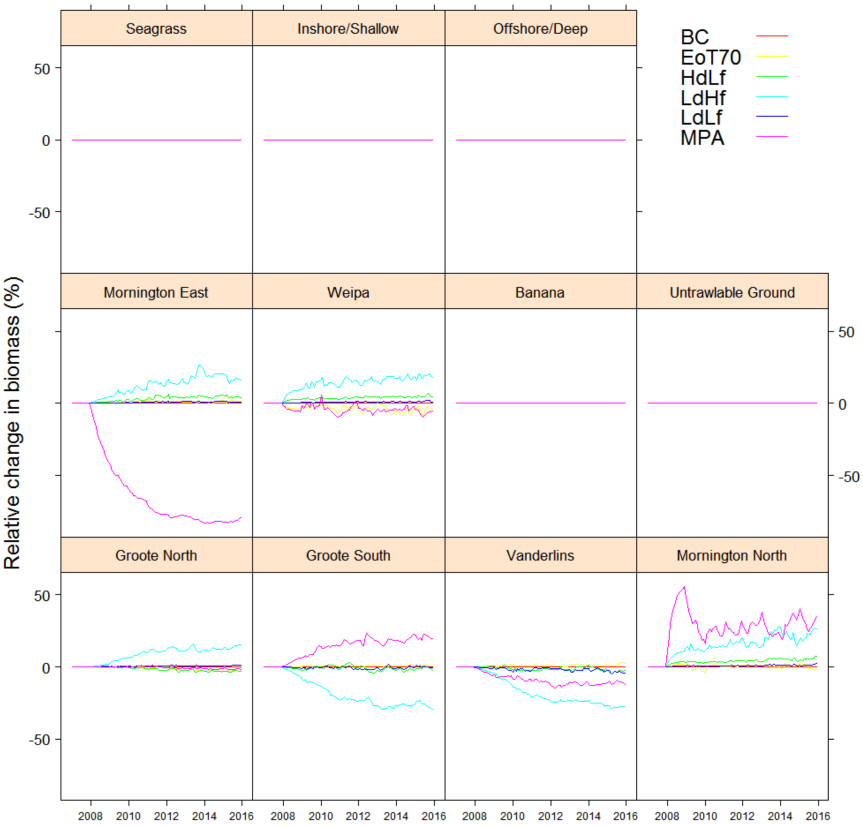
Cephalopods



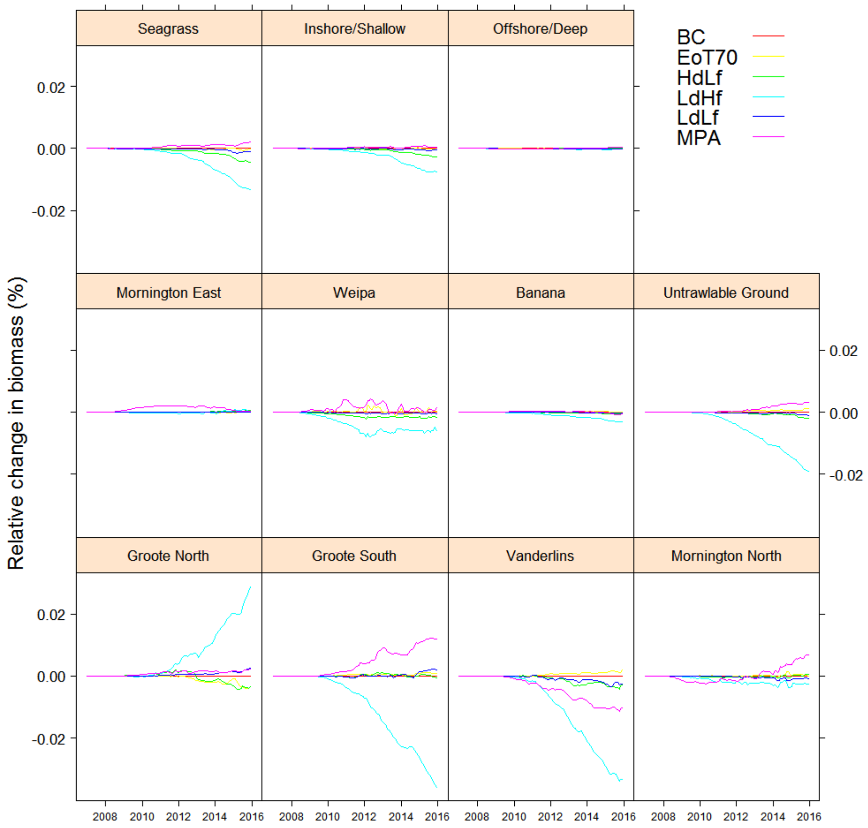
Detritivores Fish



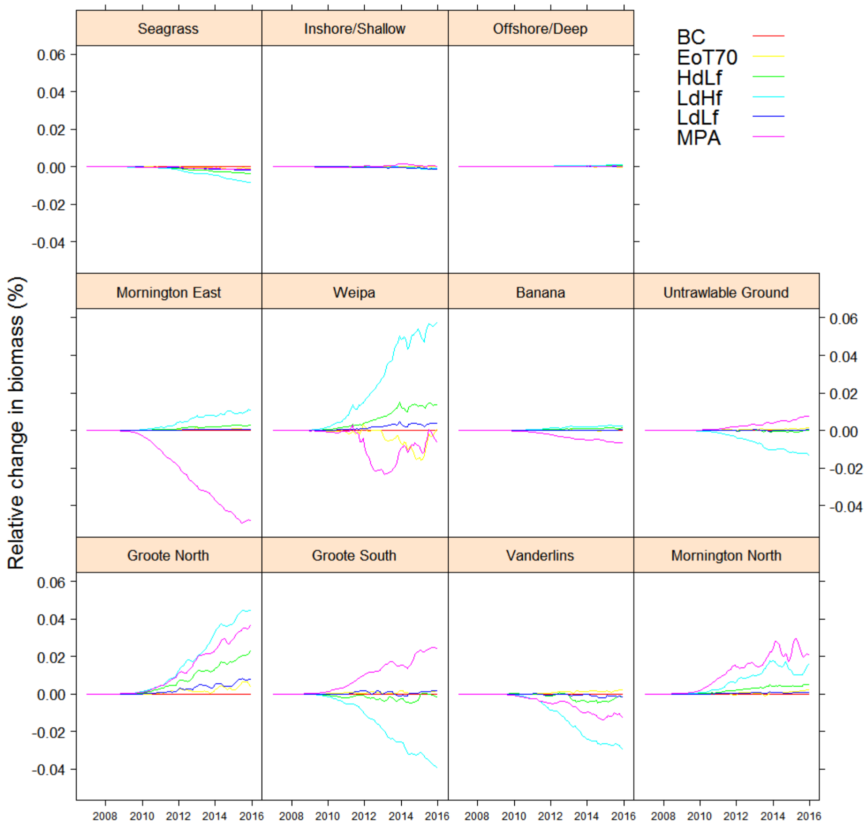
Detritus



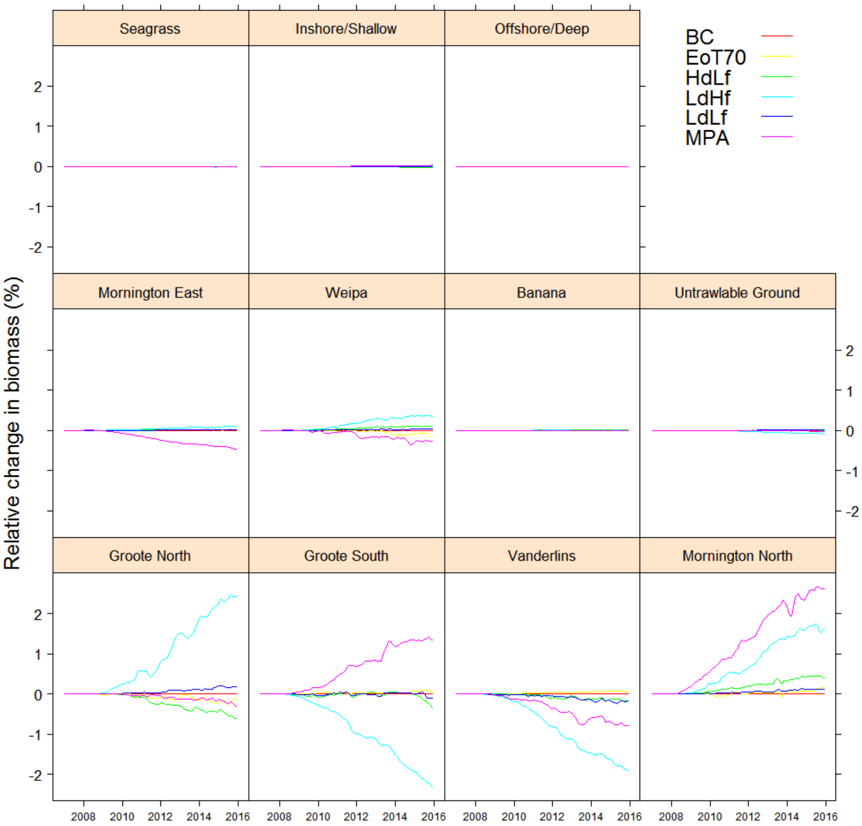
Discarded Bycatch



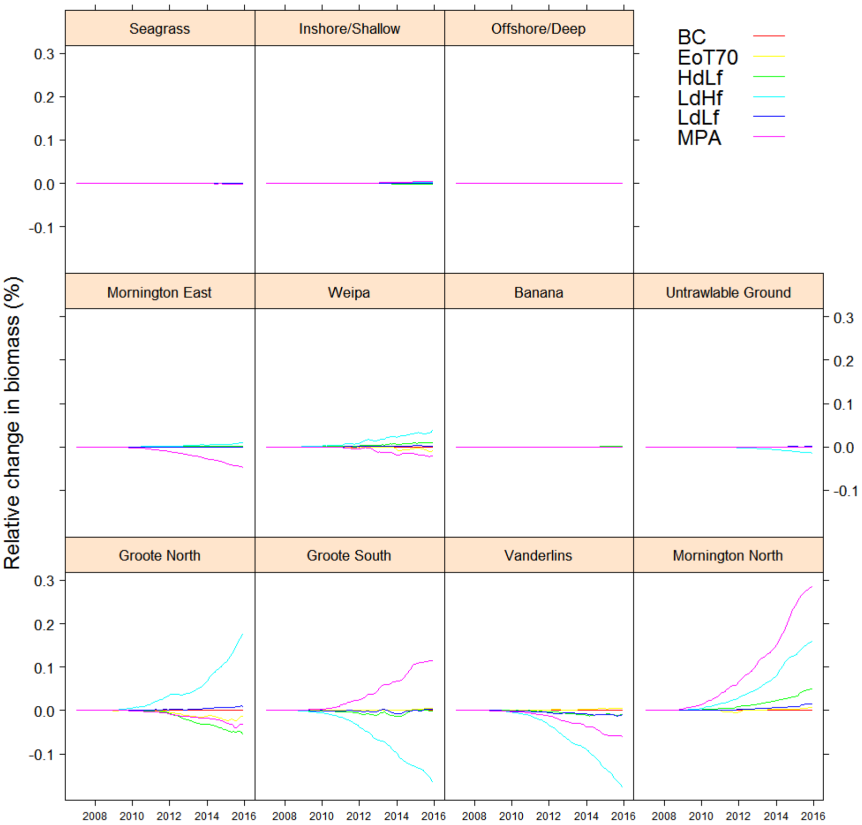
Dolphins



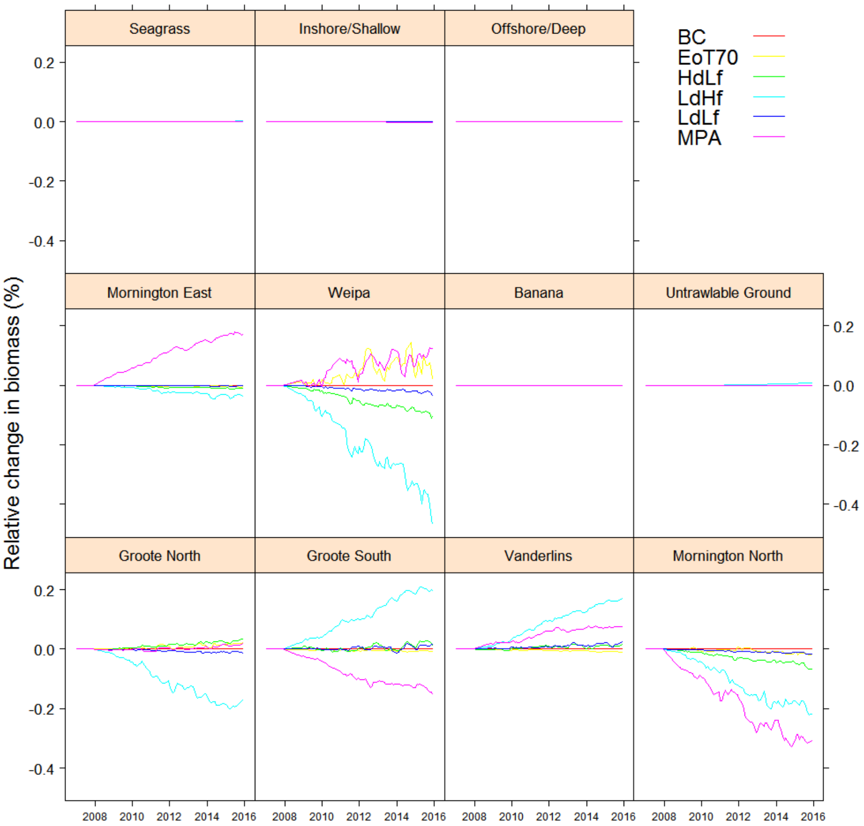
Dugongs



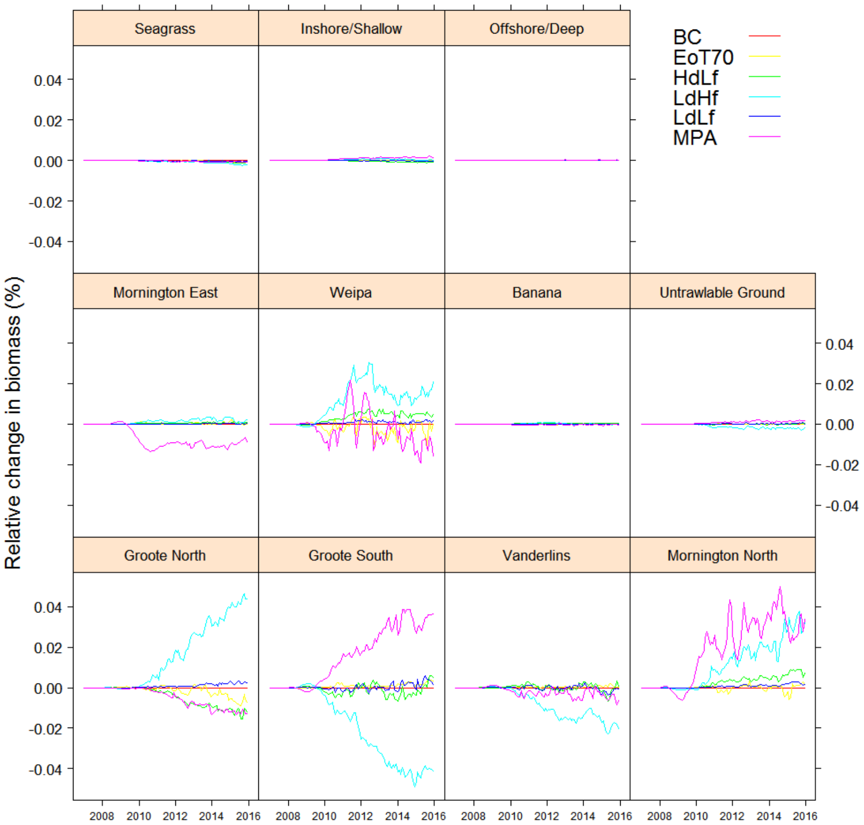
Echinoids



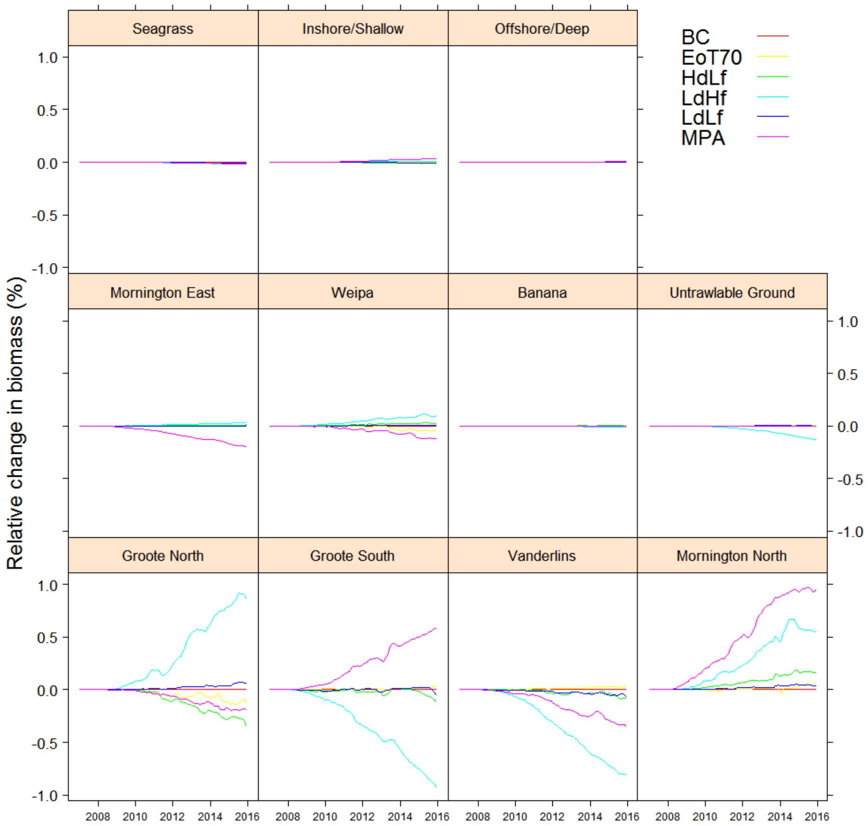
Formaninfera



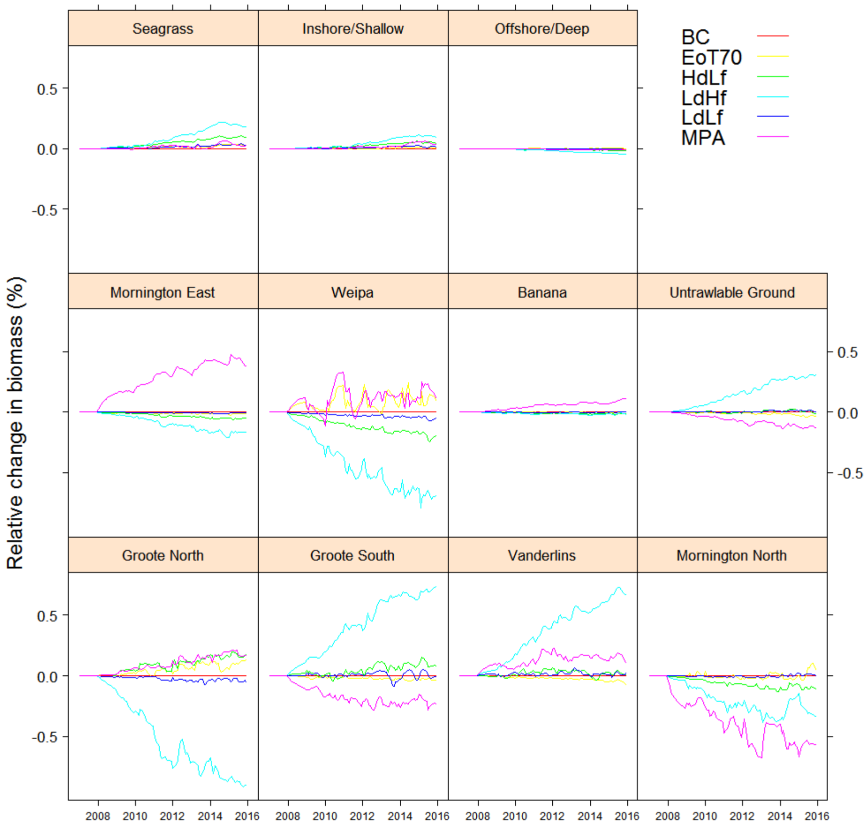
Holothurians



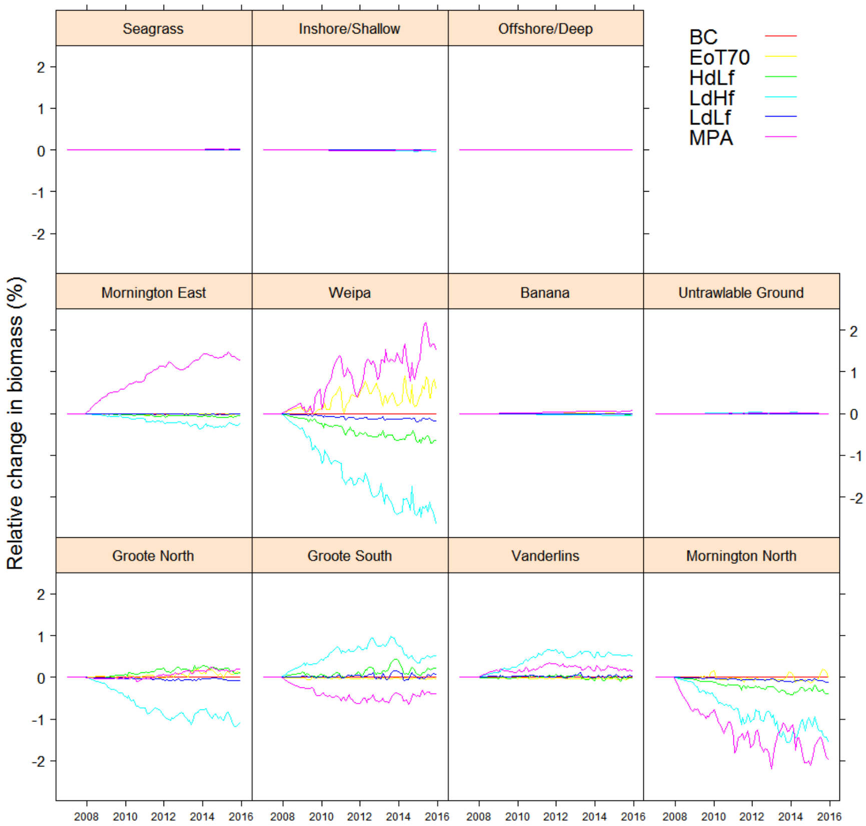
Infaunal detritivores/carnivores



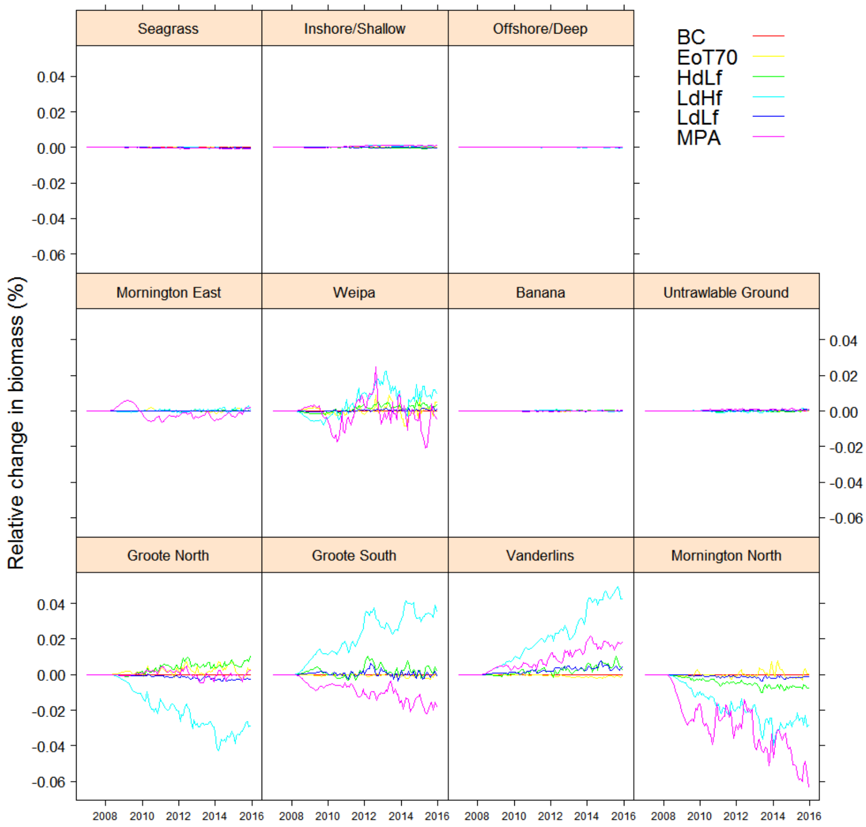
Large Gastropod Carnivore



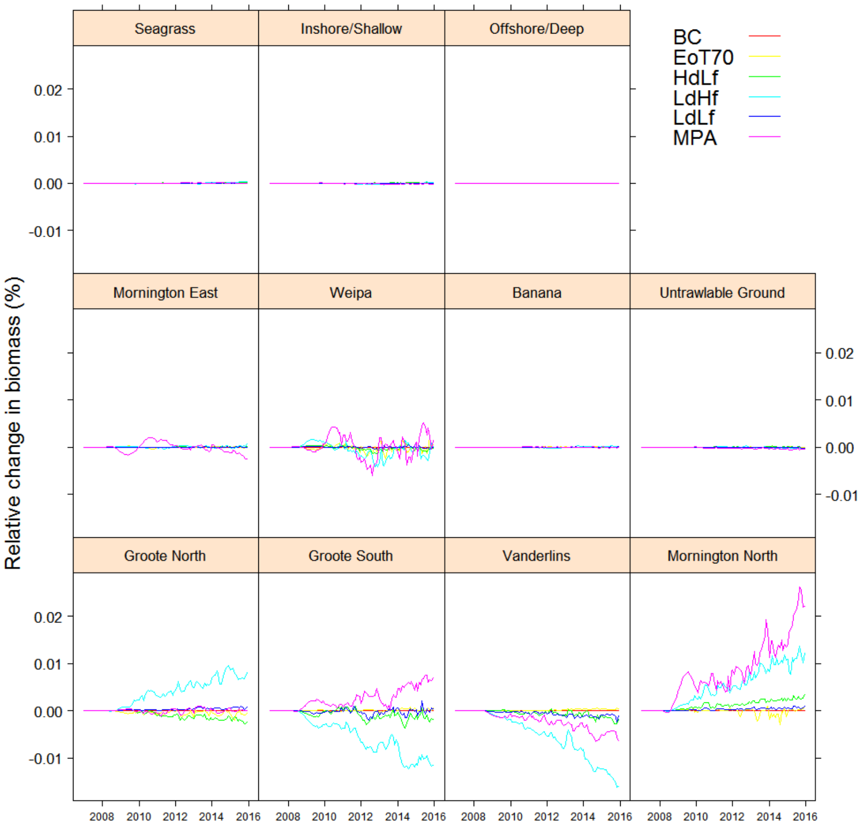
Large Sharks



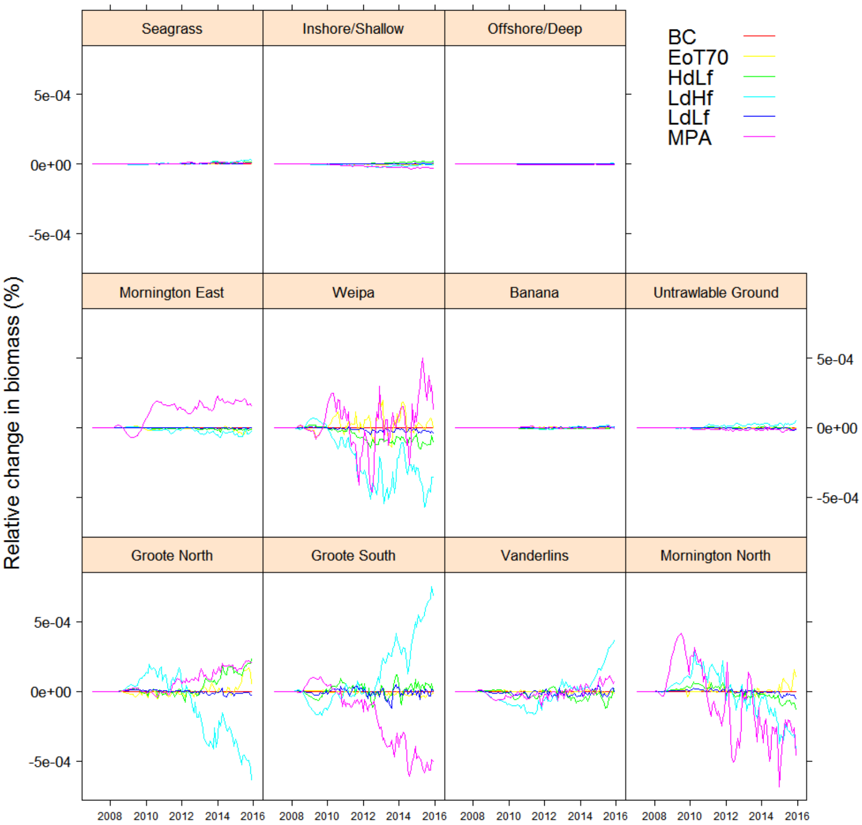
Lobsters



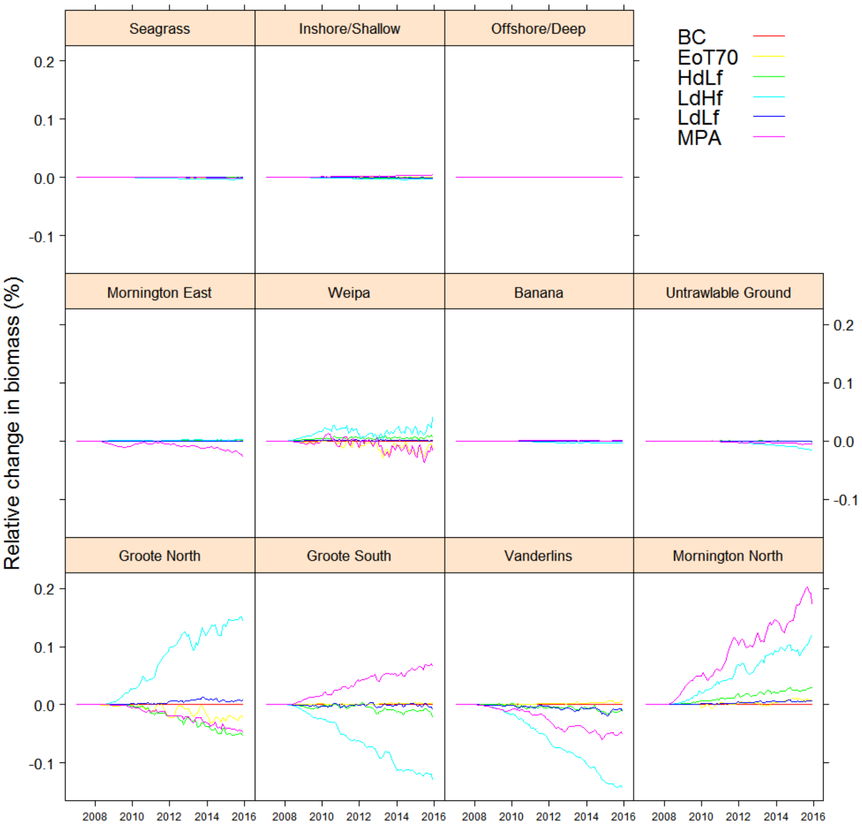
Macroalgae



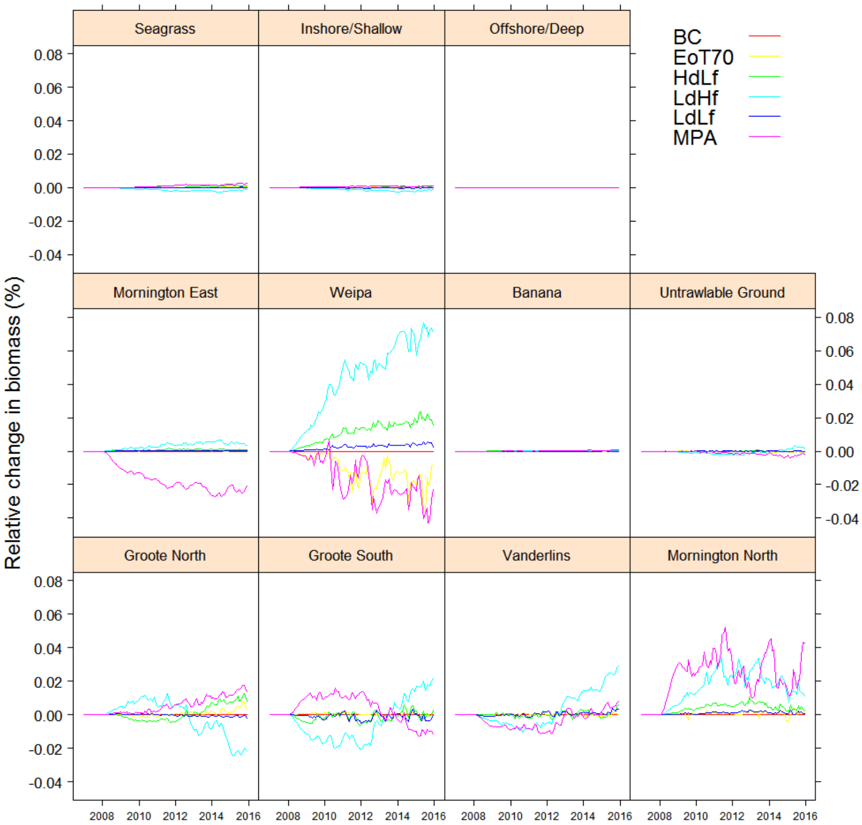
Microbial Heterotrophs



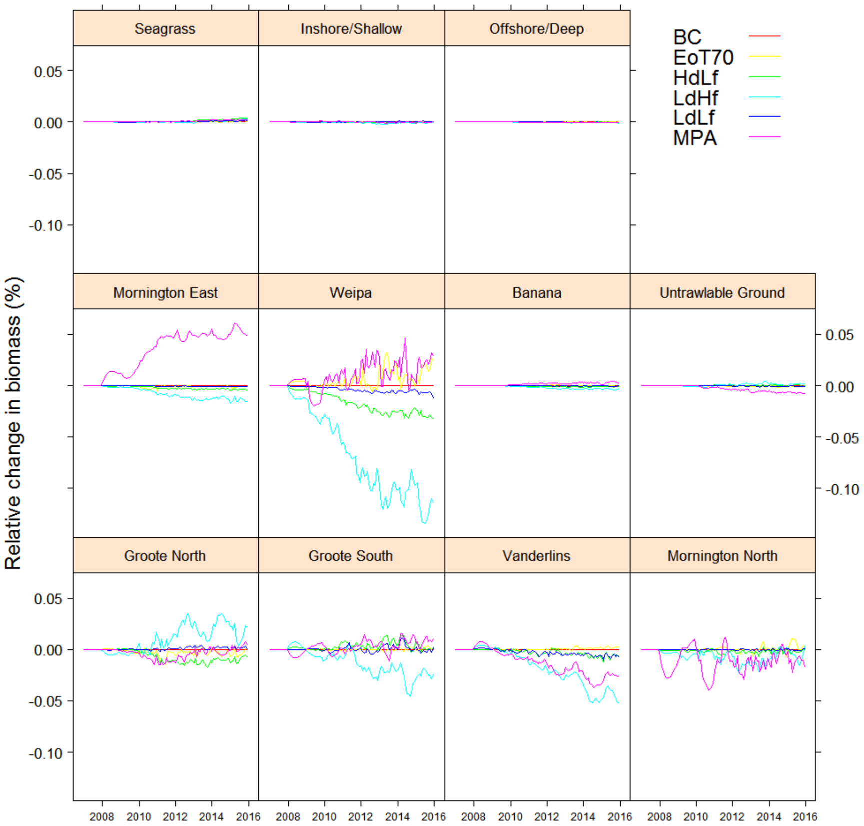
Microphytobenthos



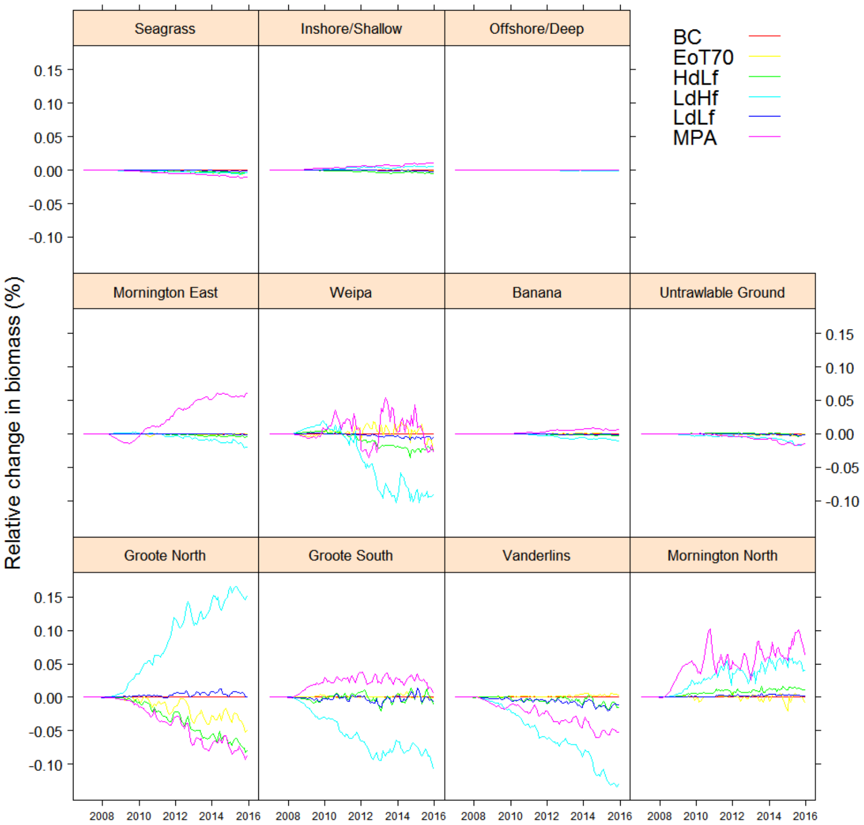
Mud Crab



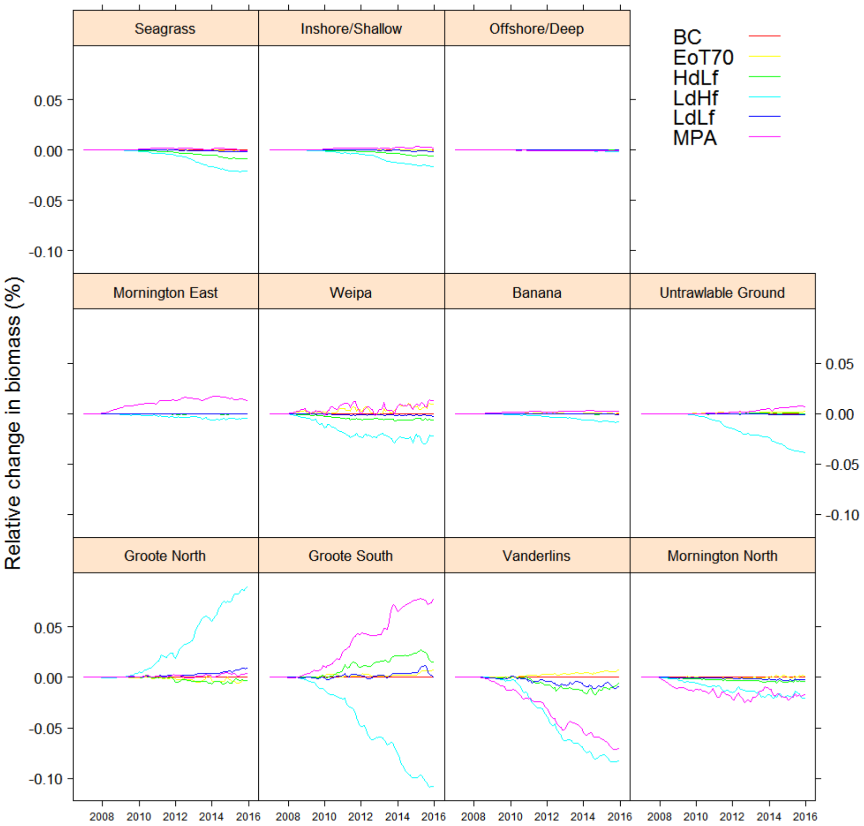
Ophiuroids



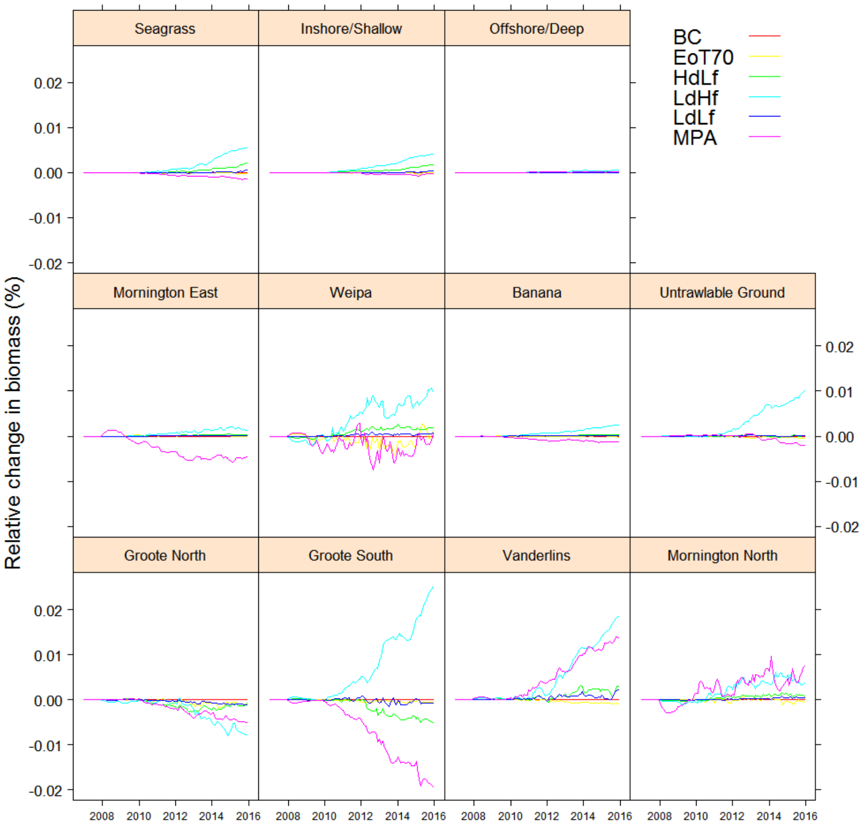
Other Commercial Prawns



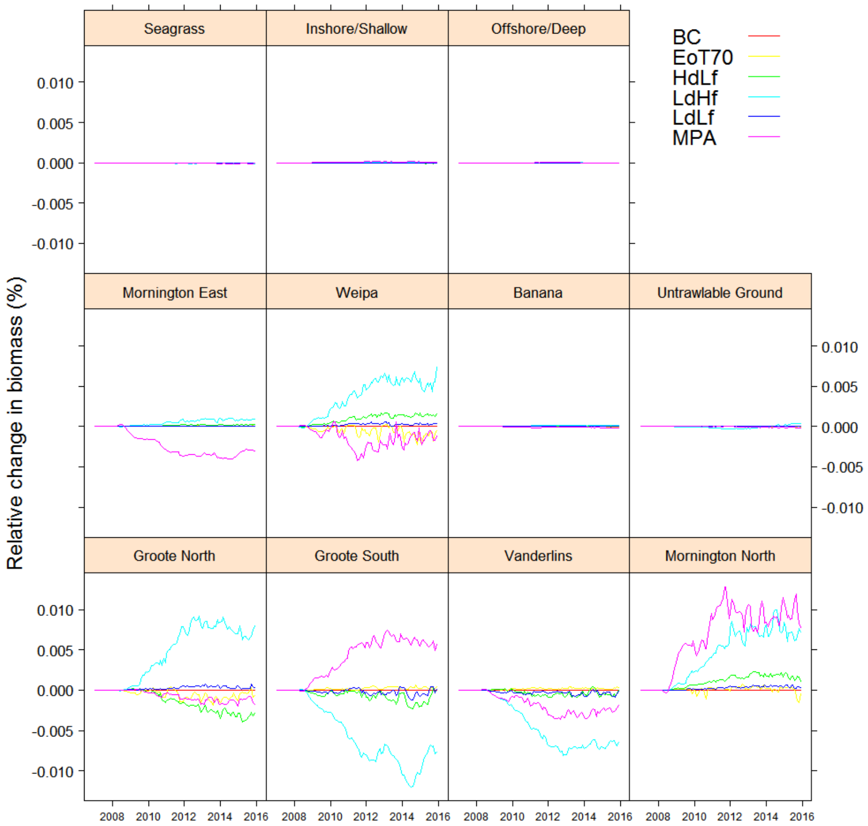
Other Non-commercial Prawns



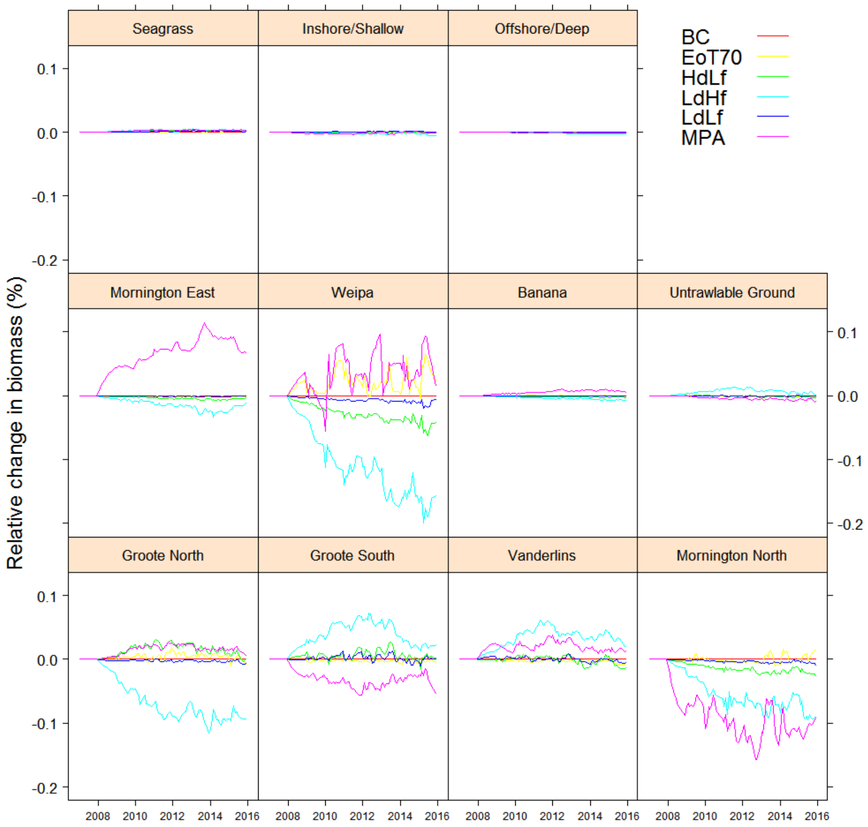
Pelagic Carnivores Fish



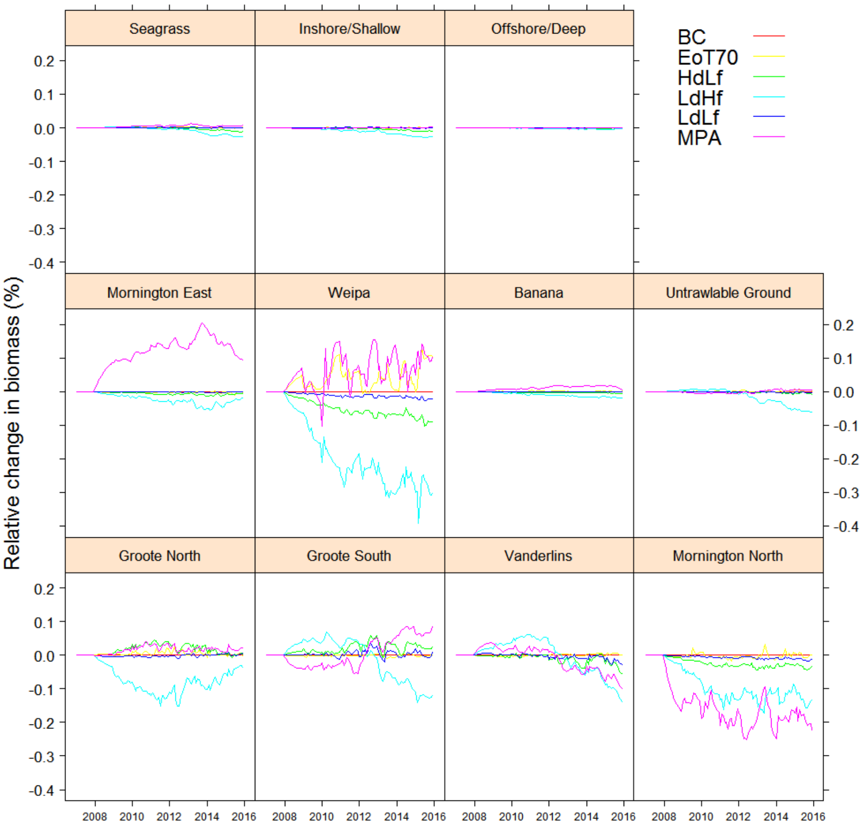
Pelagic Invert Feeder Fish



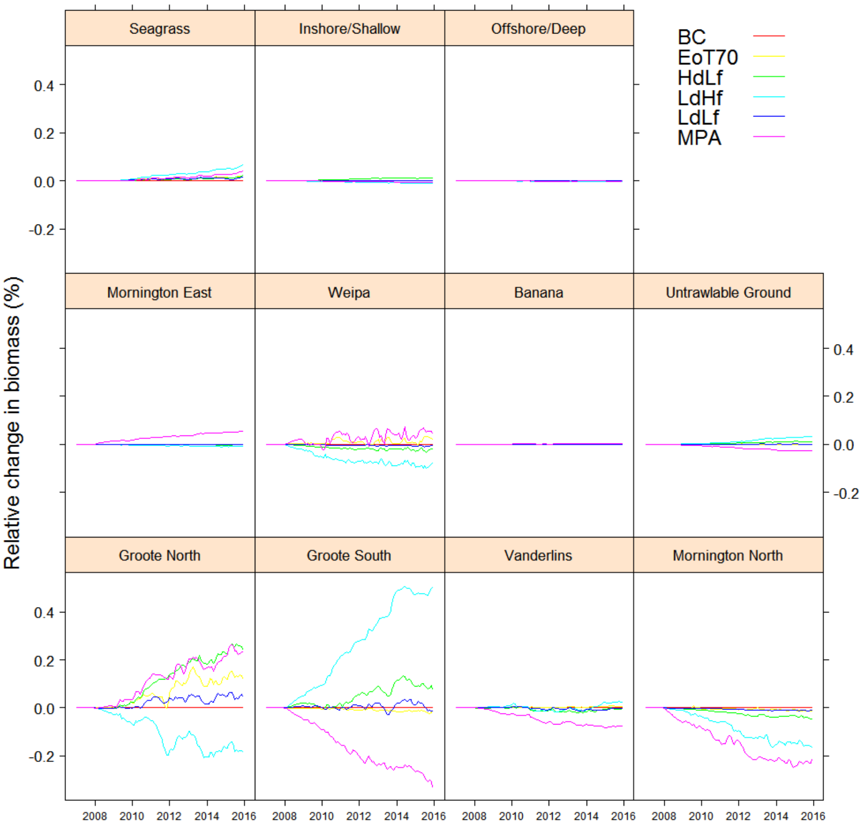
Phytonplankton



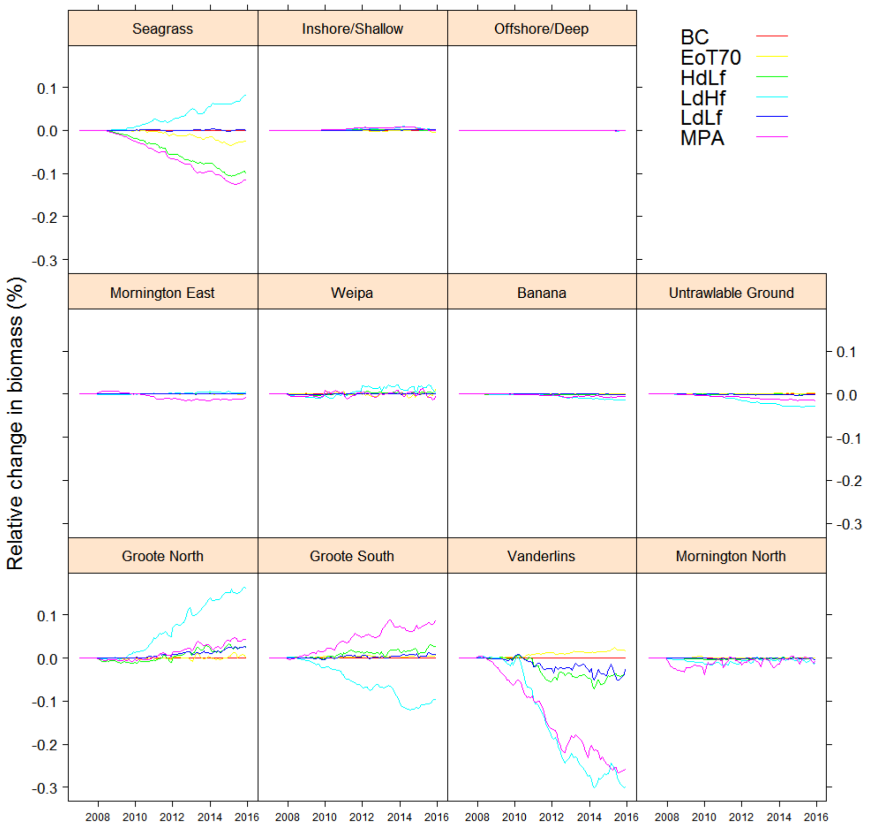
Rays



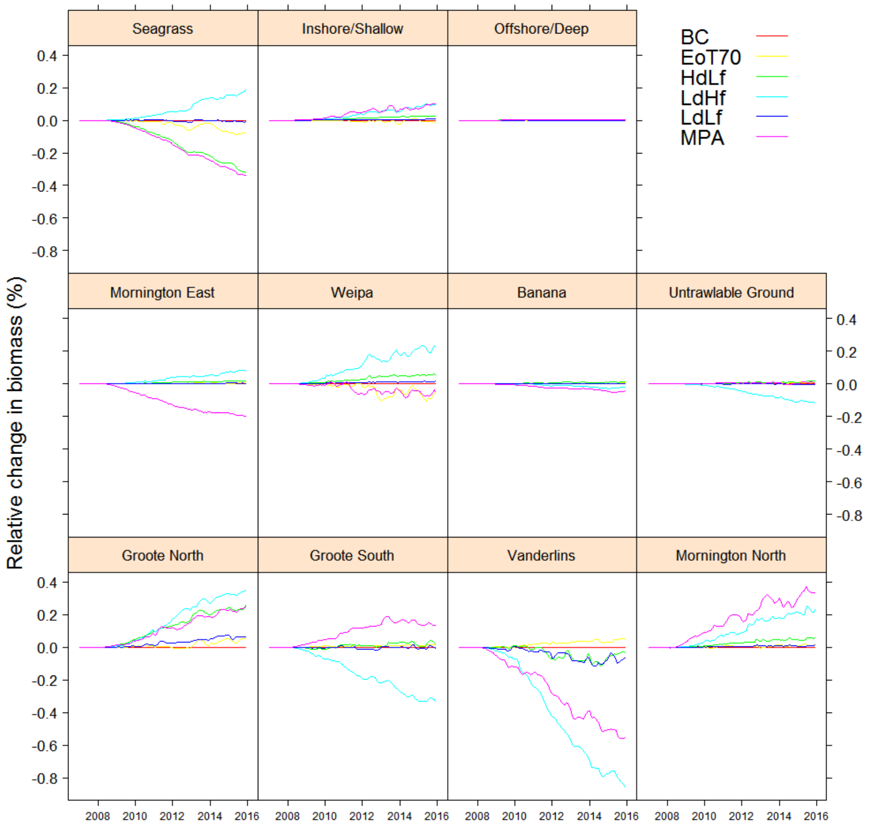
Red Snapper Fish



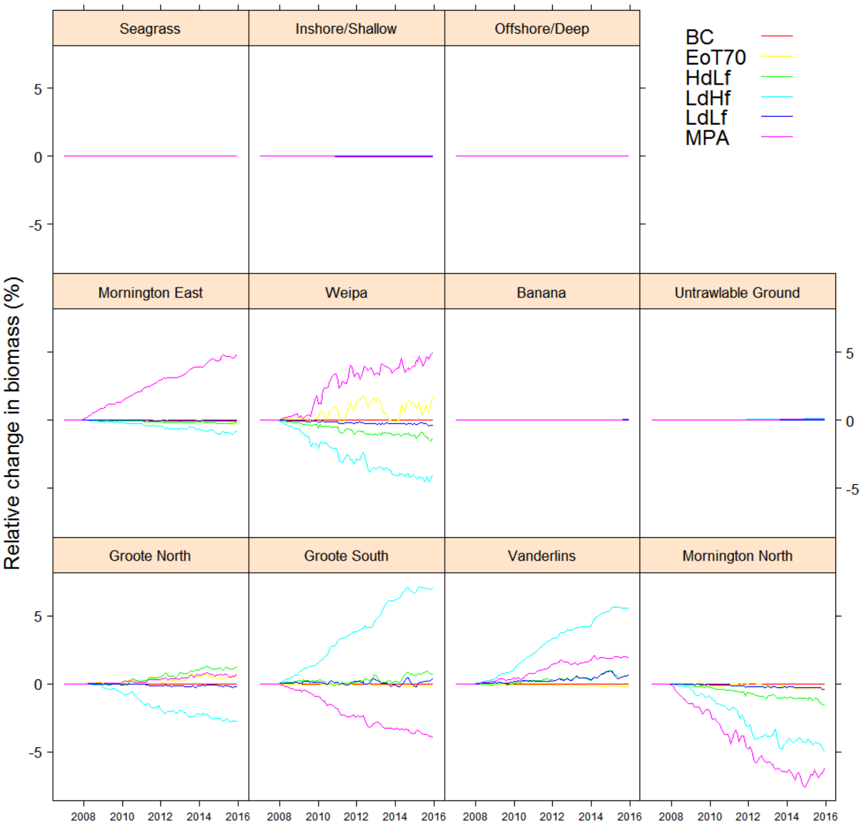
Reef-associated Carnivores Fish



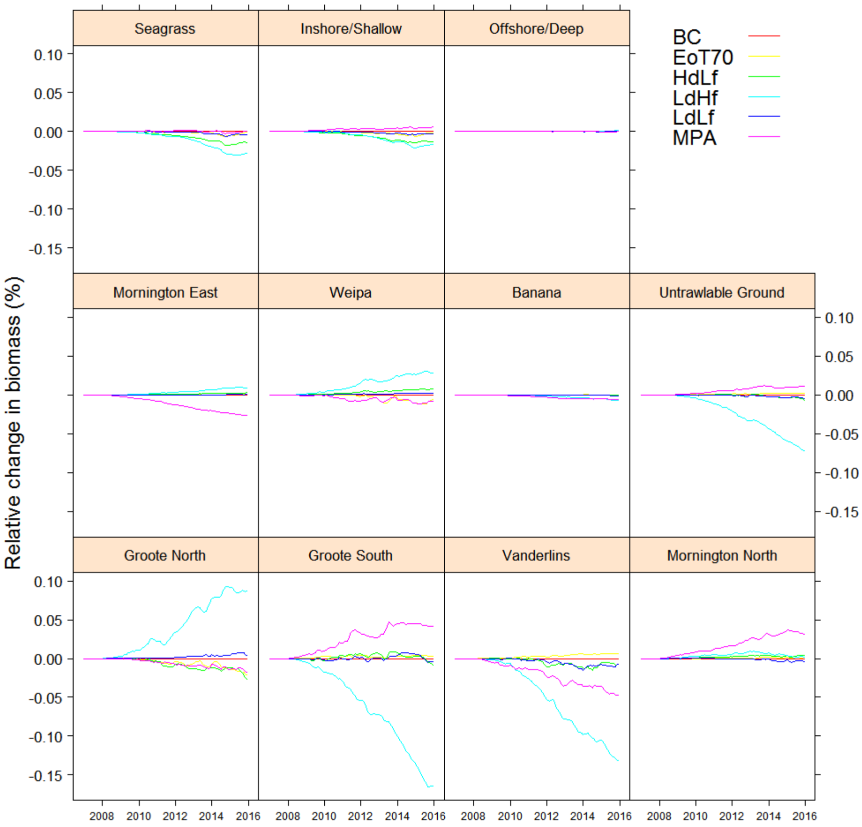
Reef-associated Hervivores Fish



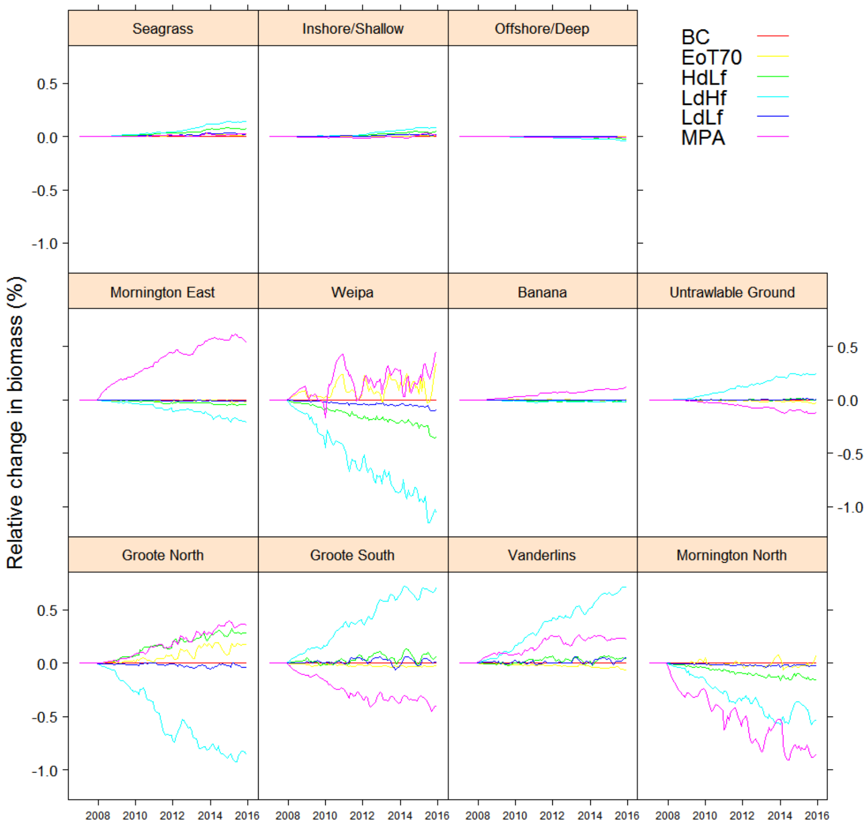
Reef-associated Invert Feeder Fish



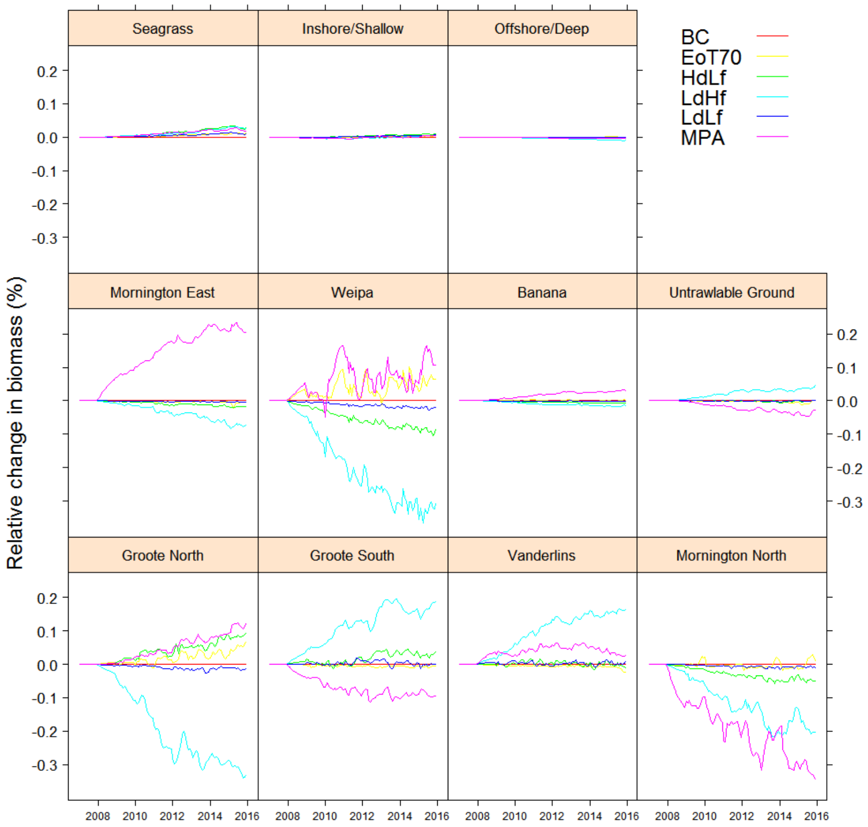
Sand Crab and other Large Crabs



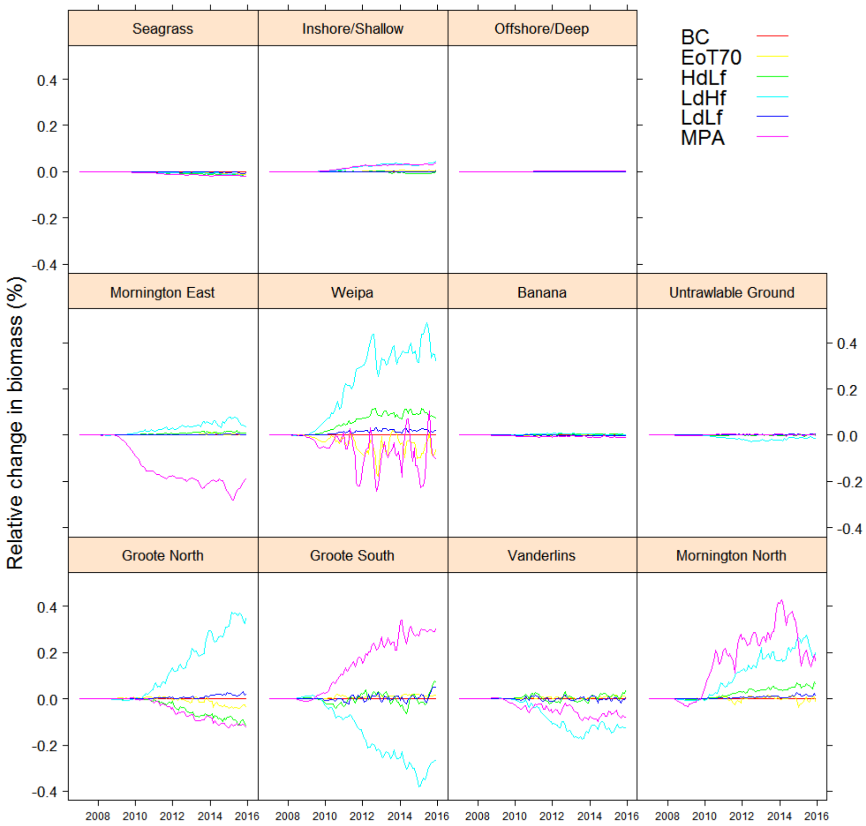
Sawfish



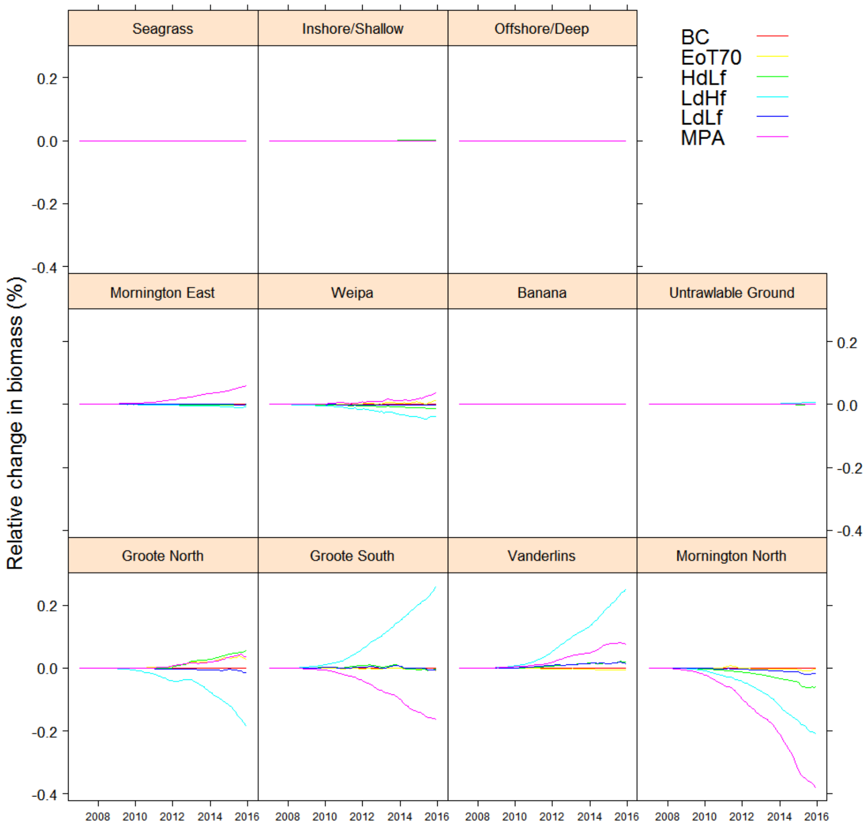
Seabirds



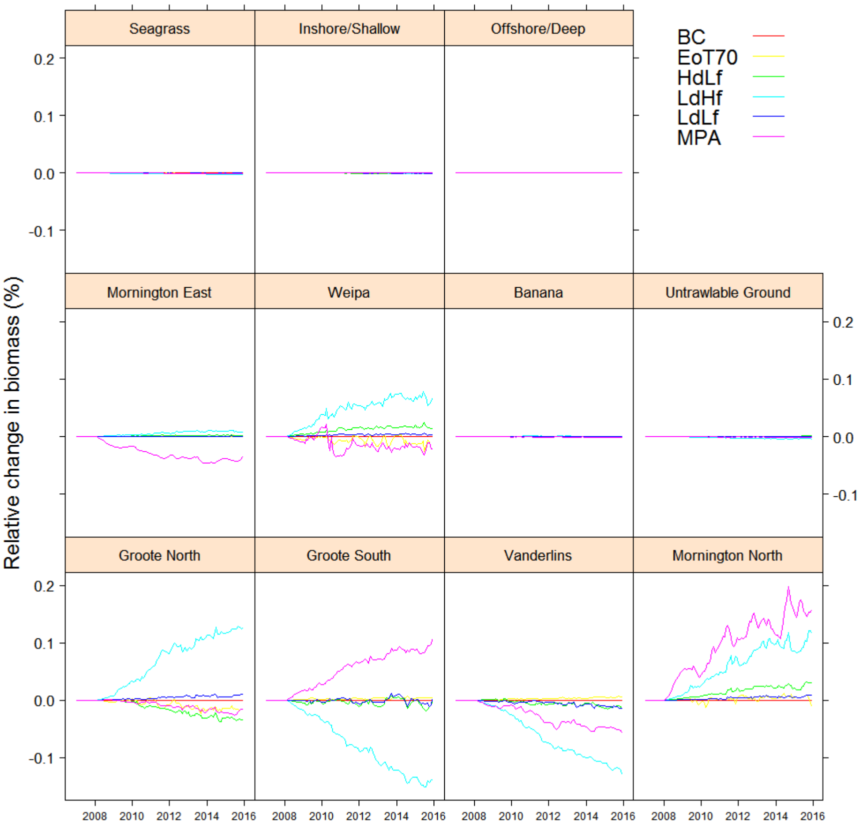
Sea Snakes



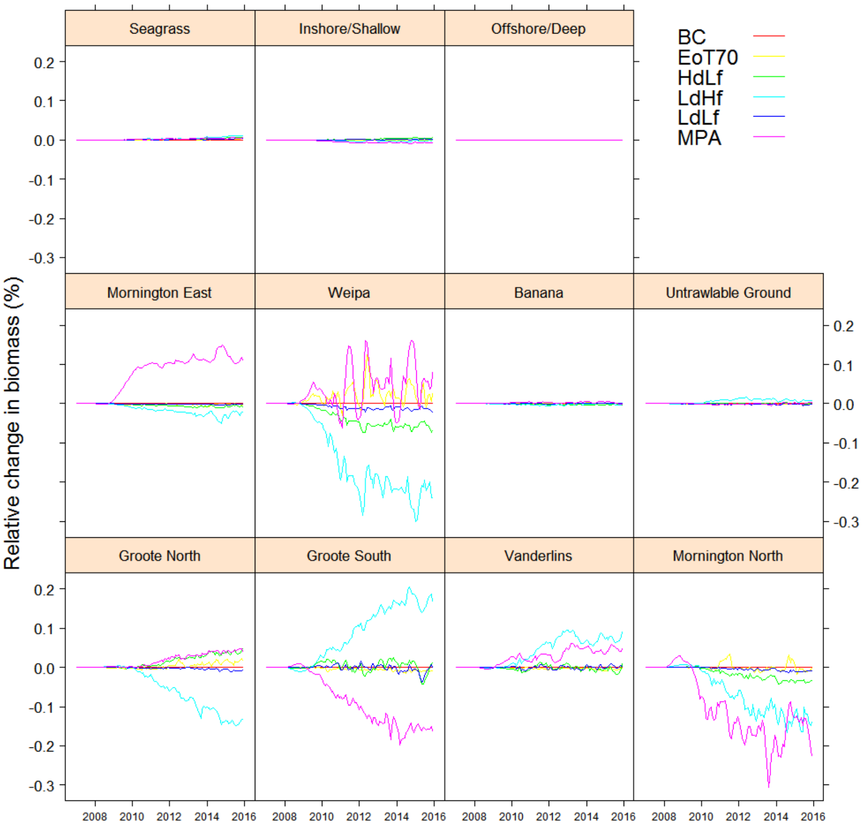
Seagrass



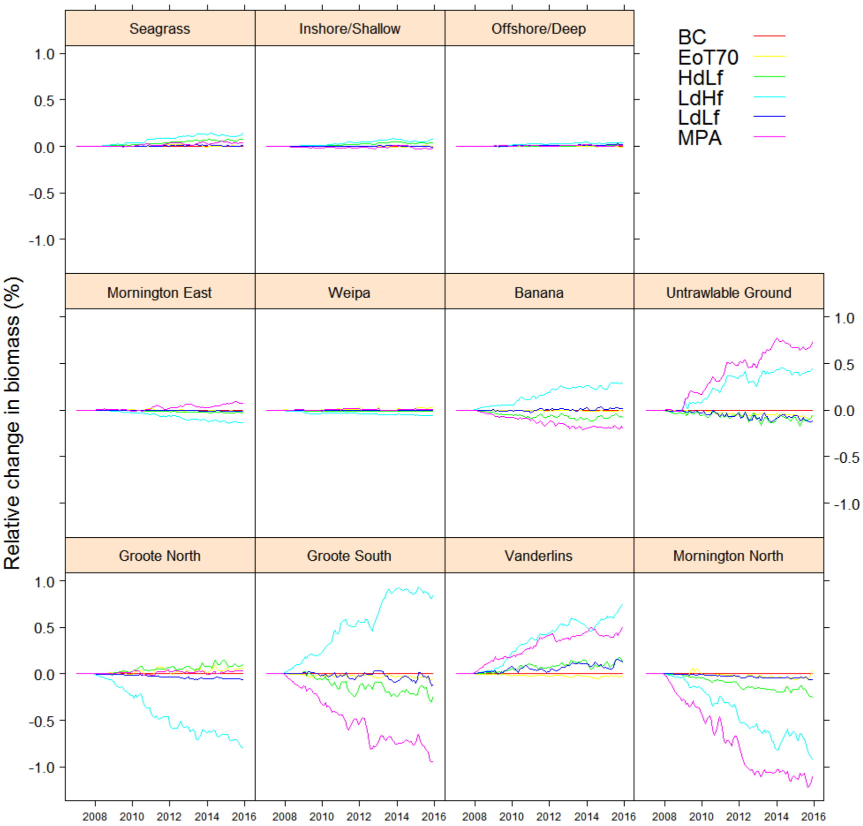
Sessile Epibenthos



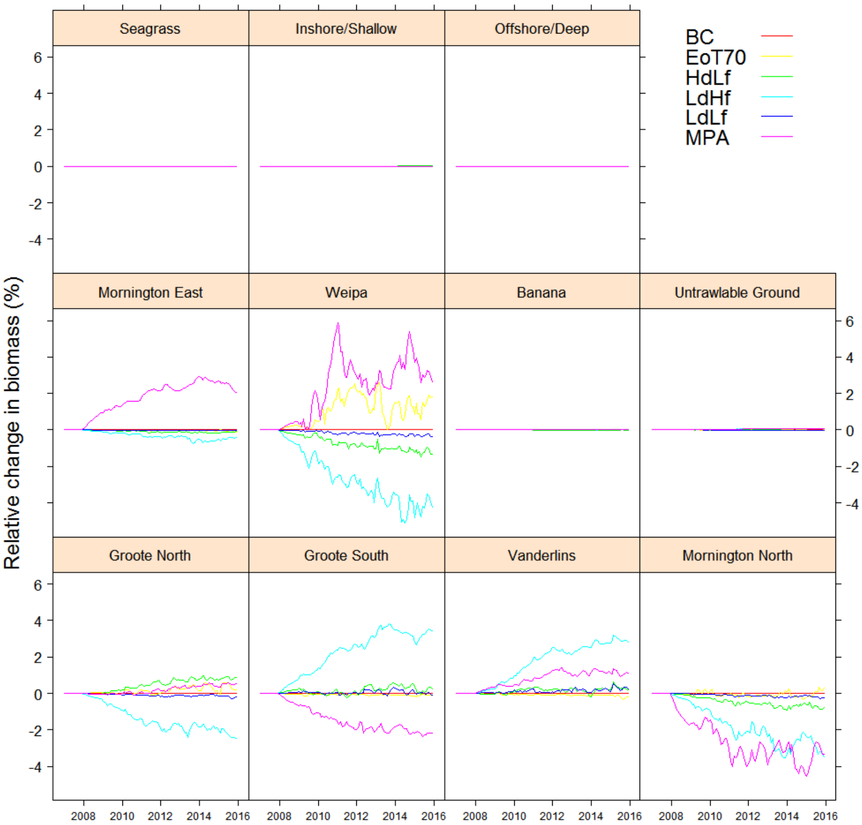
Small Crustaceans



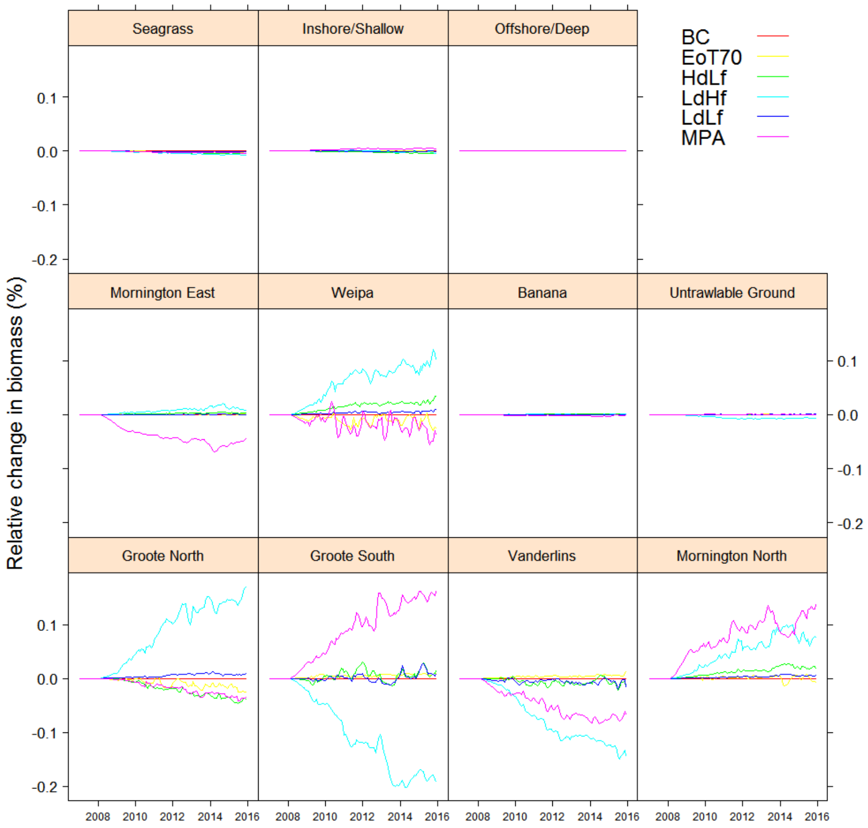
Small Gastropod Omnivore/Carnivore



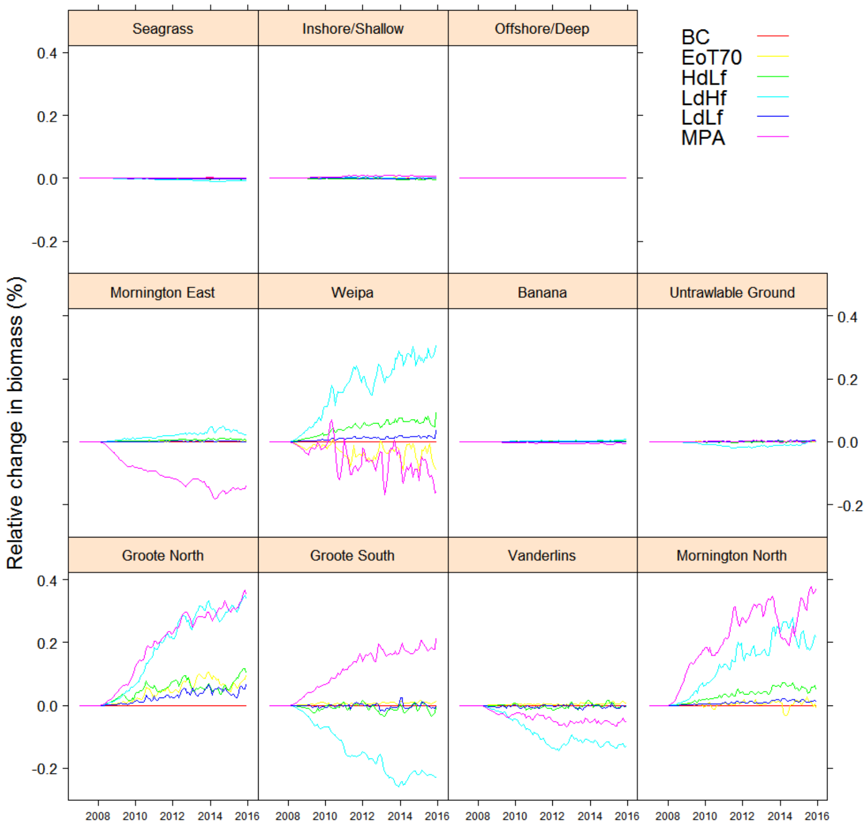
Small sharks



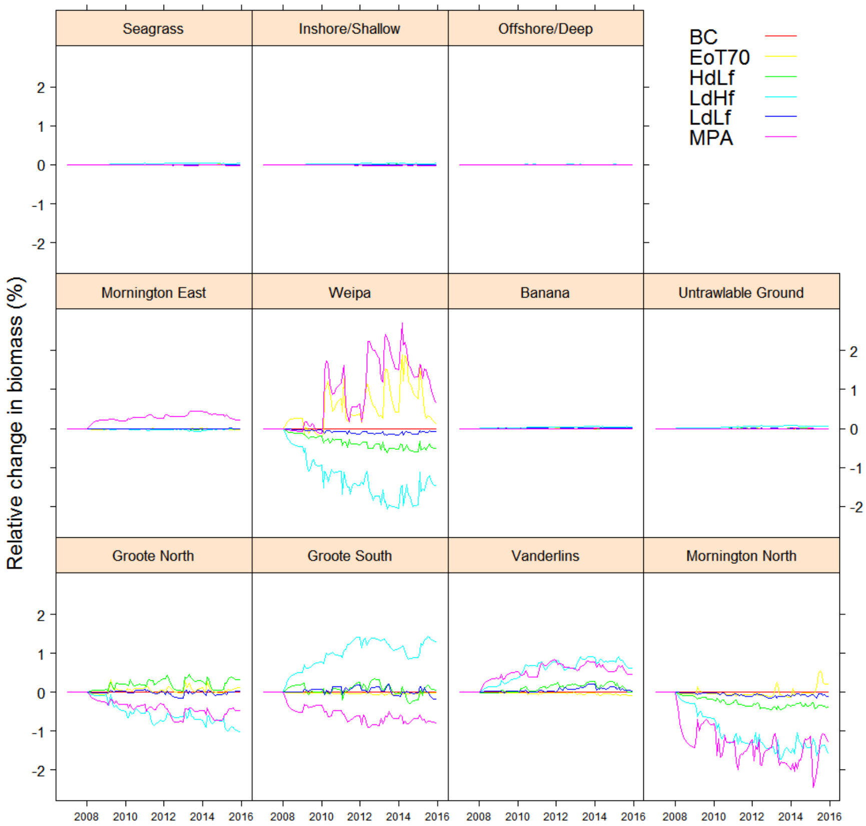
Spatangoid Urchins



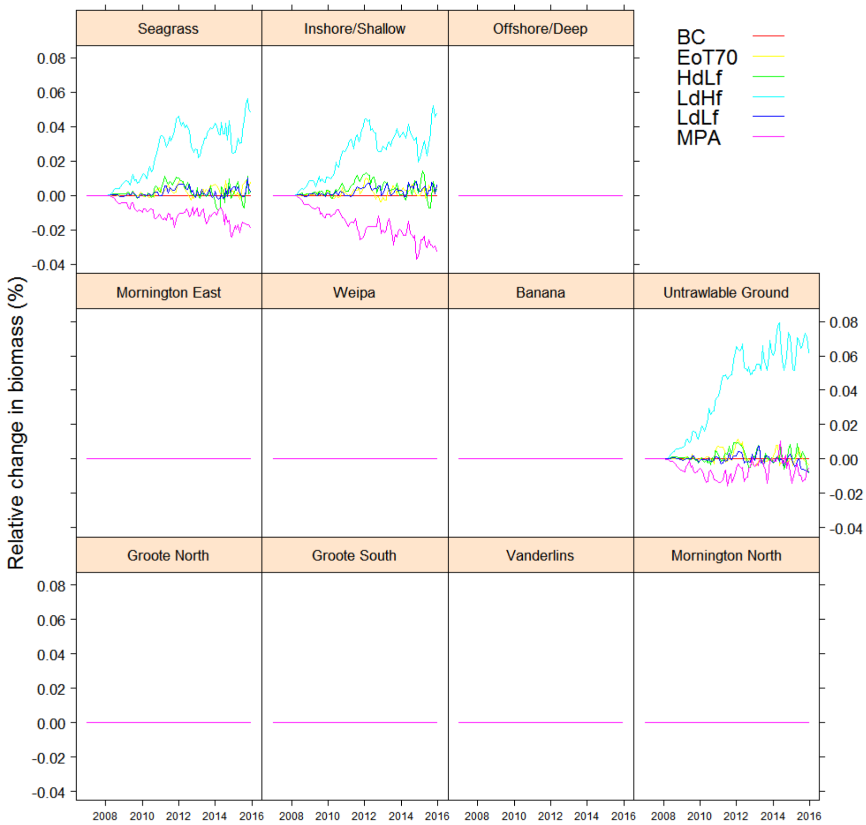
Stomatopods



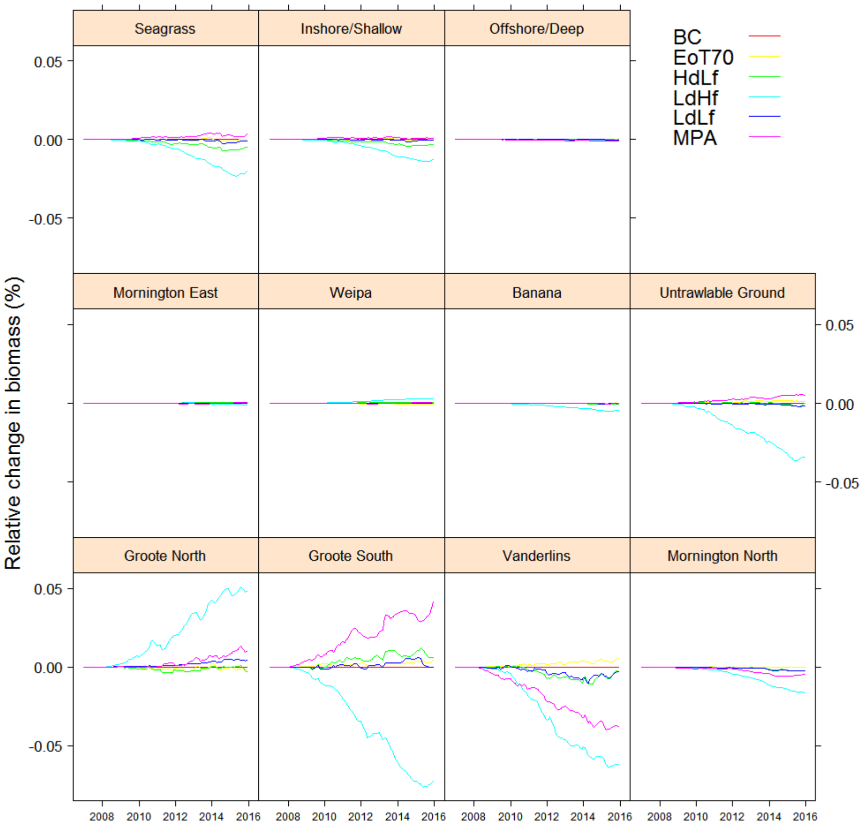
Thallasinid Prawns



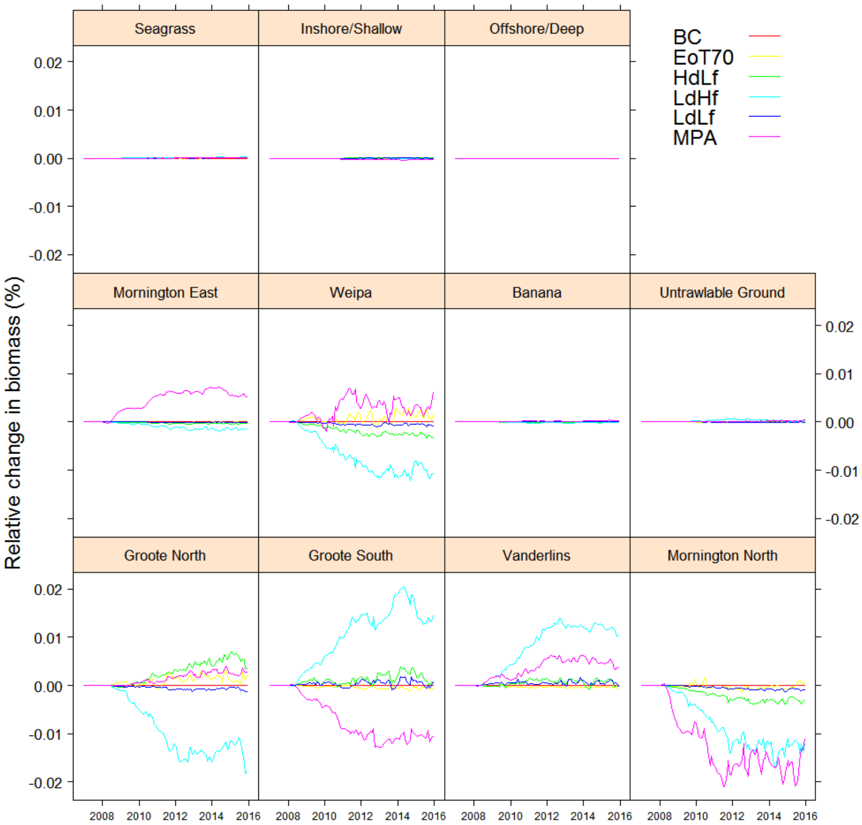
Tiger Prawn Adult



Tiger Prawn Juvenile

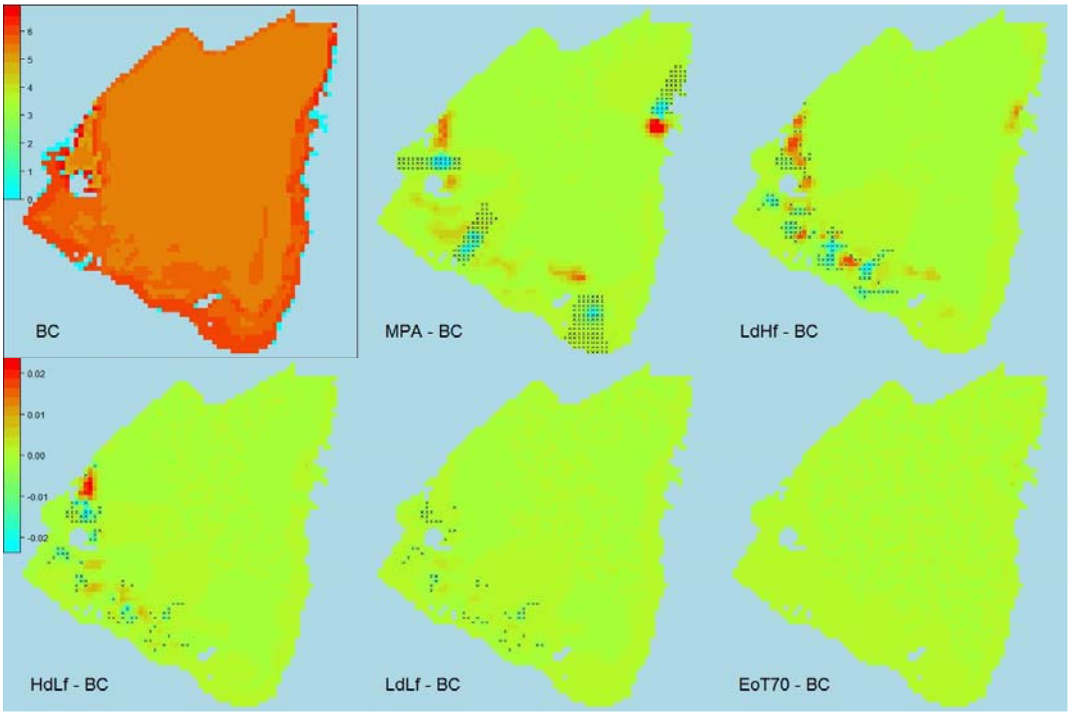


Turtles

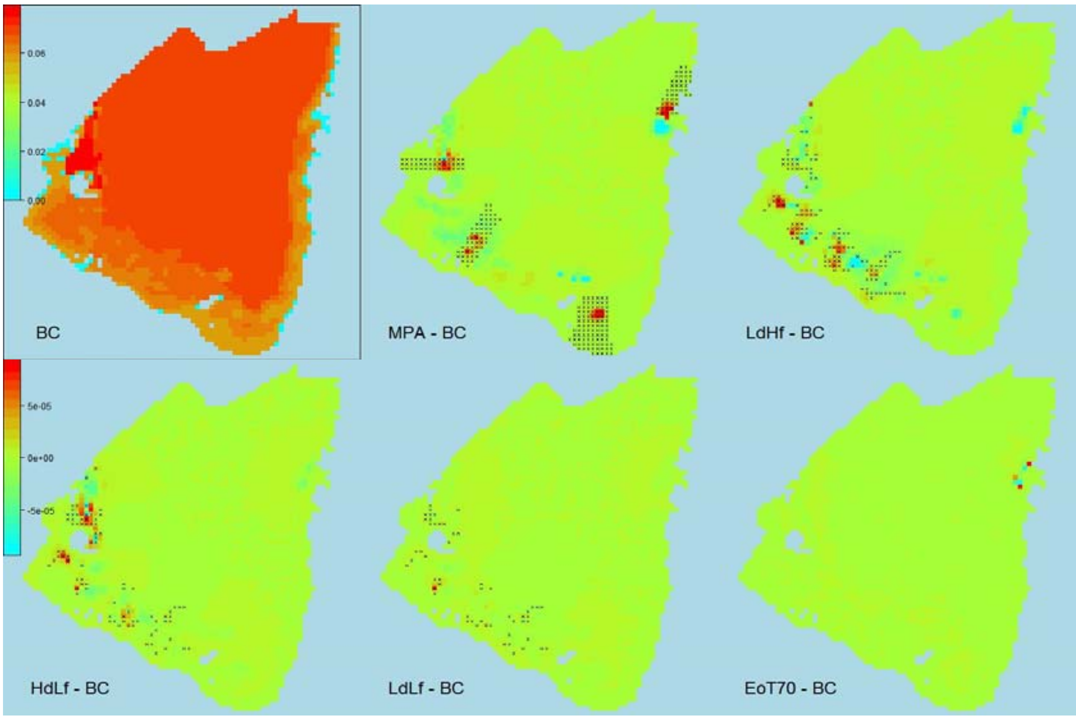


Zooplankton

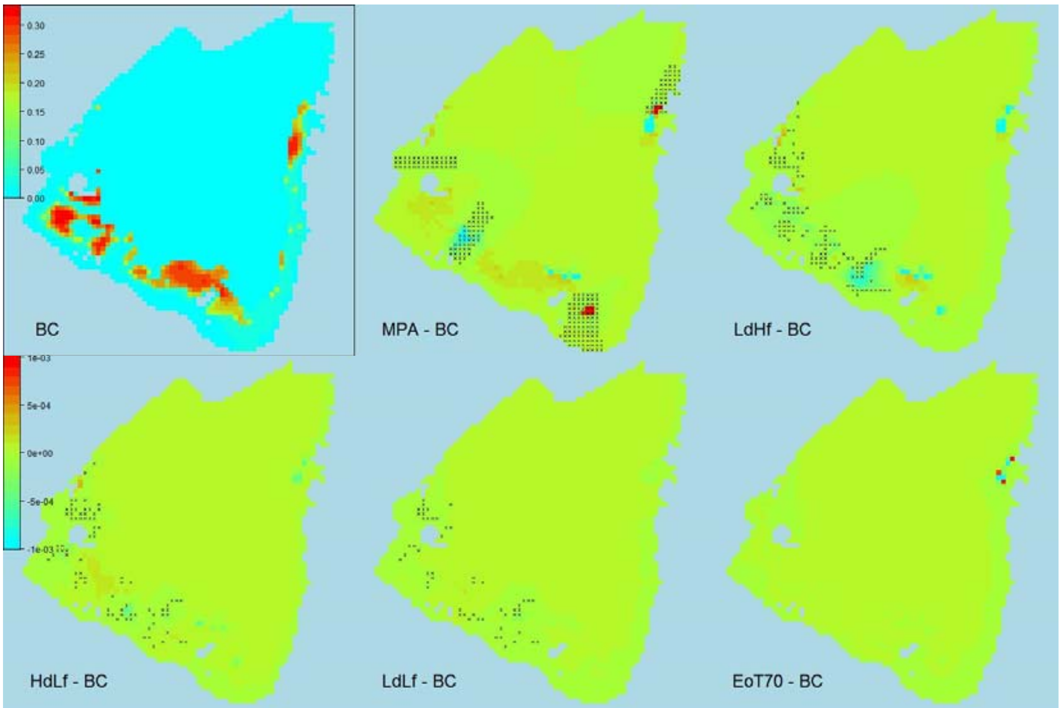
13.2 FINE-SCALE PREDICTIONS AT 2016 OF THE CHANGES OF THE RELATIVE BIOMASS PER FUNCTIONAL GROUPS IN RELATION TO THE BASE CASE (BC)



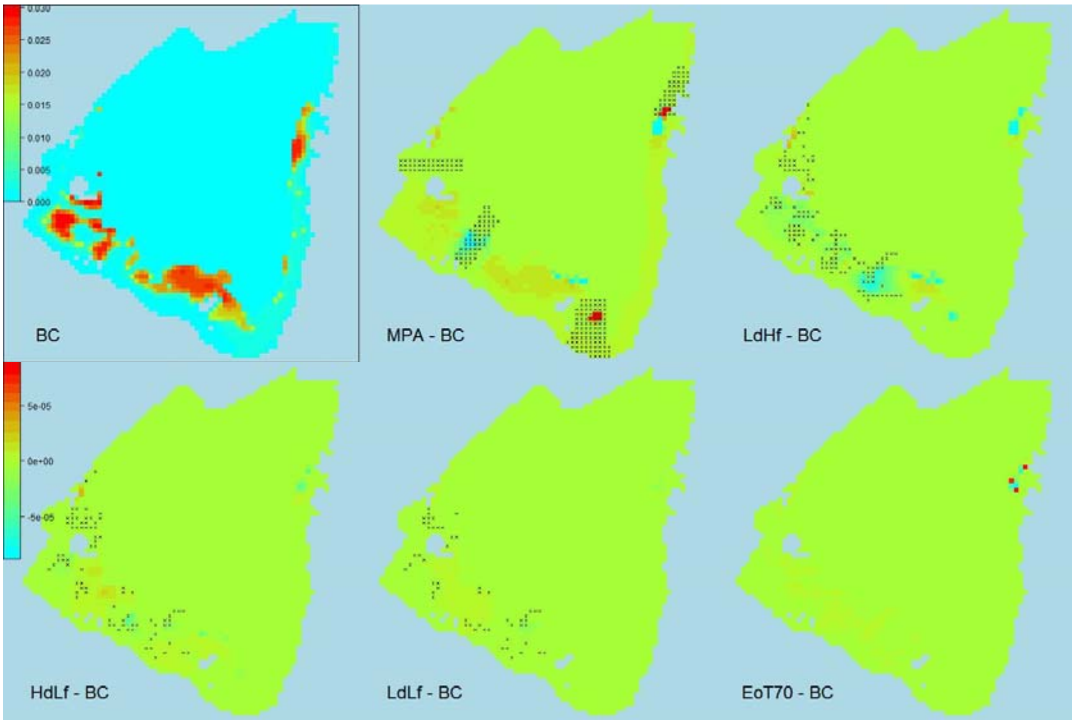
Annelids Detritrivore /Carninivores



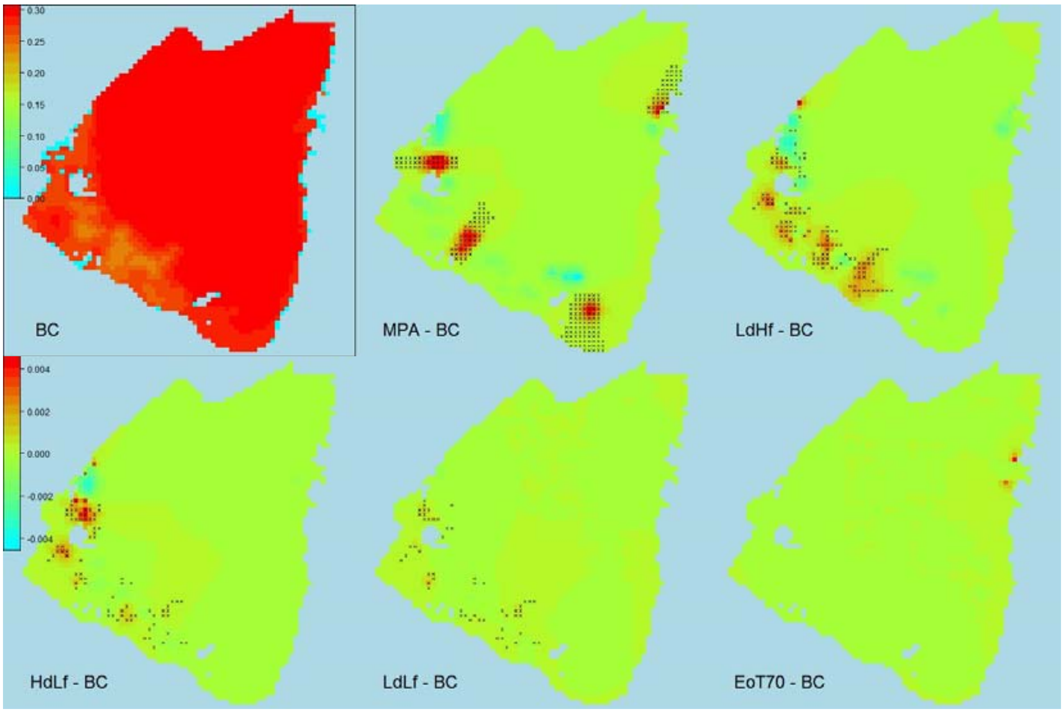
Asteriods



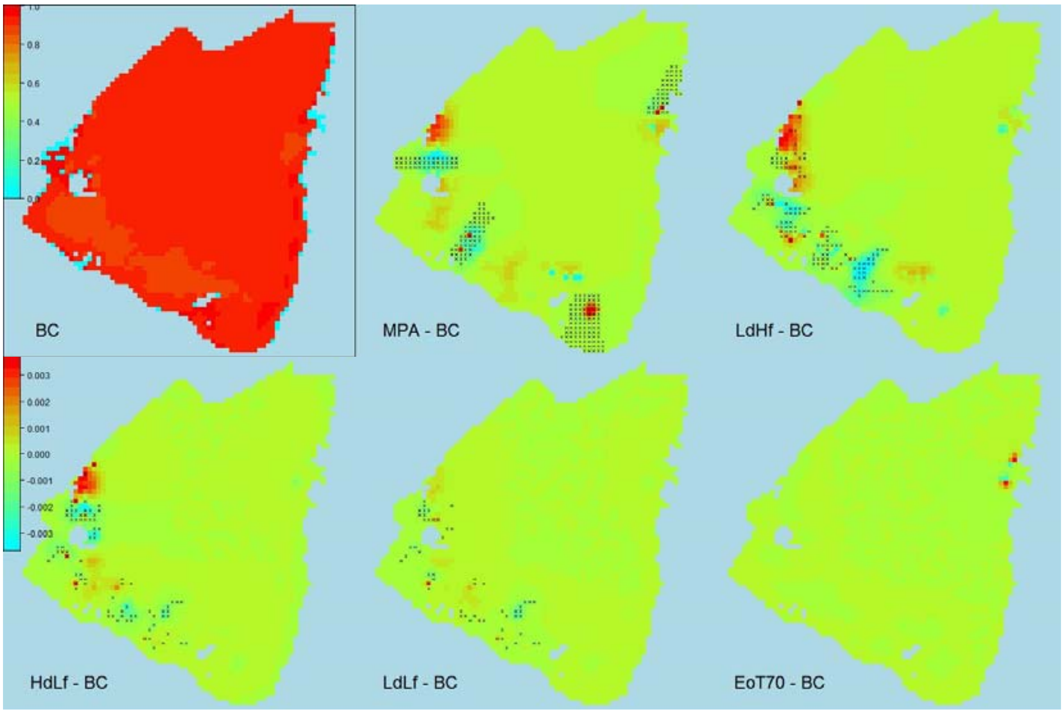
Banana Prawn Adult



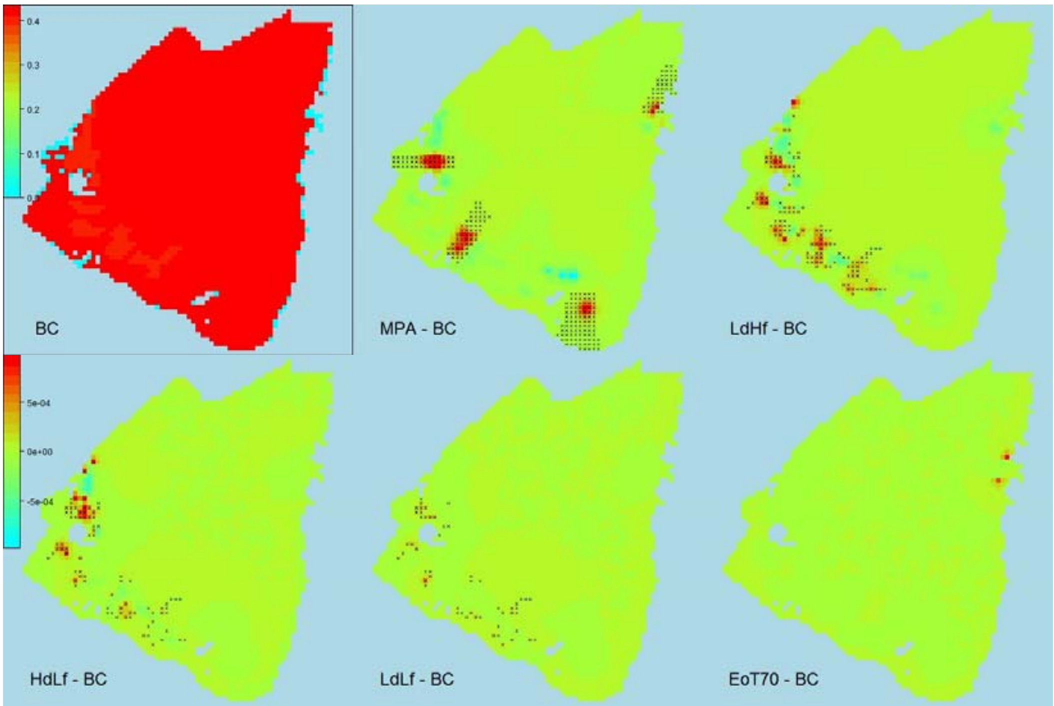
Banana Prawns Juvenile



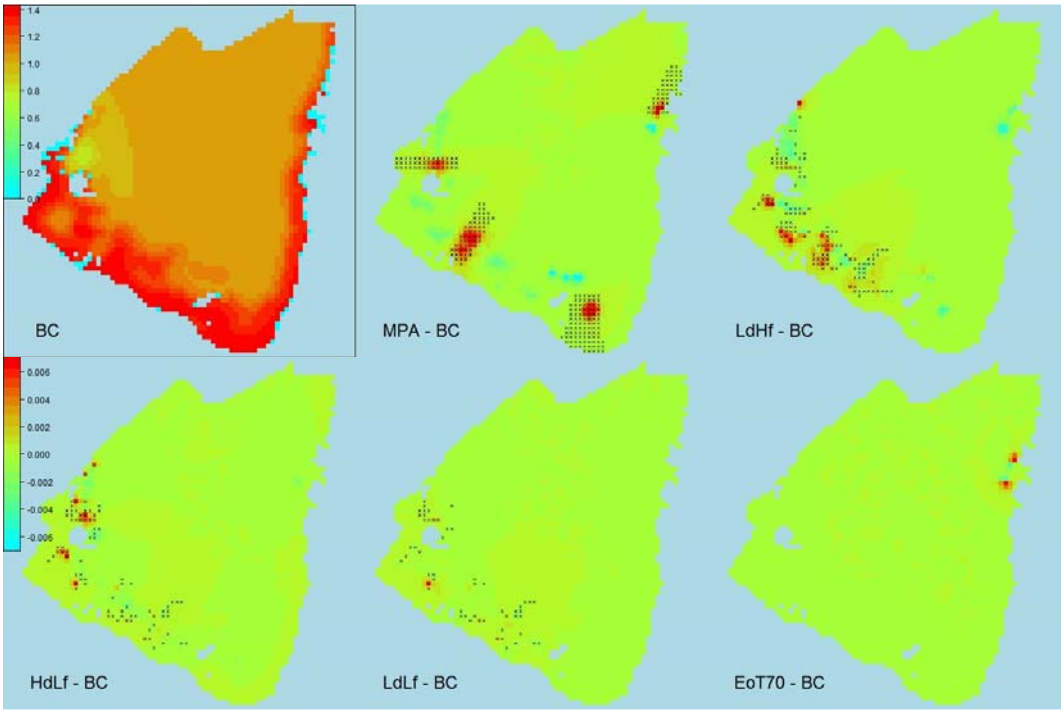
Benthic Carnivores Fish



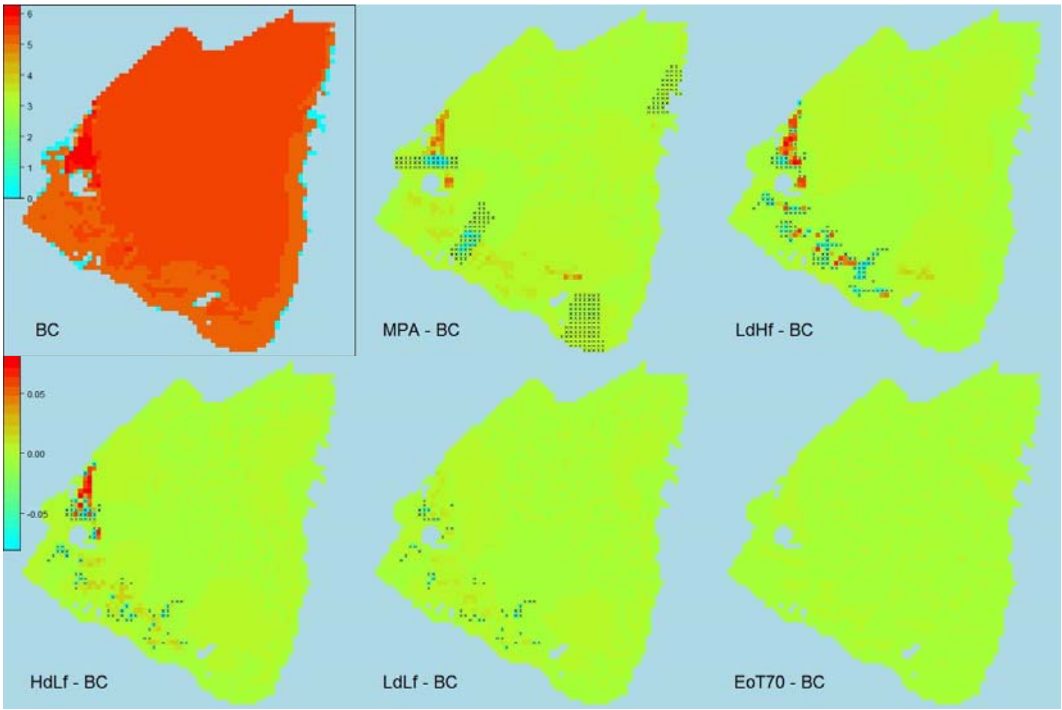
Benthic Invert Feeders Fish



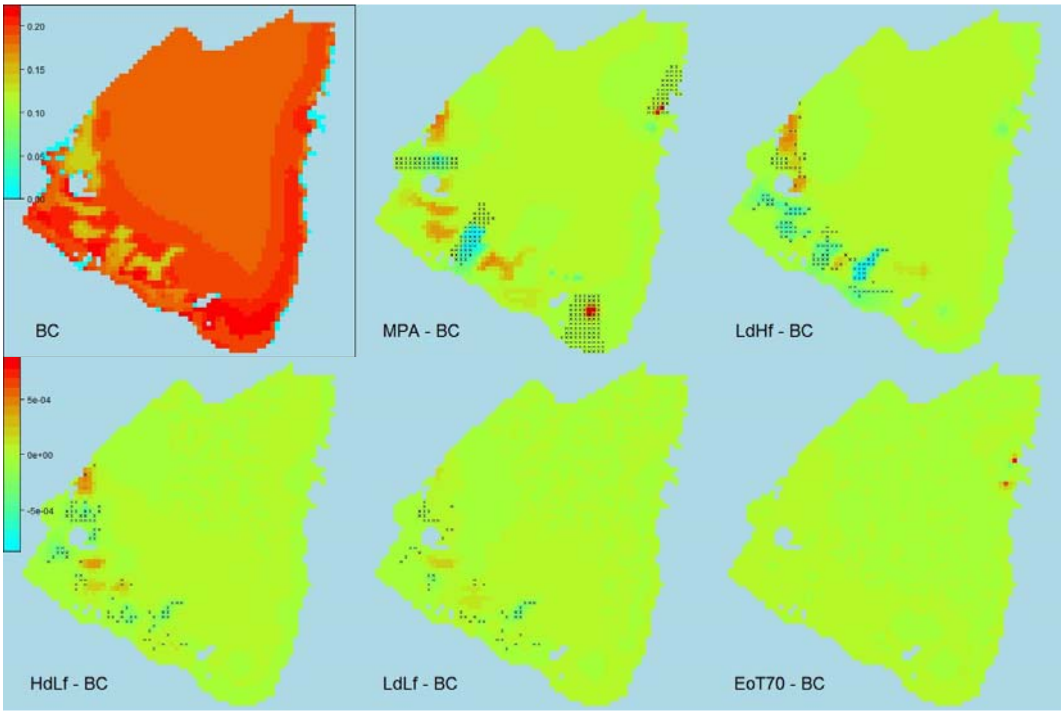
Benthopelagic Carnivores Fish



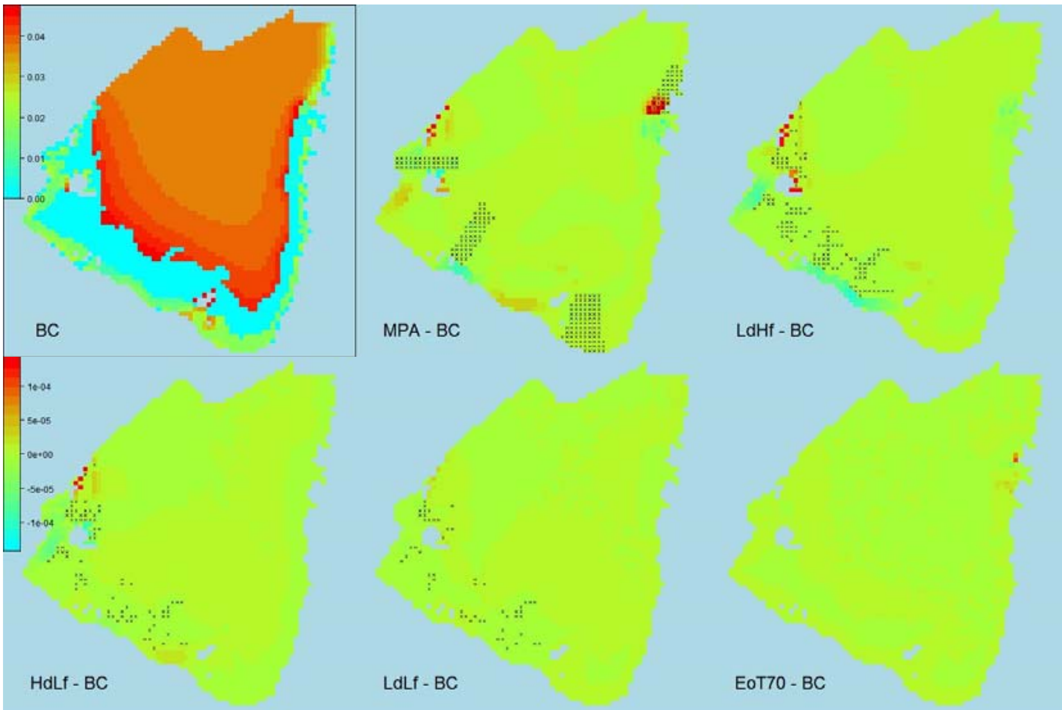
Benthopelagic Invertebrate Feeder Fish



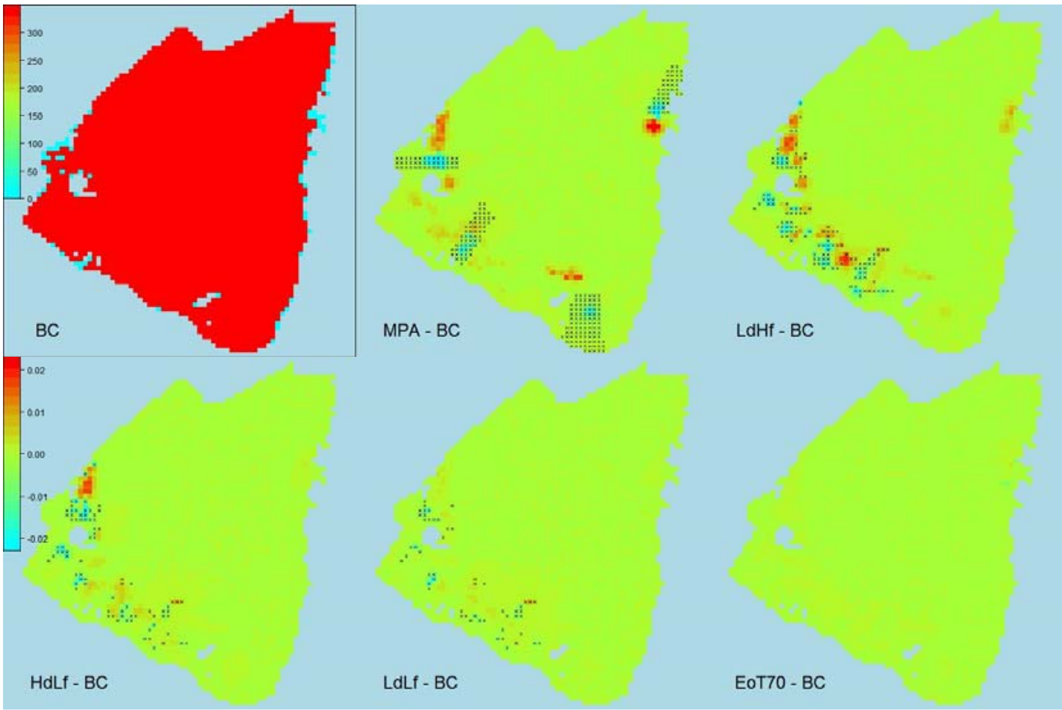
Bivalves



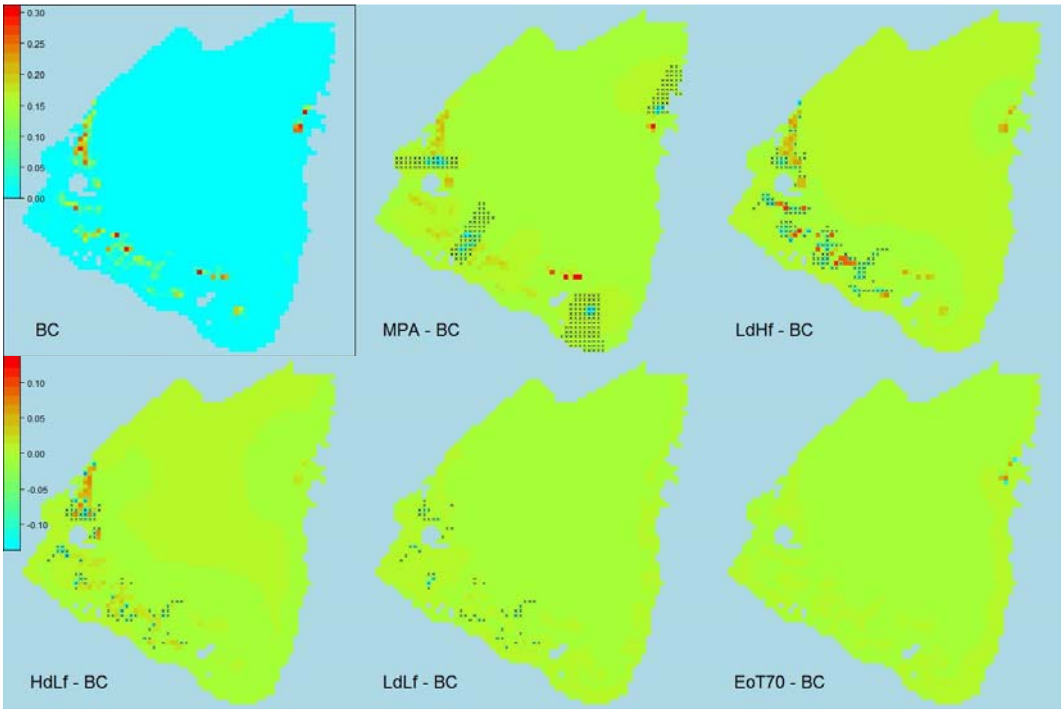
Cephalopods



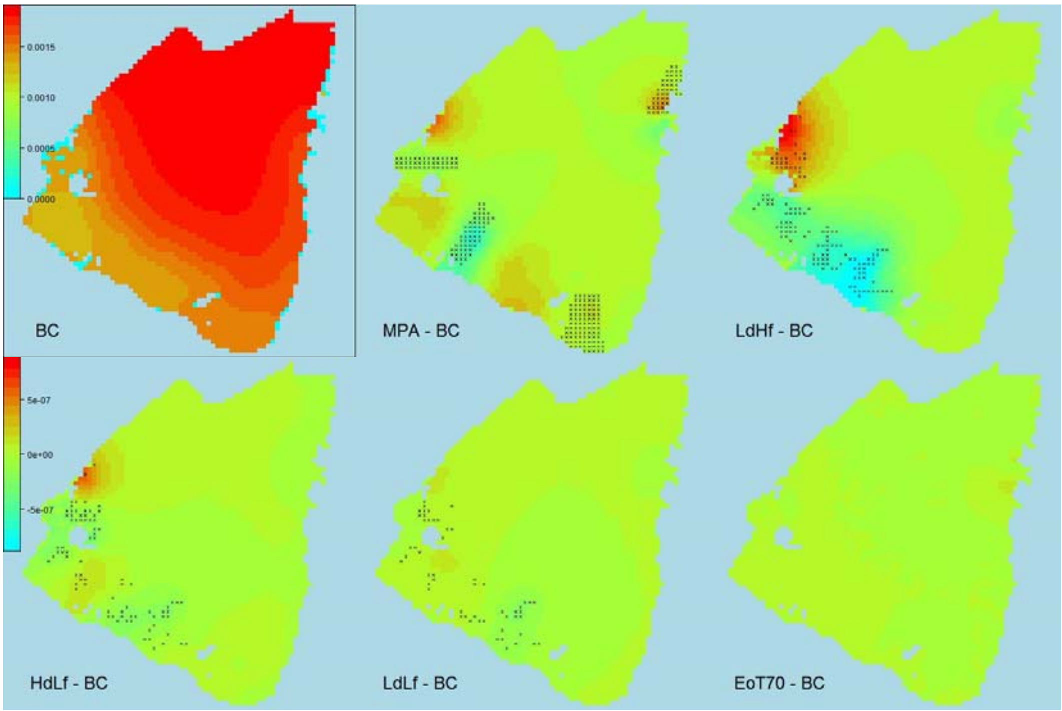
Detritivores Fish



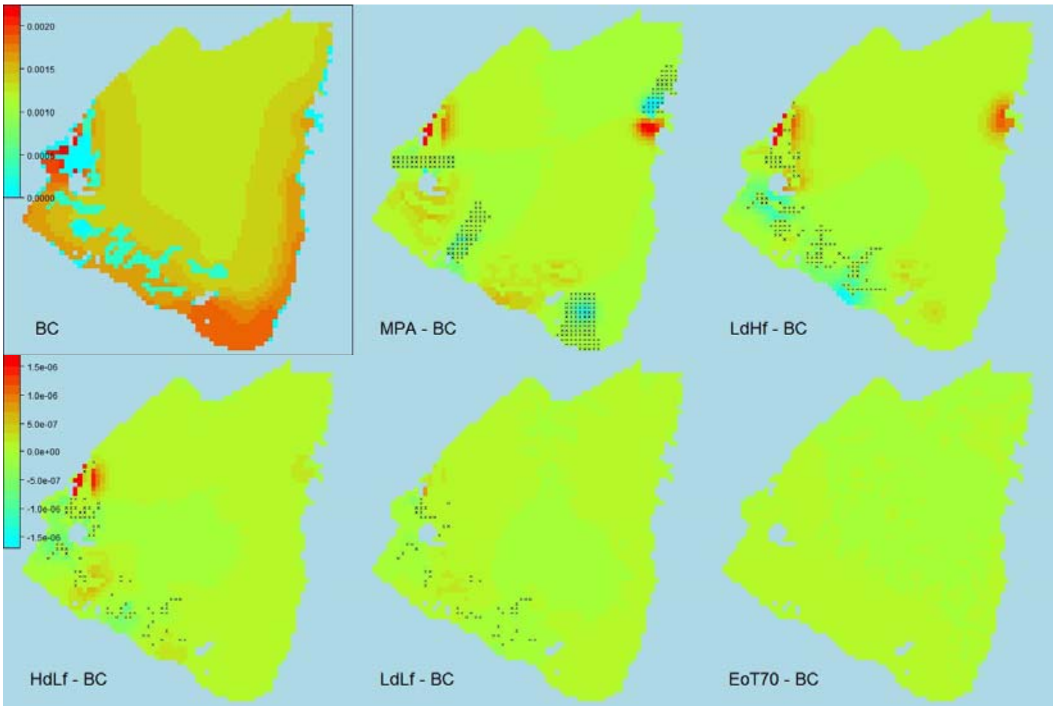
Detritus



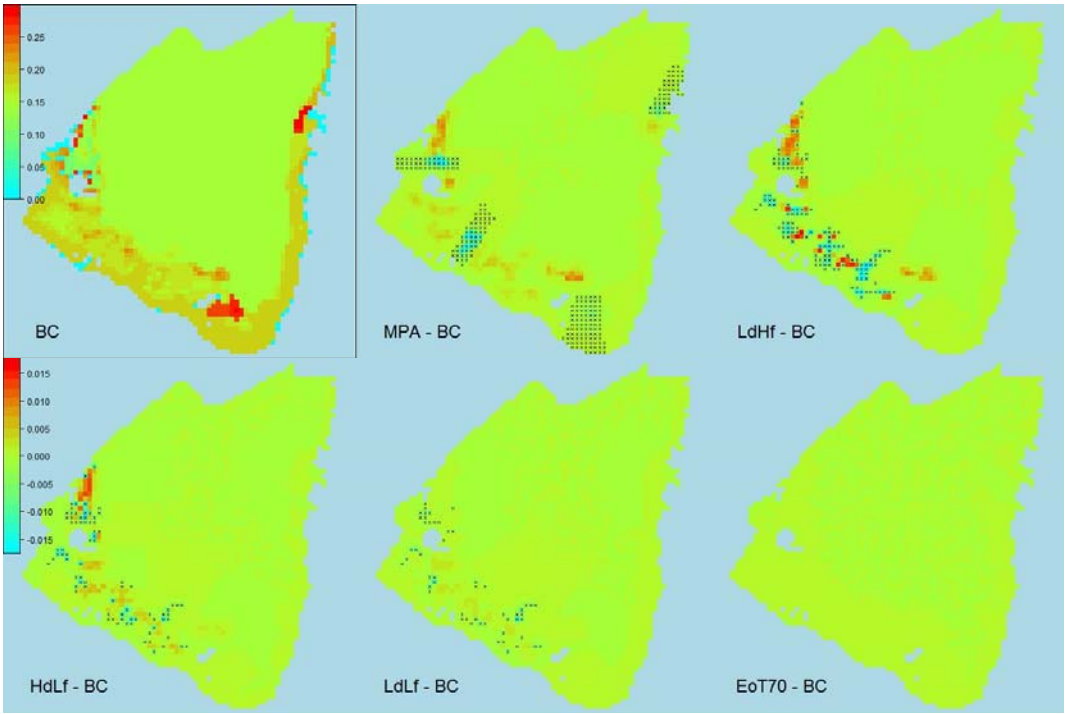
Discarded Bycatch



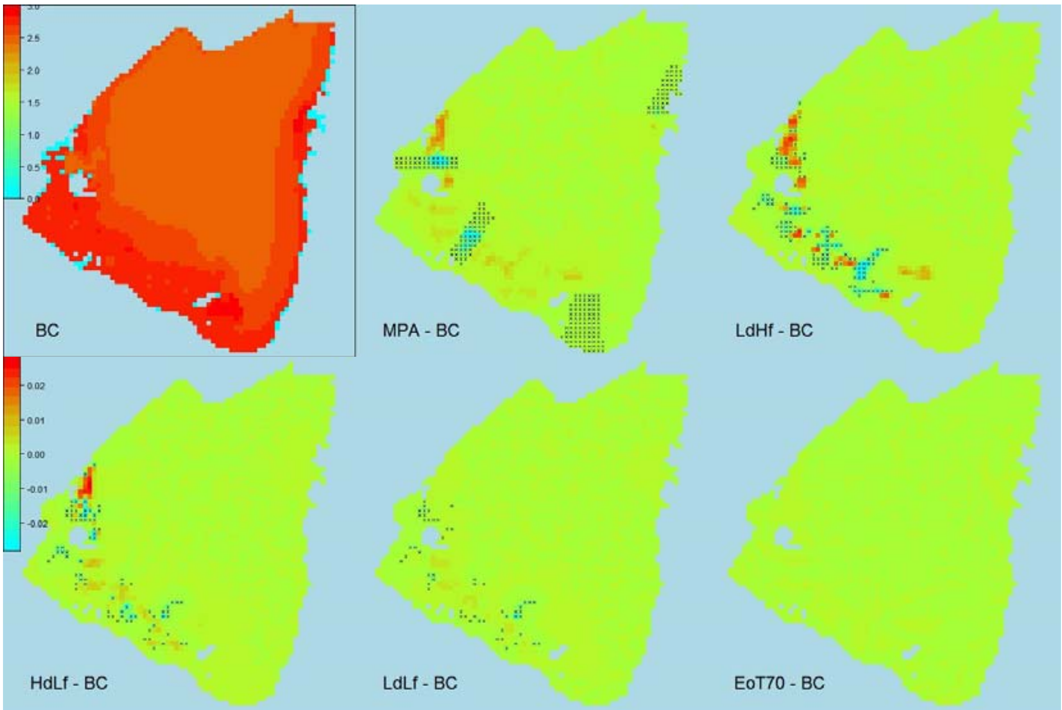
Dolpins



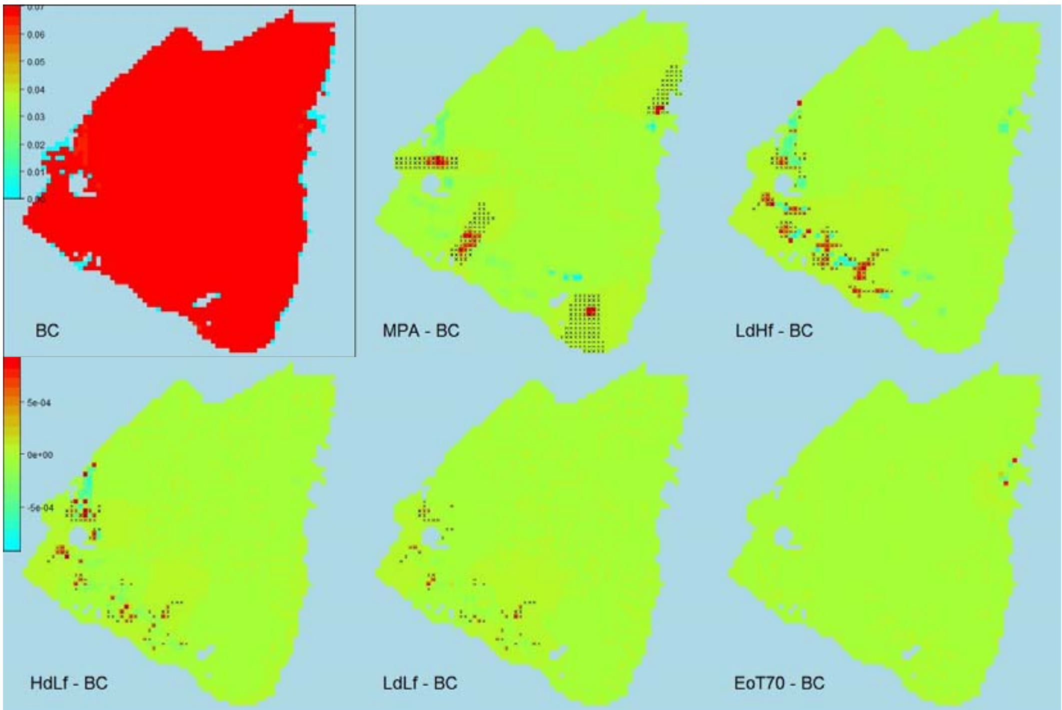
Dugongs



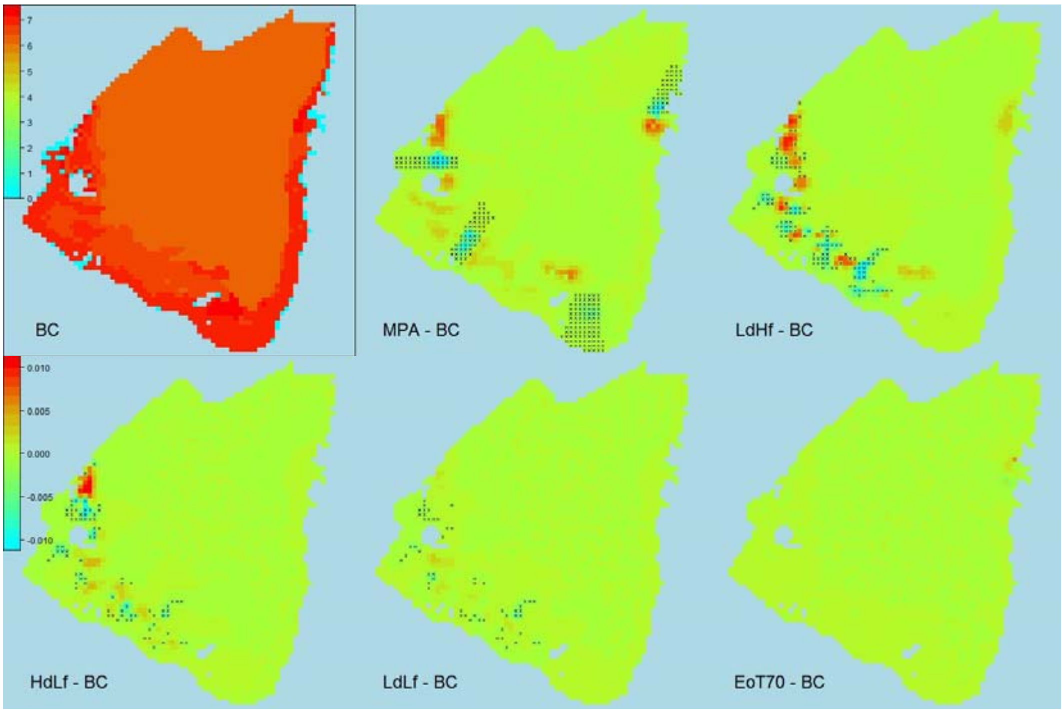
Echinoids



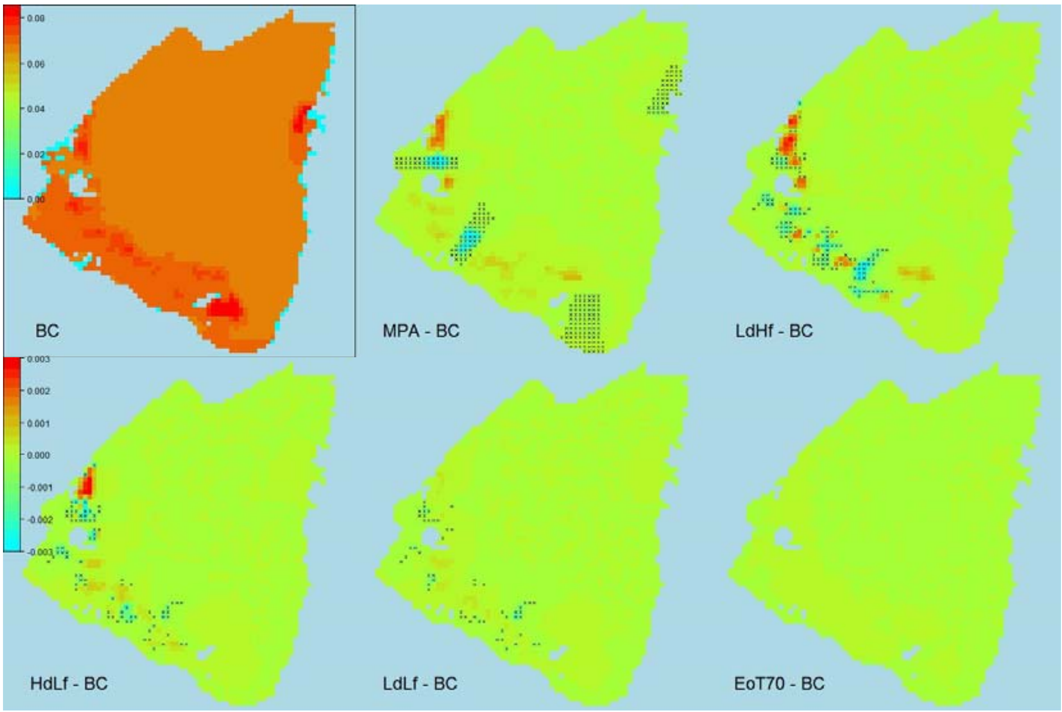
Foraminifera



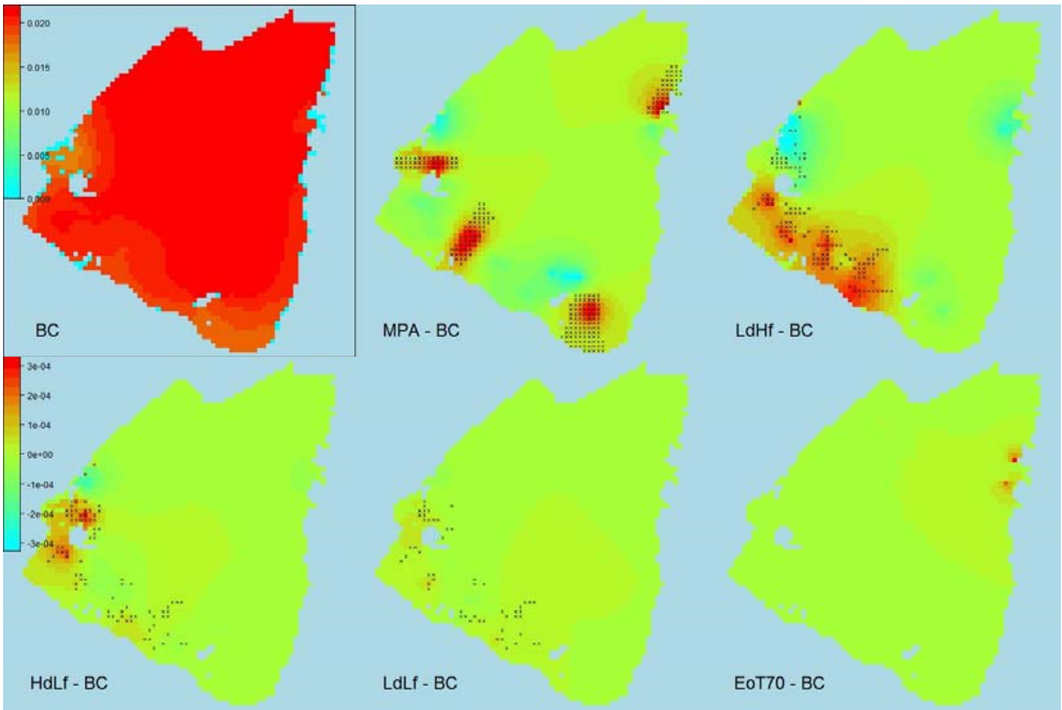
Holothurians



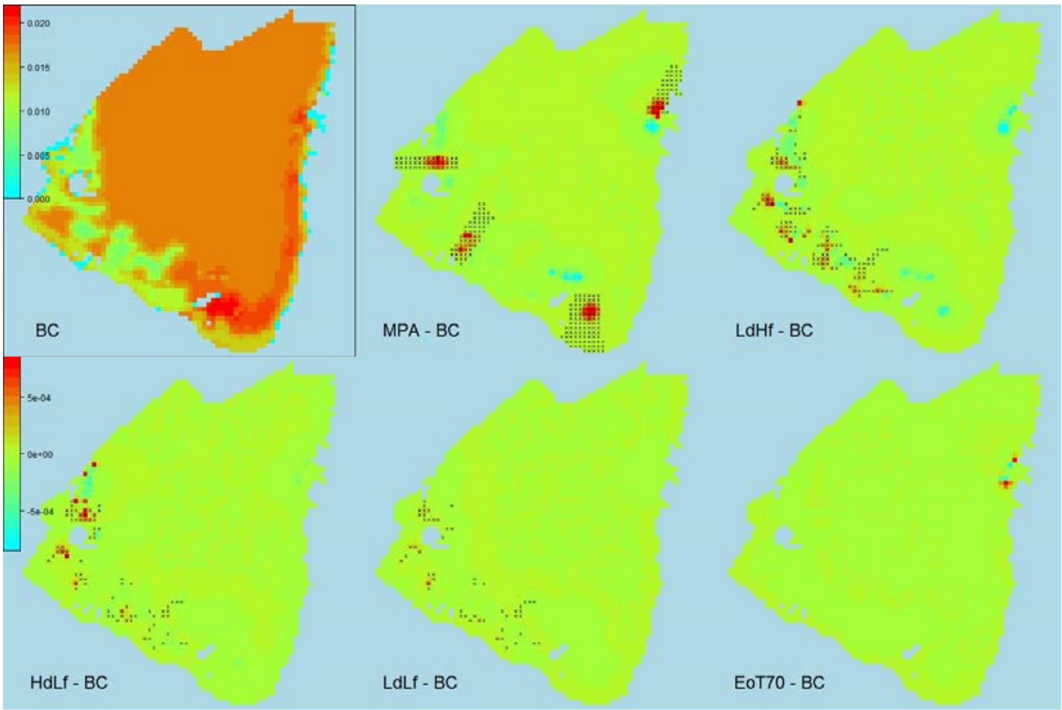
Infaunal Detritritoves/Carnivores



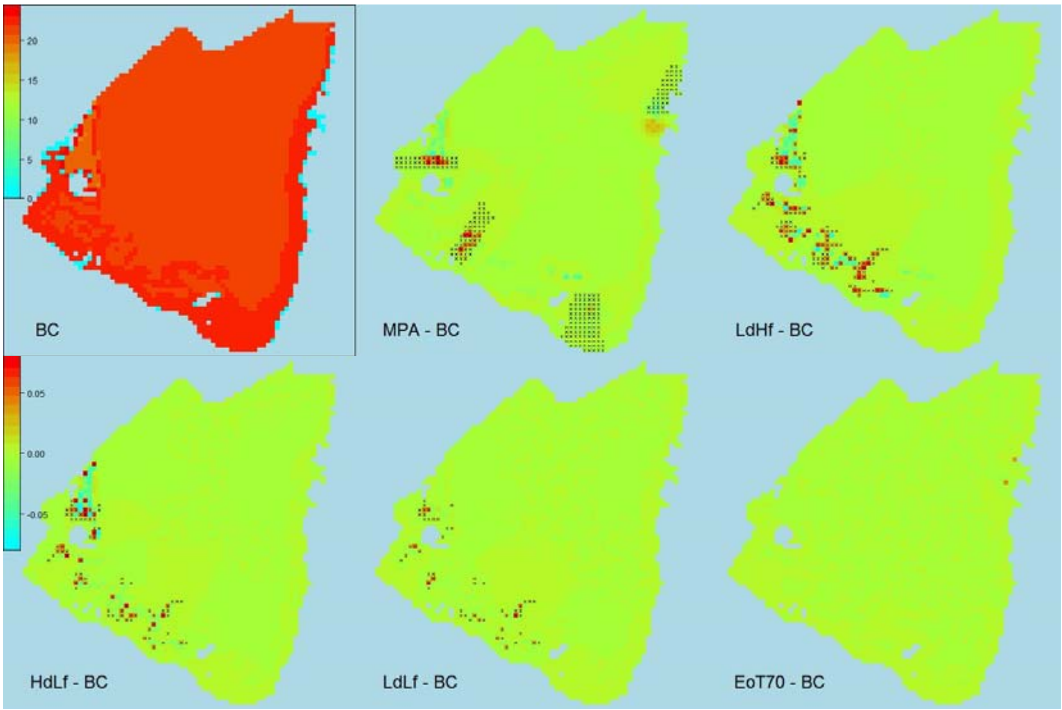
Large Gastropod Carnivore



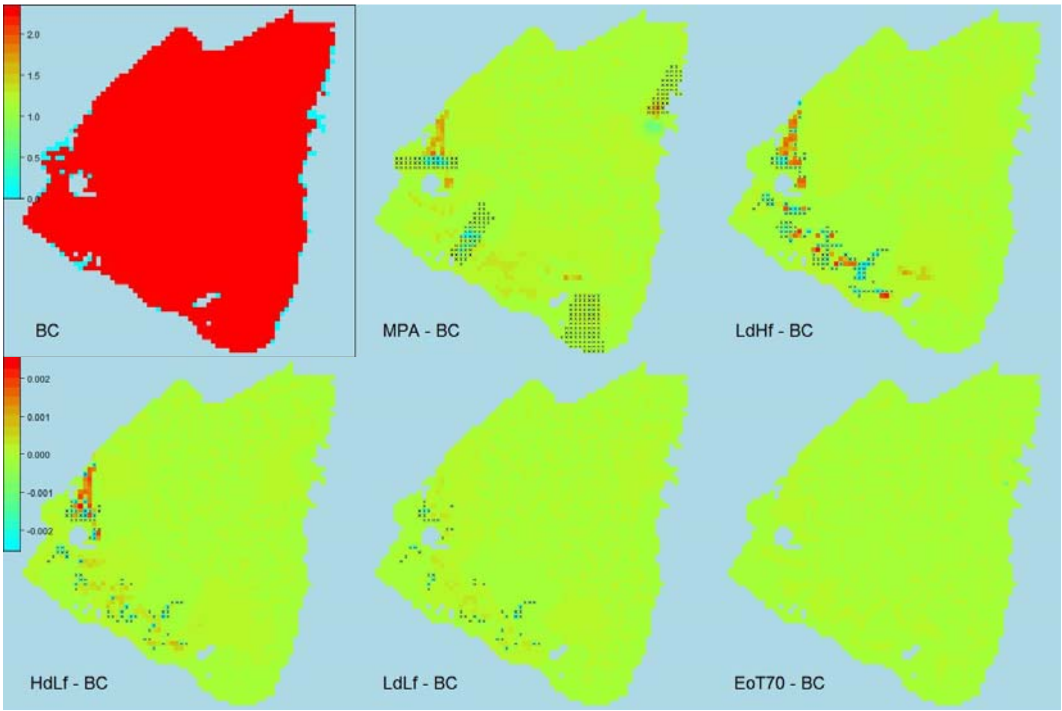
Large Sharks



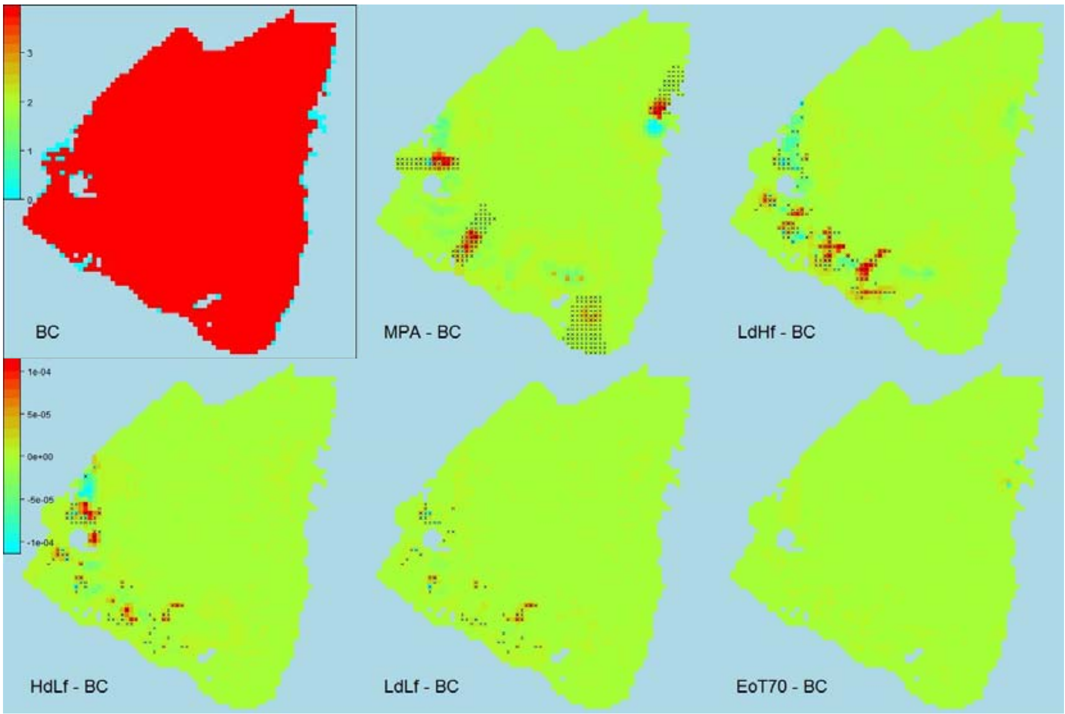
Lobsters



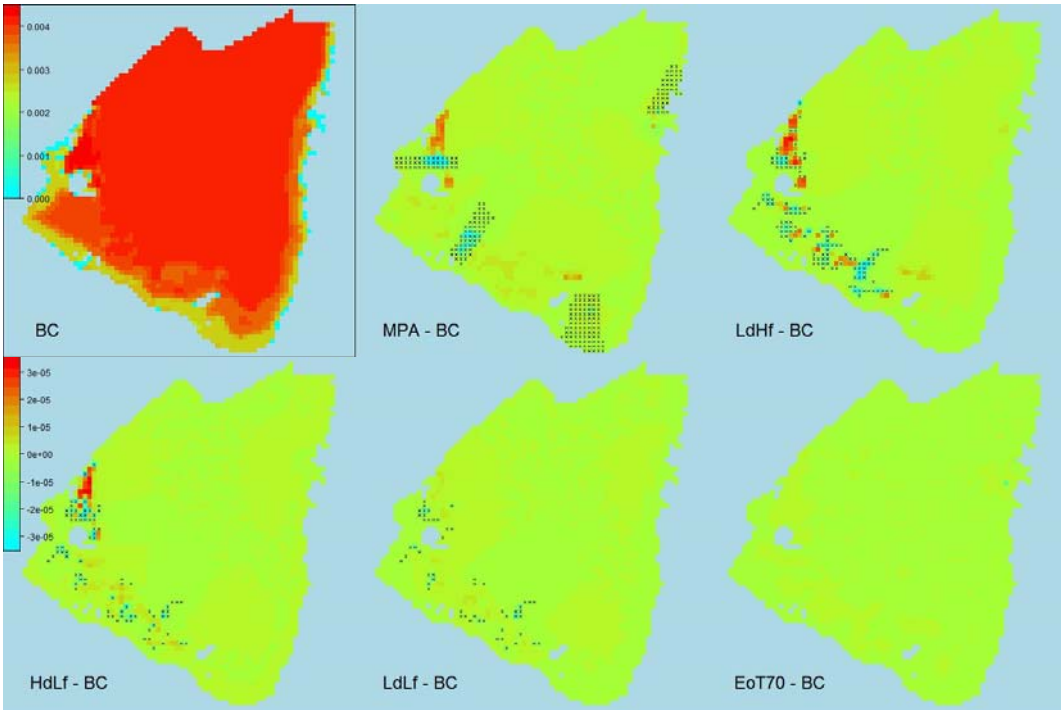
Macroalgae



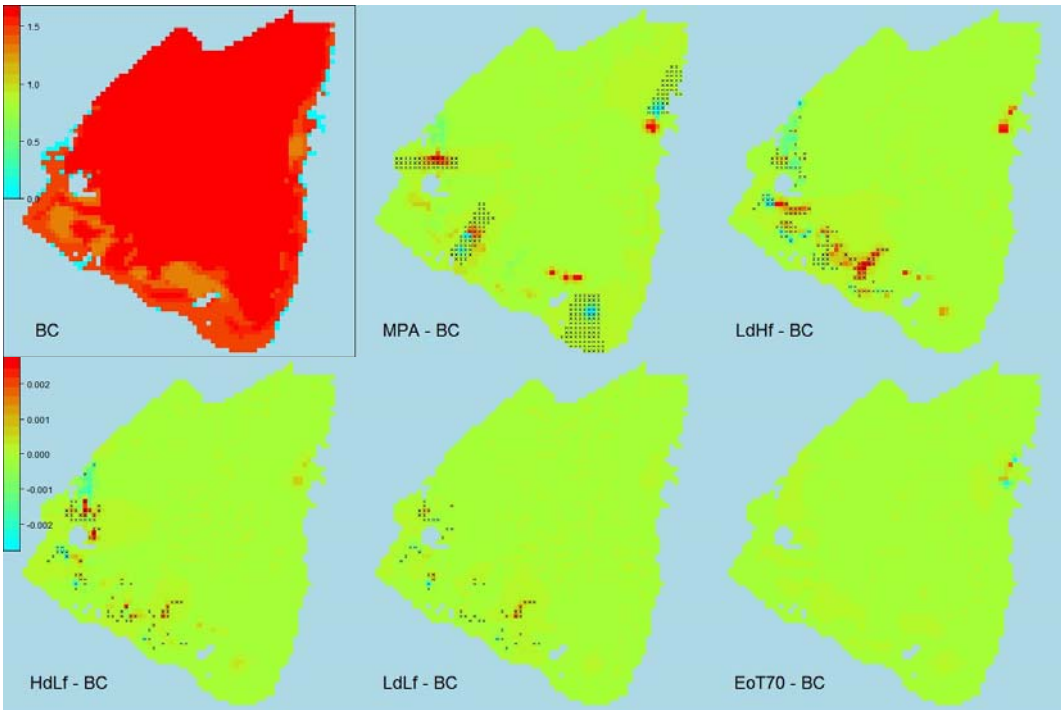
Microbial Heterotrophs



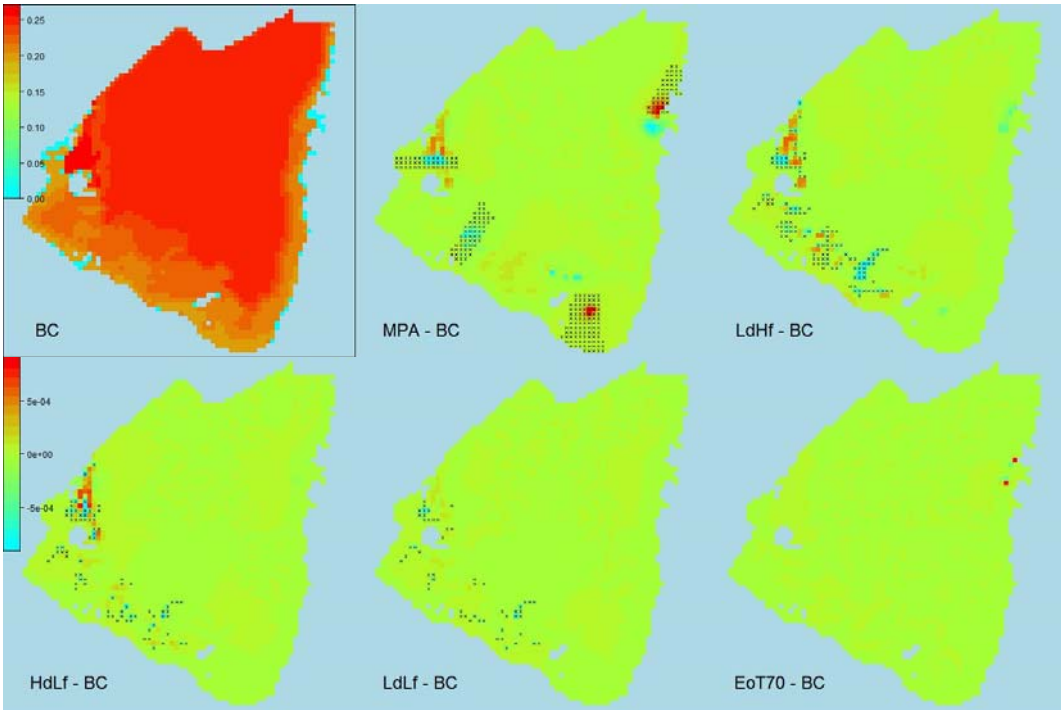
Microphytobenthos



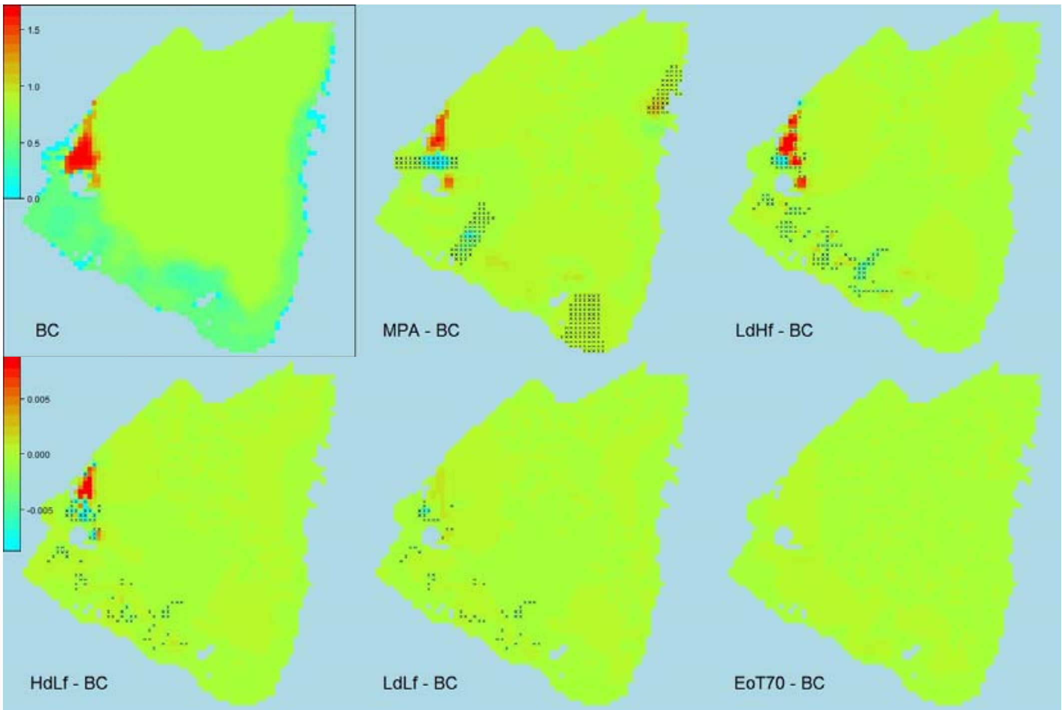
Mud Crab



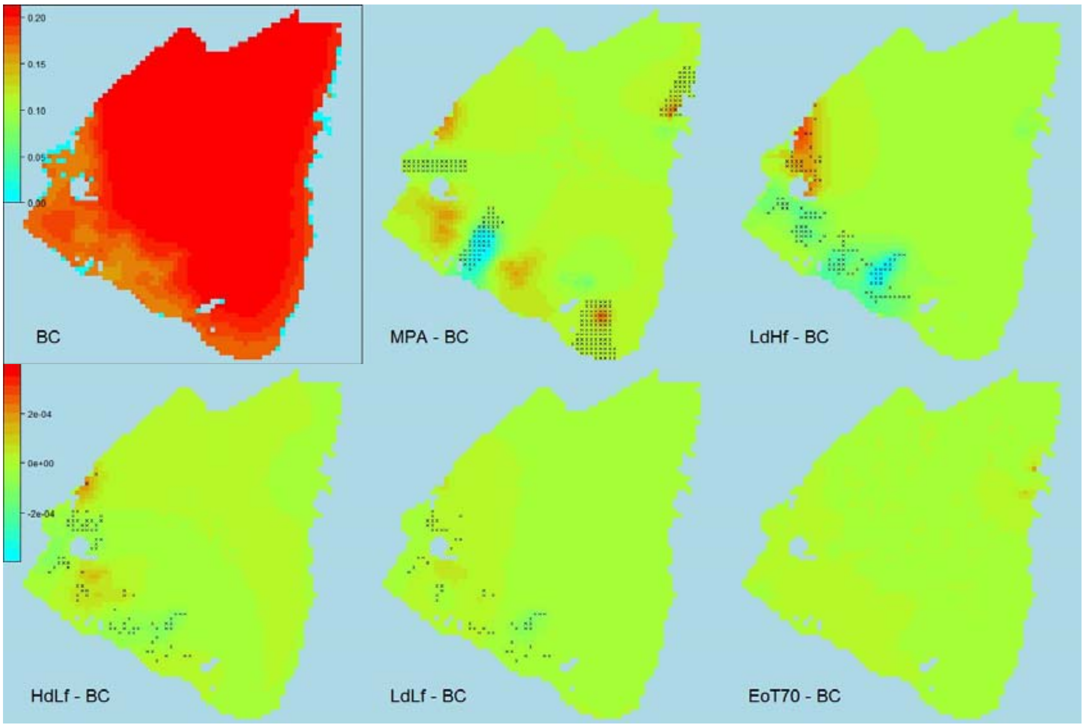
Ophiuroids



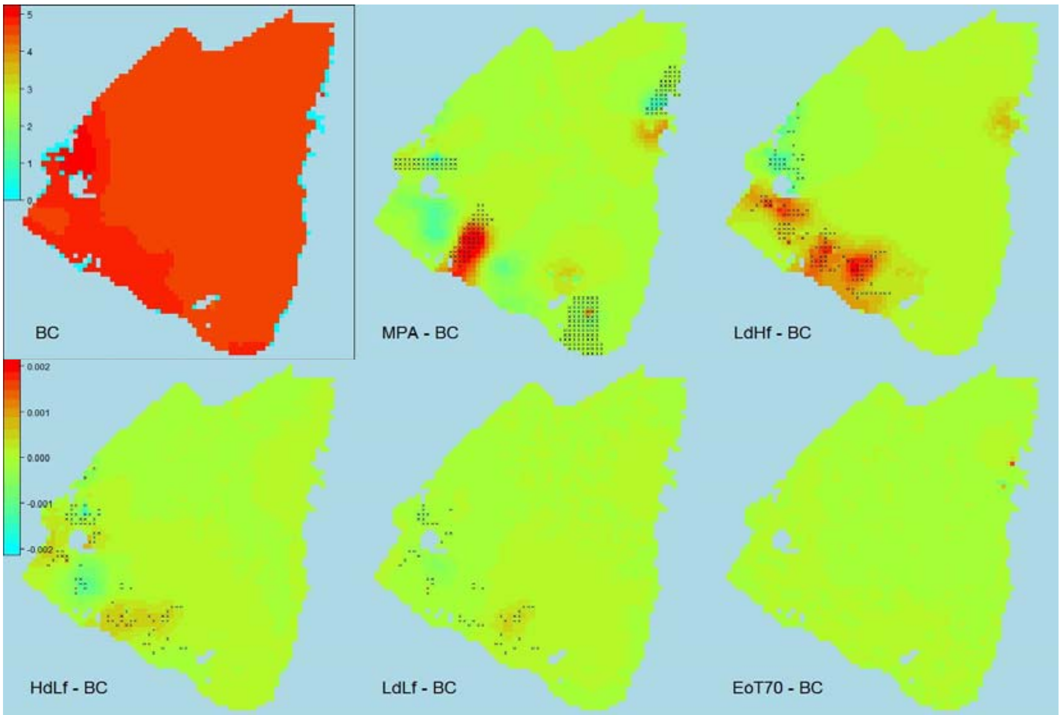
Other Commercial Prawns



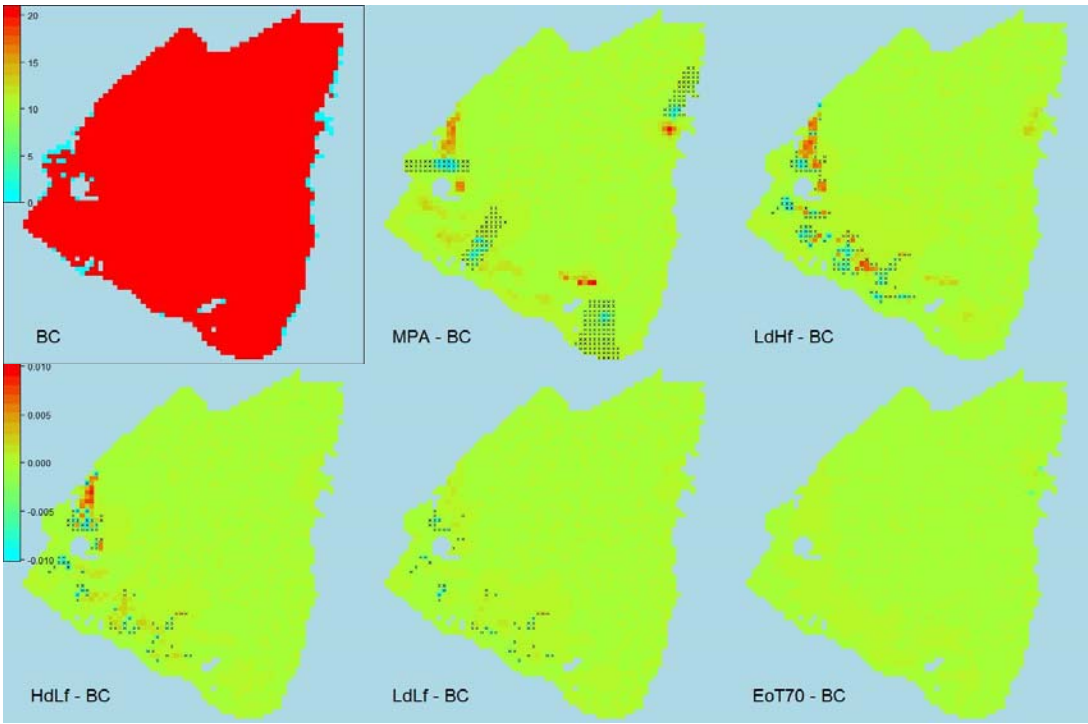
Other Non-commercial Prawns



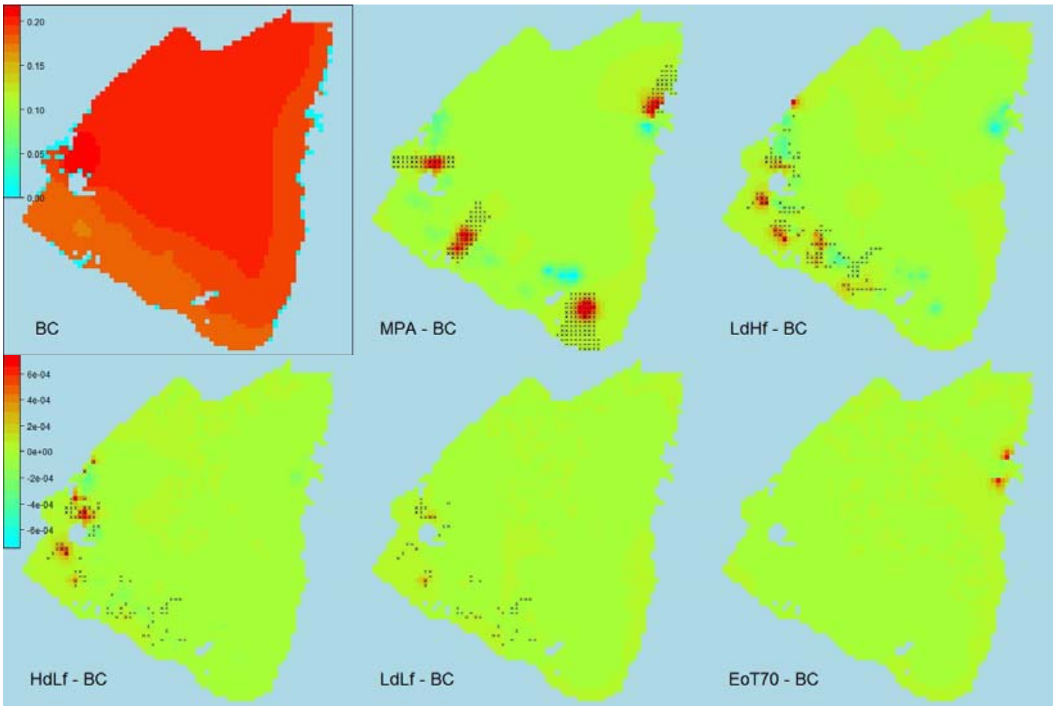
Pelagic Carnivore Fish



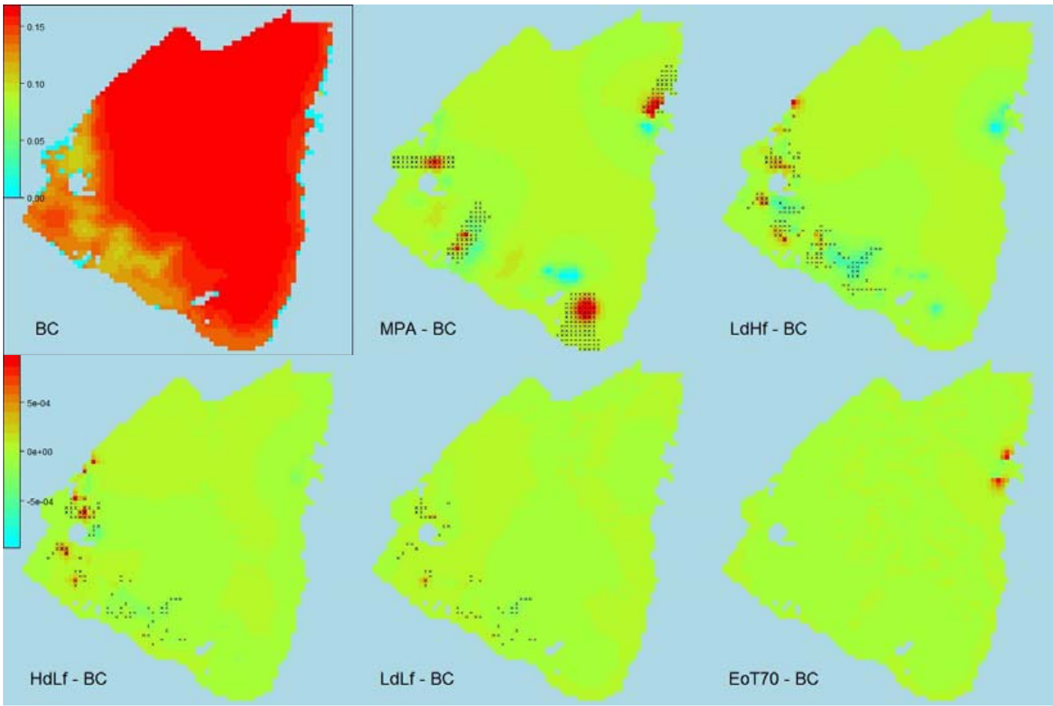
Pelagic Invert Feeders



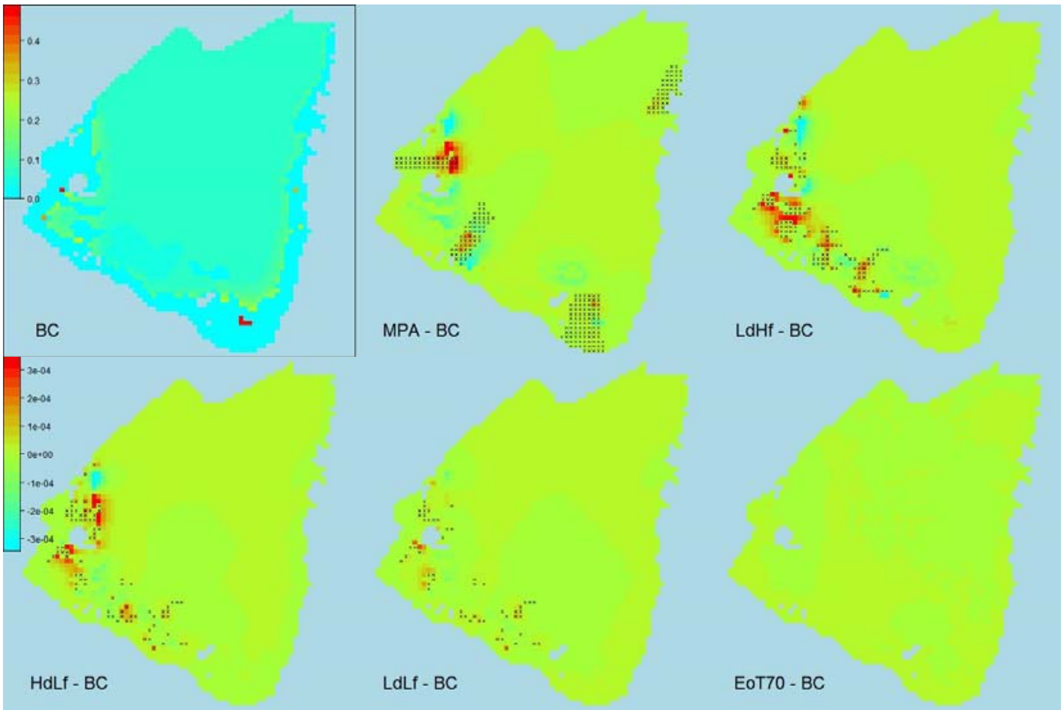
Phytoplankton



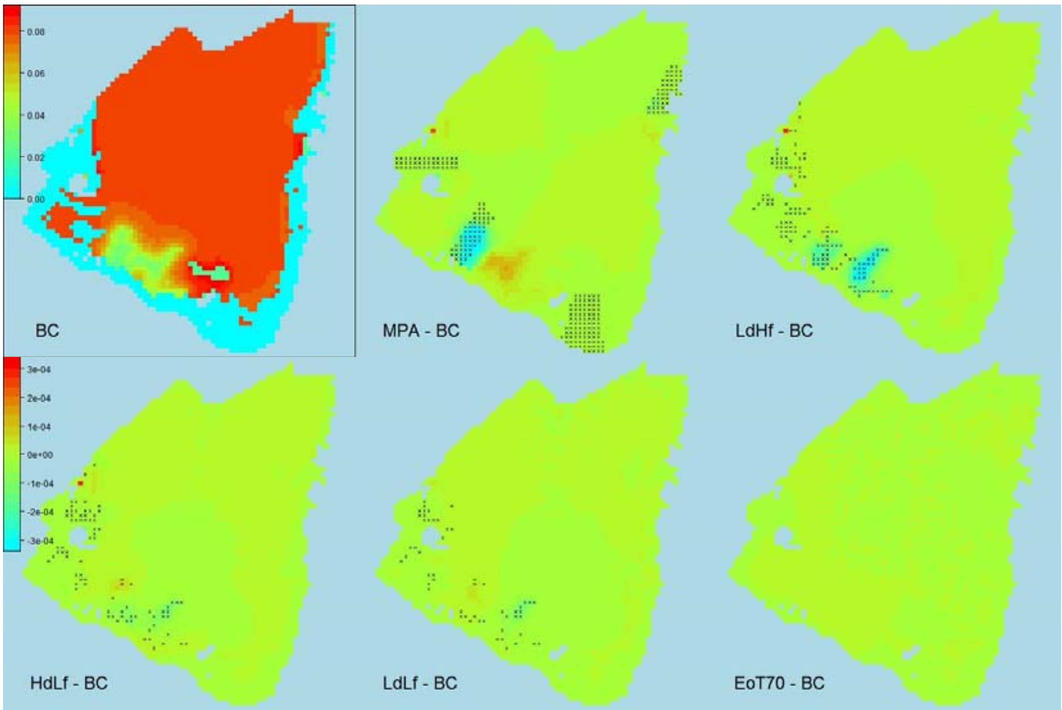
Rays



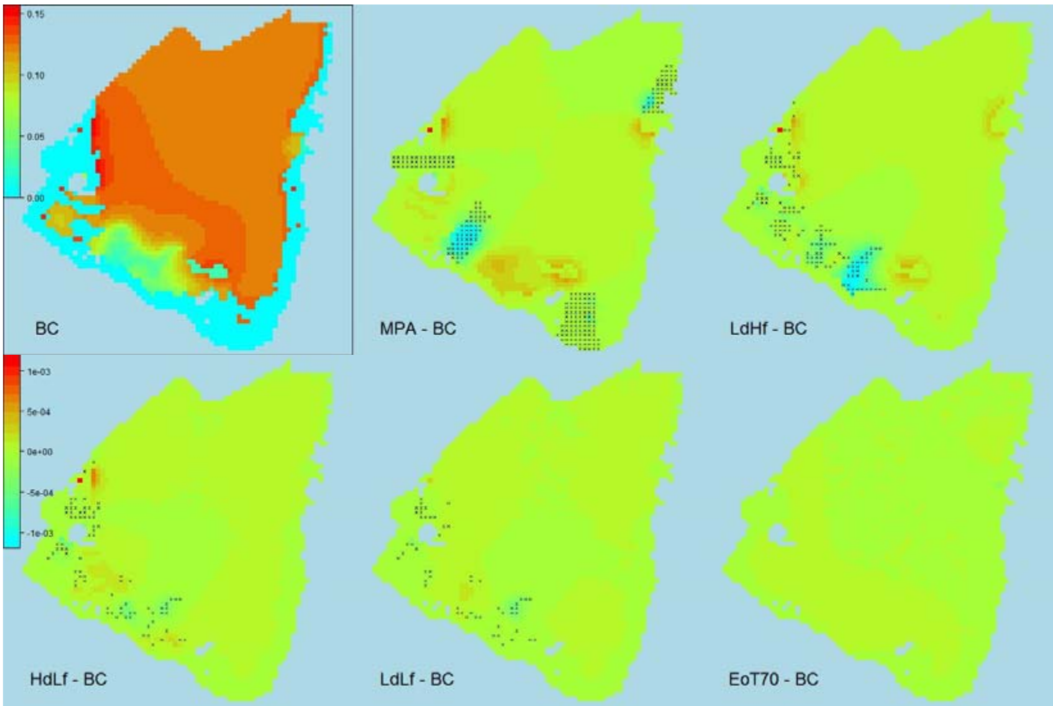
Red Snappers Fish



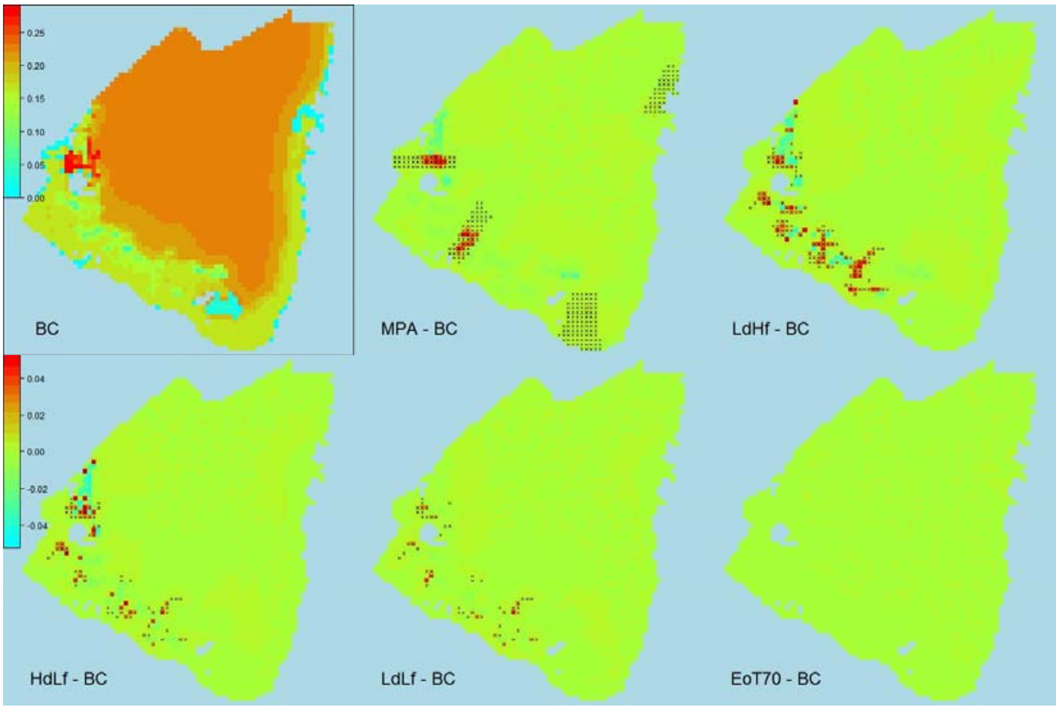
Reef-associated Carnivore Fish



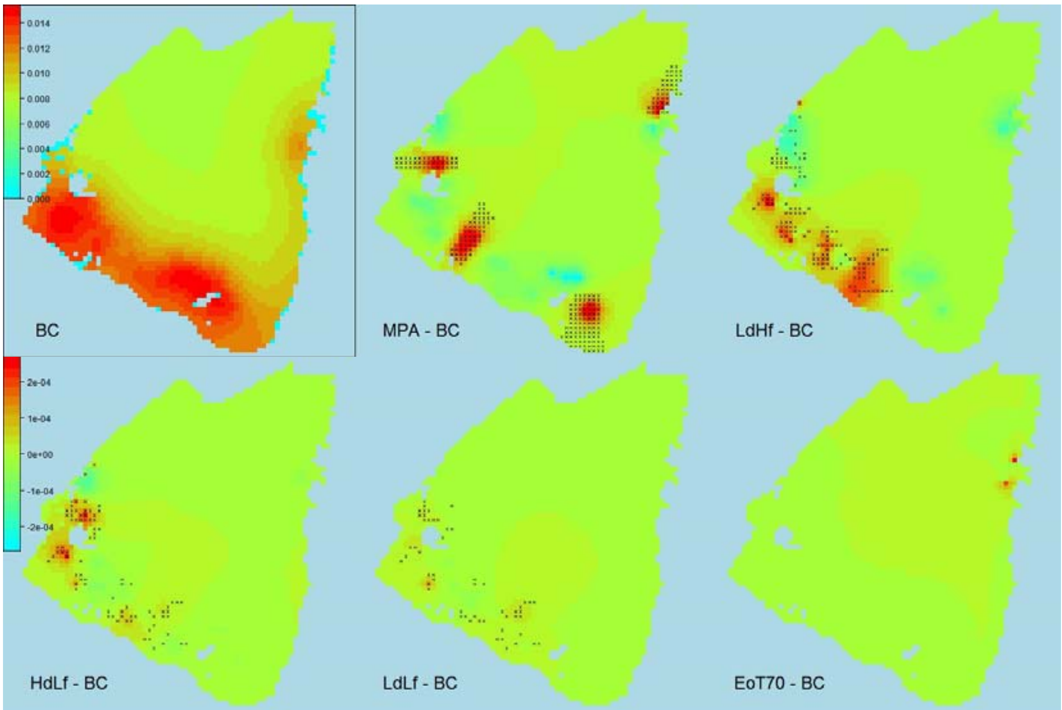
Reef-associated Herbivores Fish



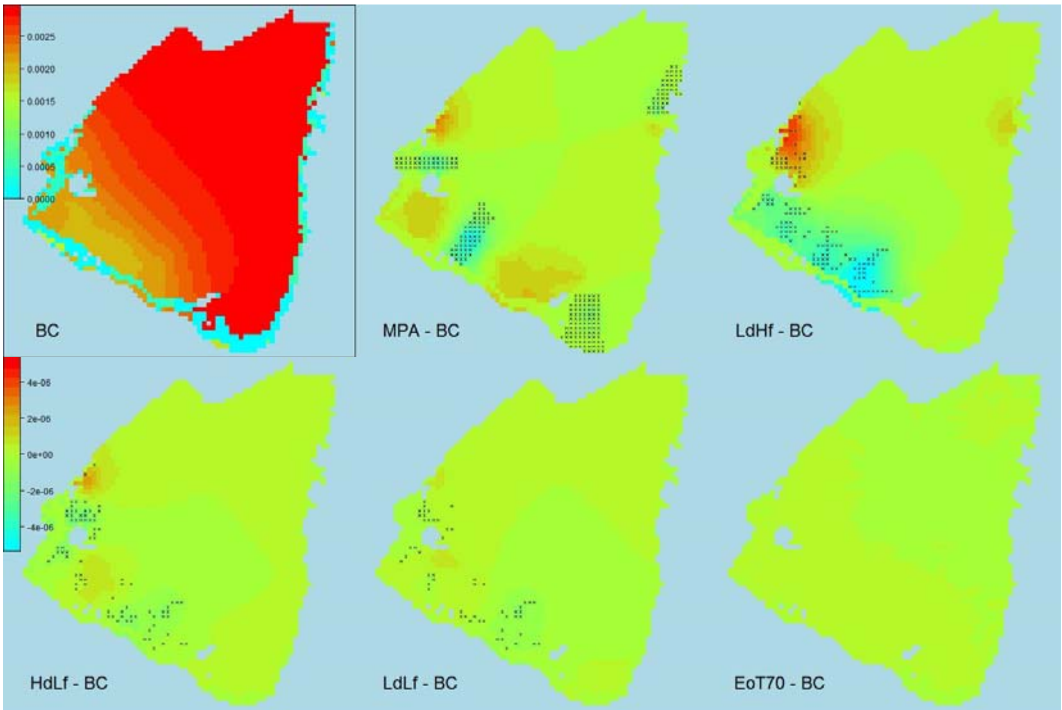
Reef-assoc Invertebrate Feeders



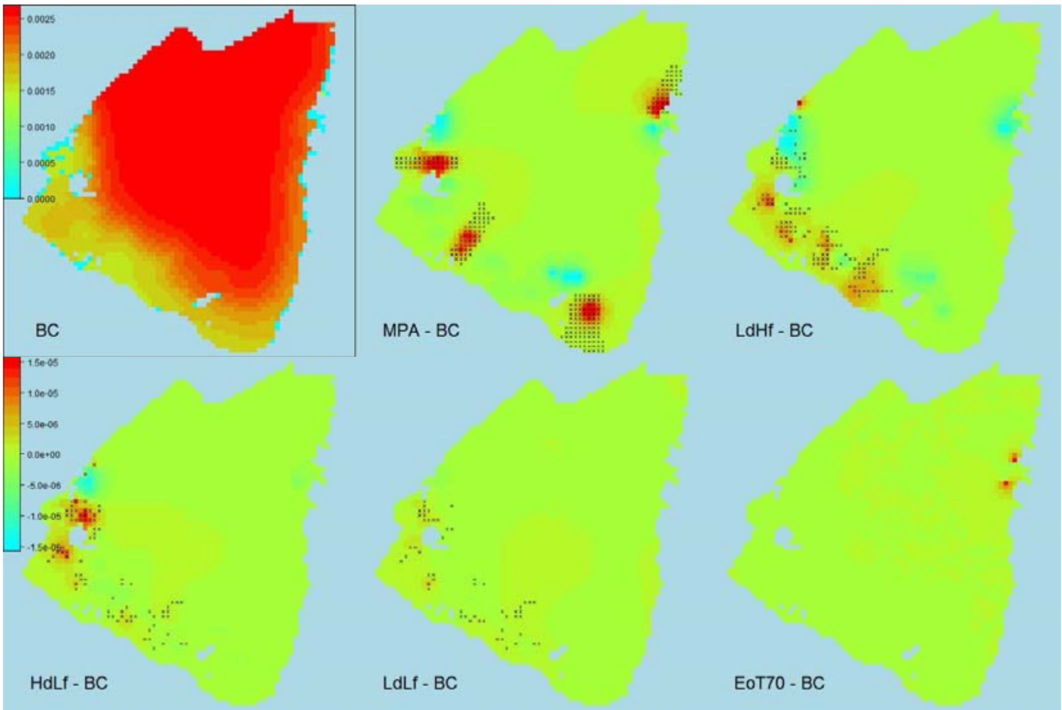
Sand and other large crabs



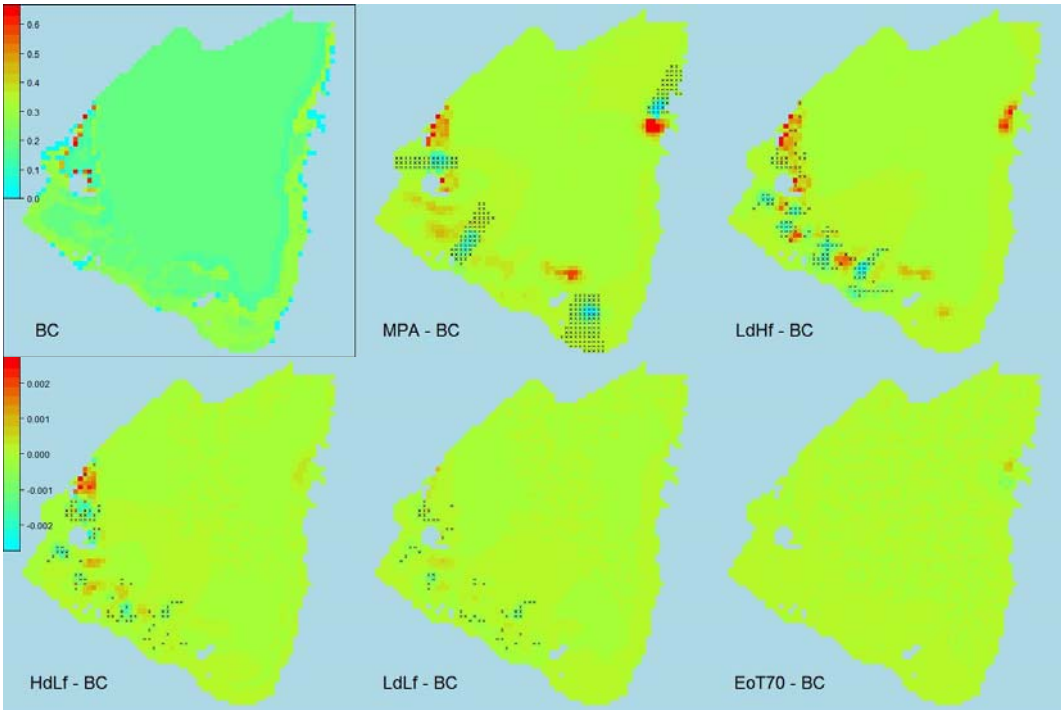
Sawfishes



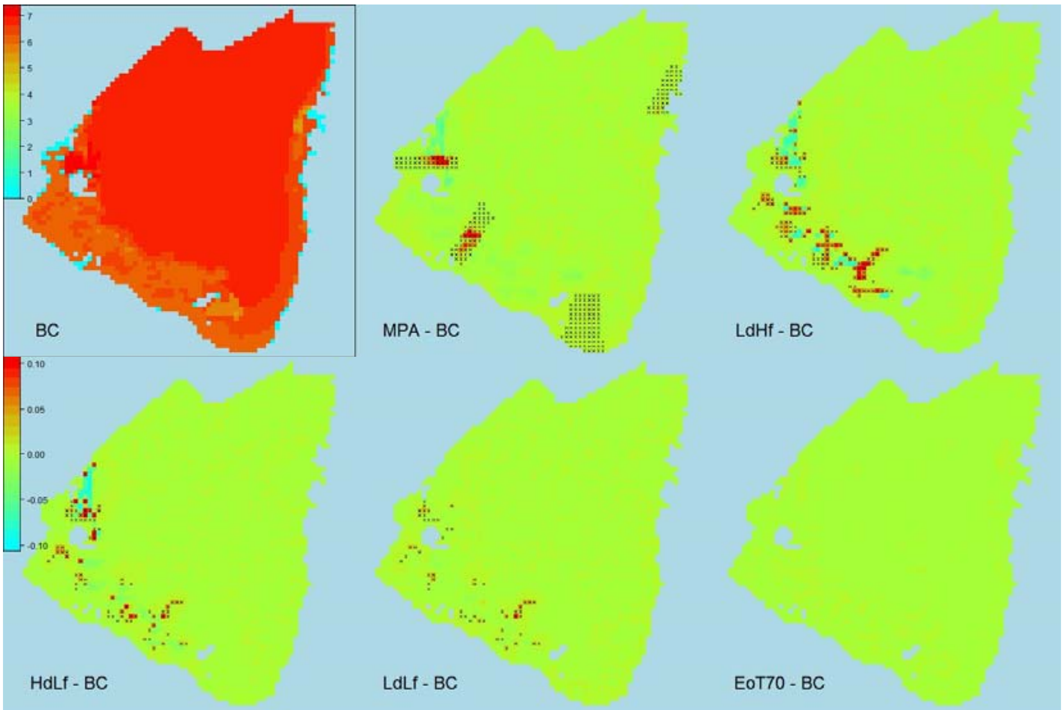
Sea birds



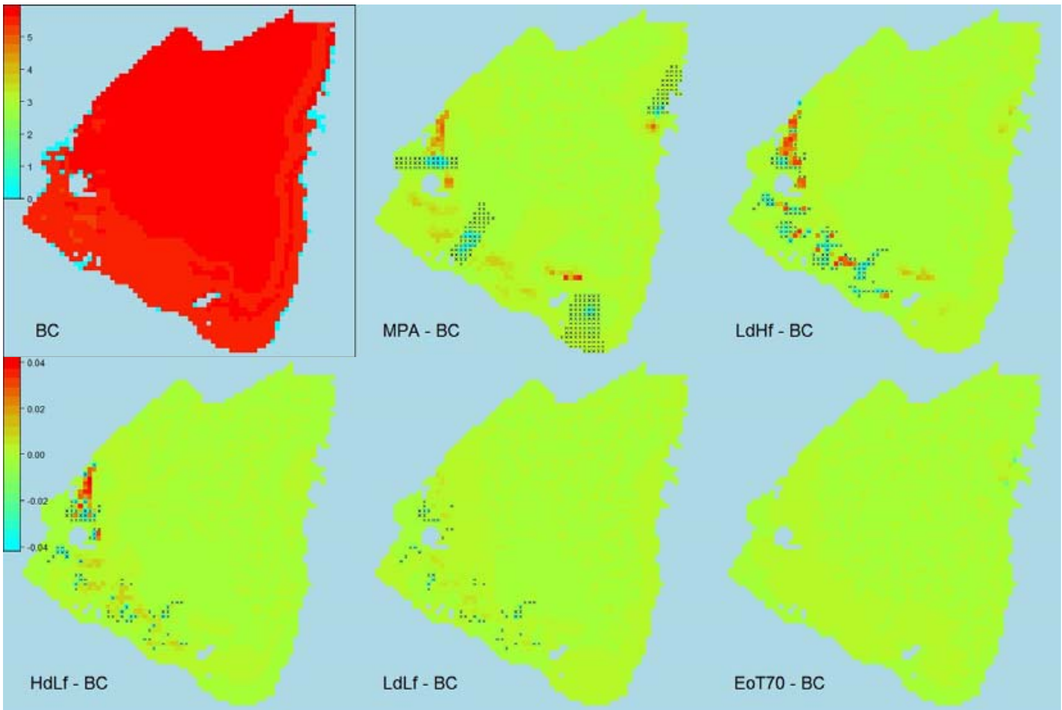
Sea snakes



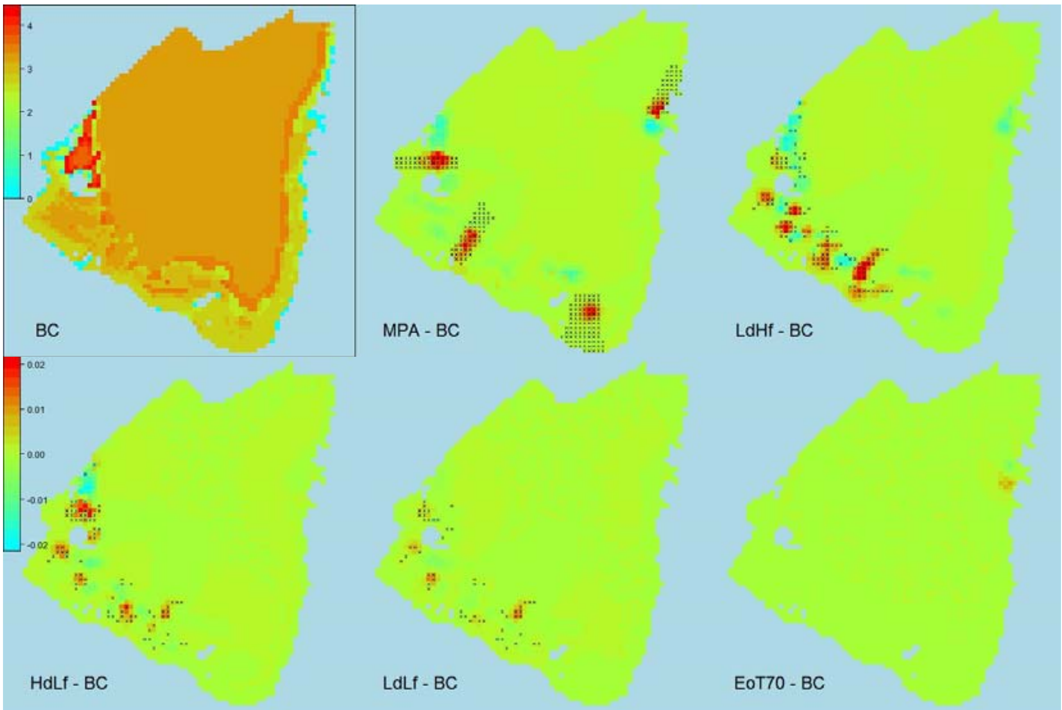
Seagrass



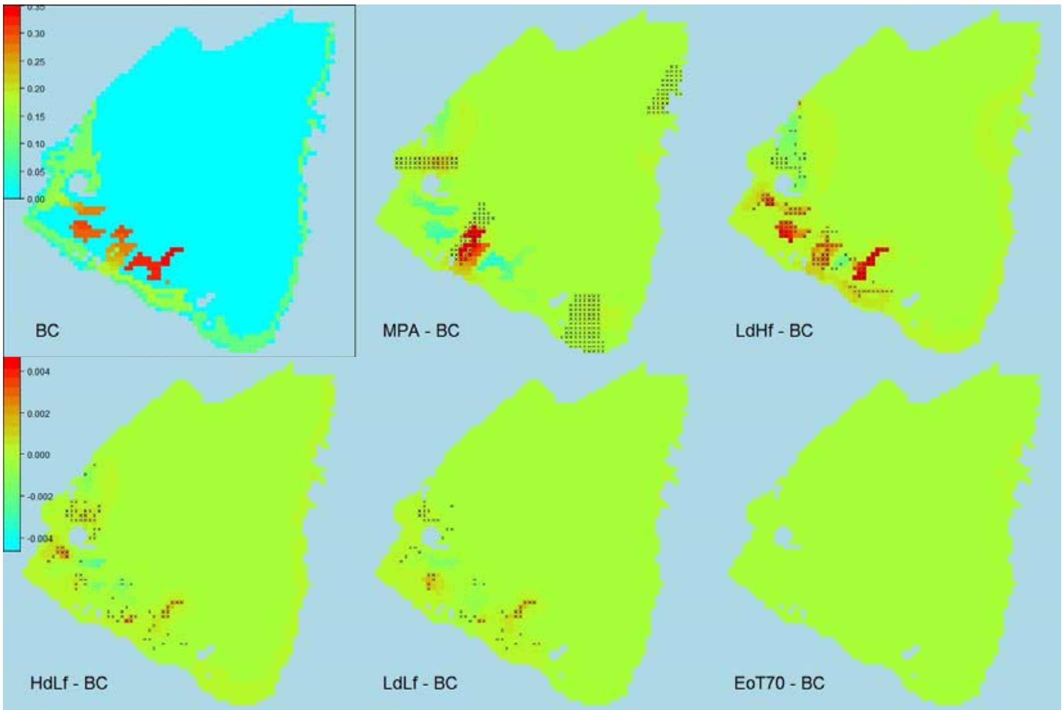
Sessile epibenthos



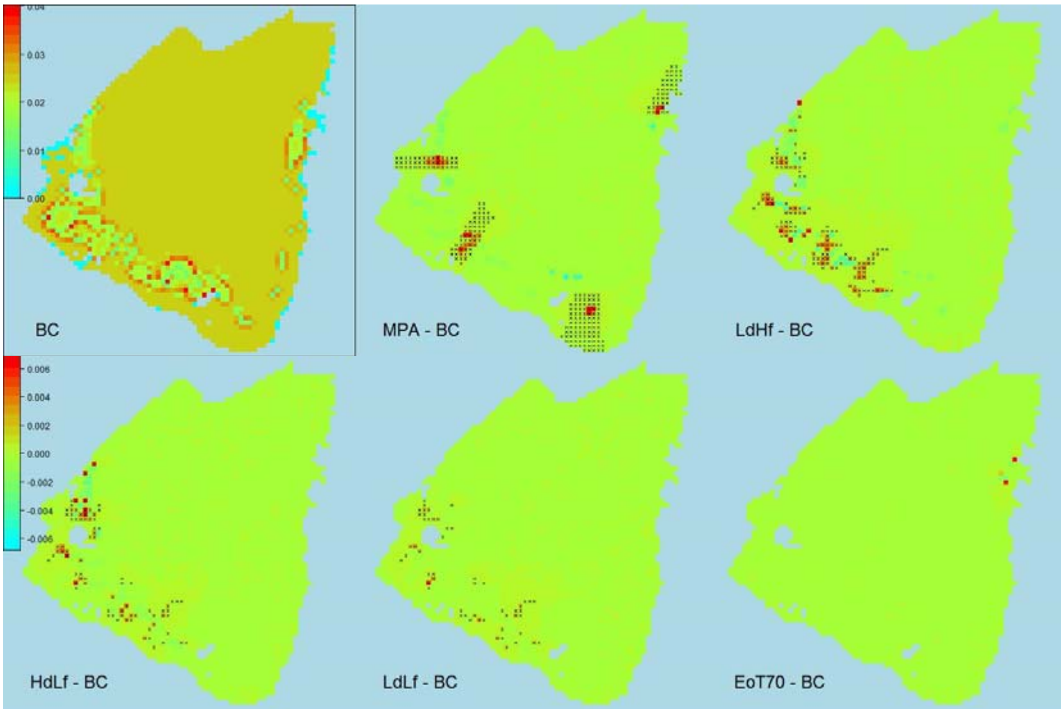
Small crustaceans



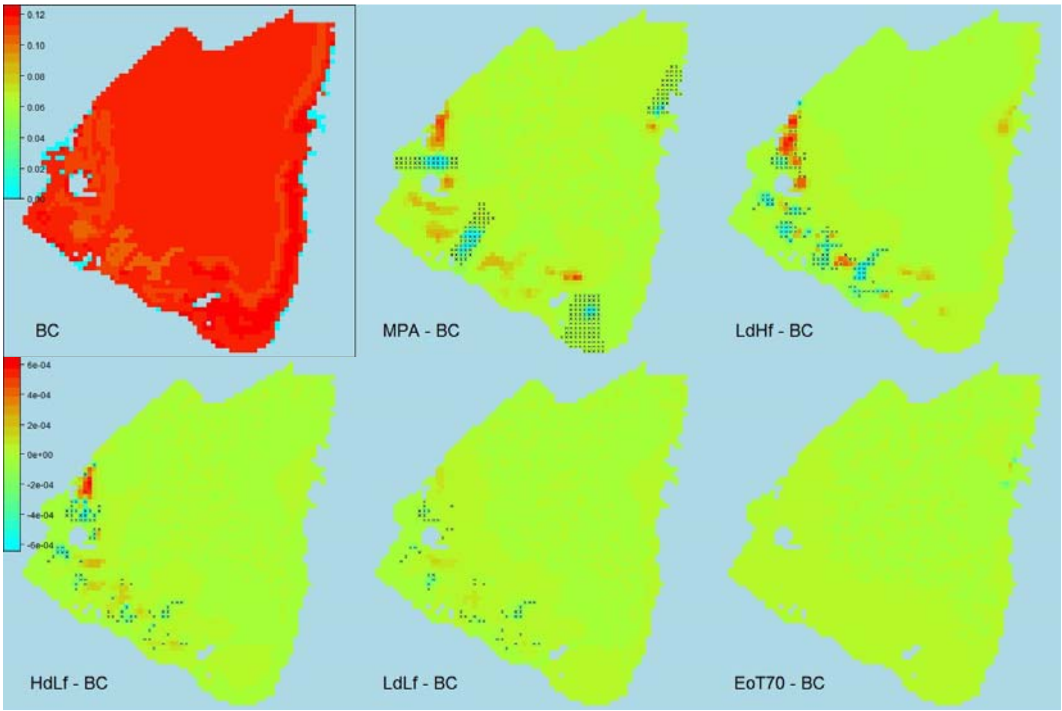
Small Gastropod omni



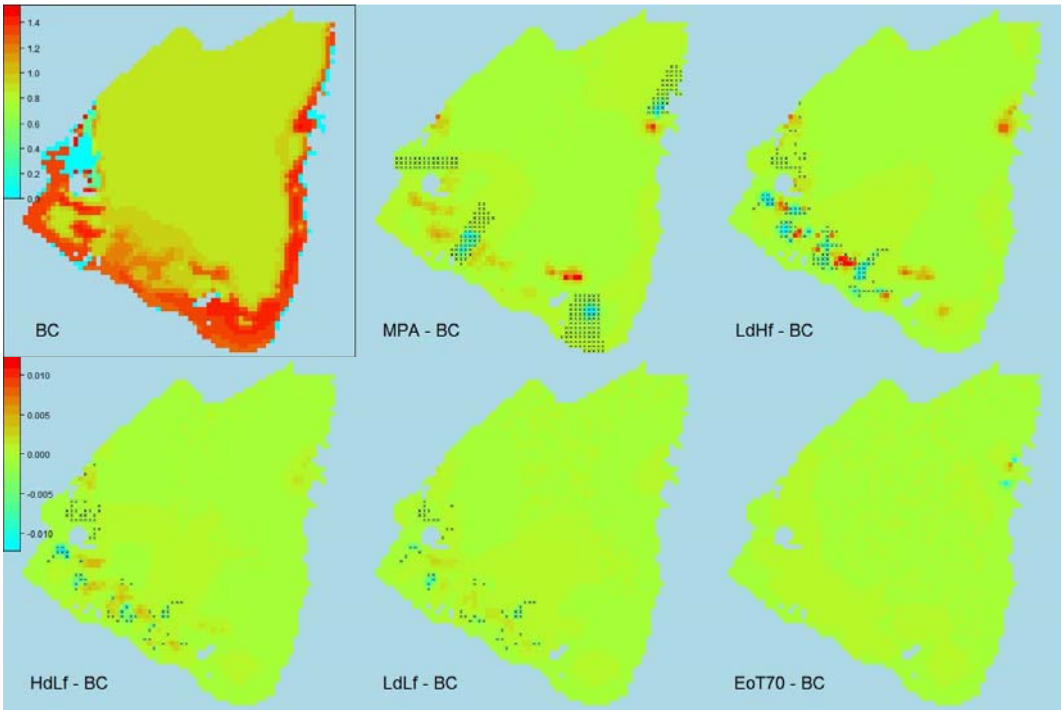
Small sharks



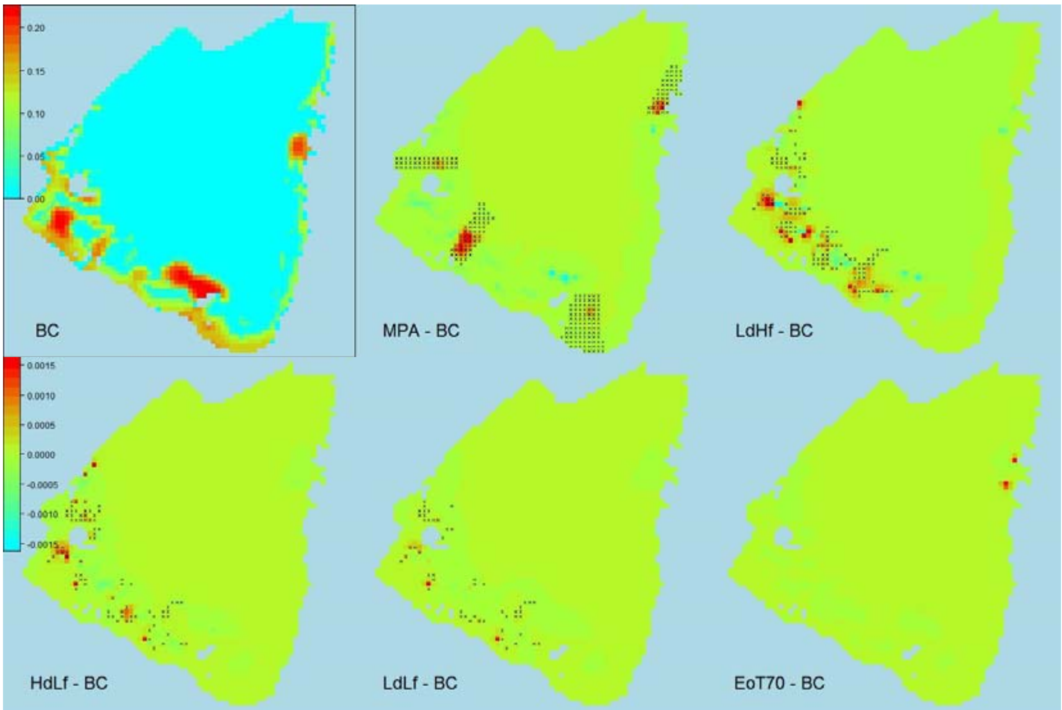
Spatangoids



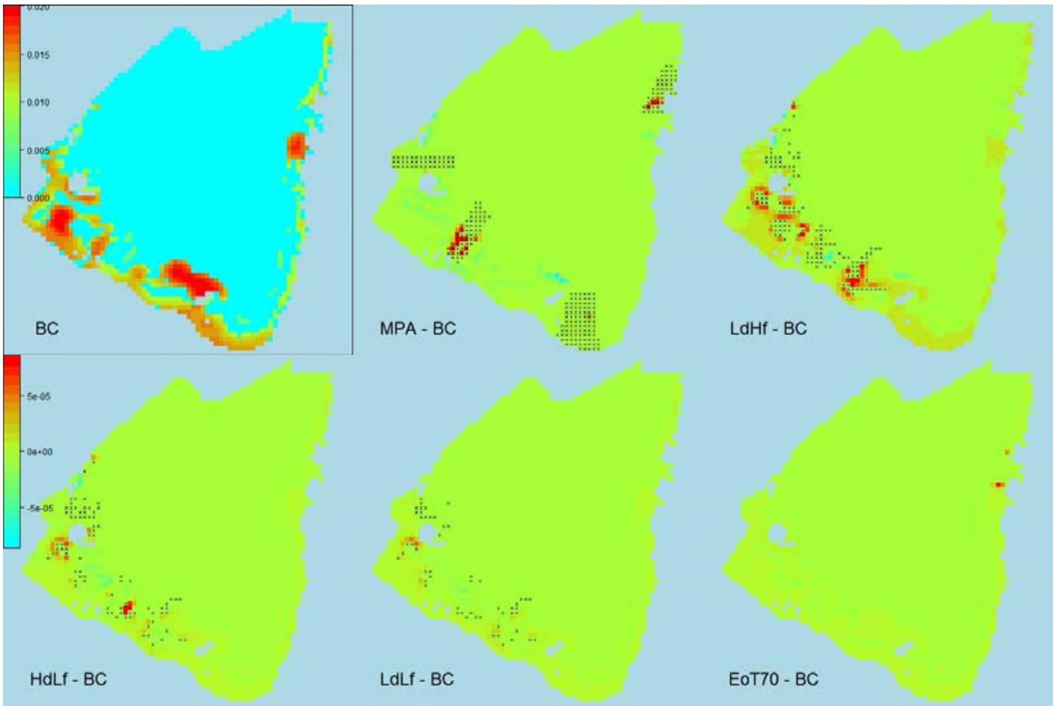
Stomatopods



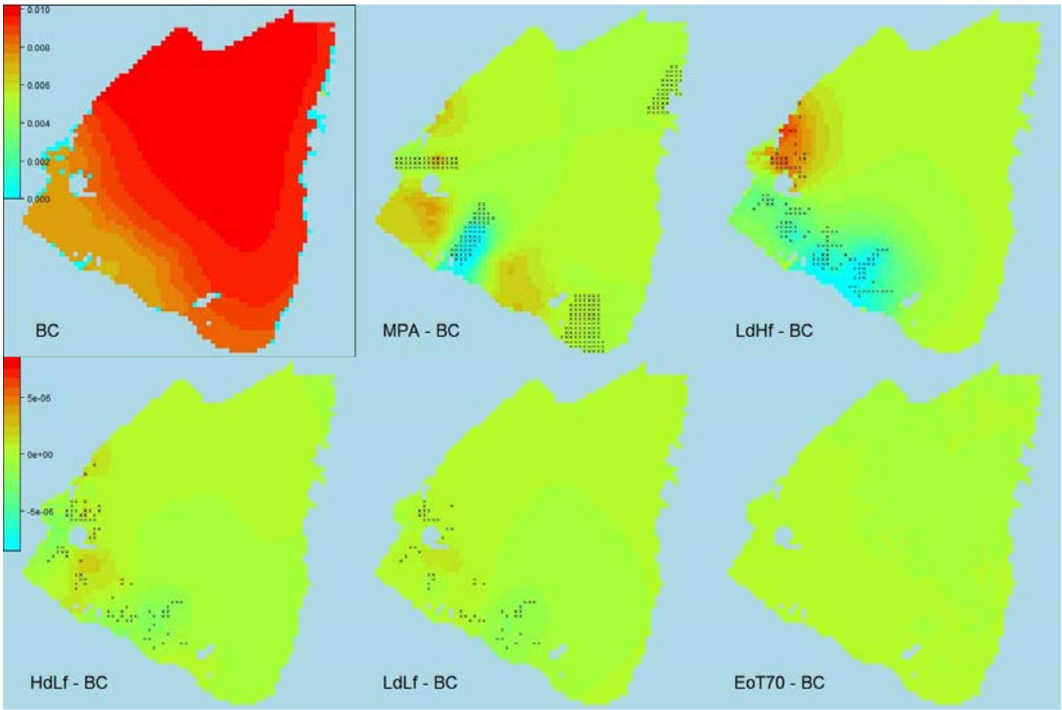
Thallasinid prawns



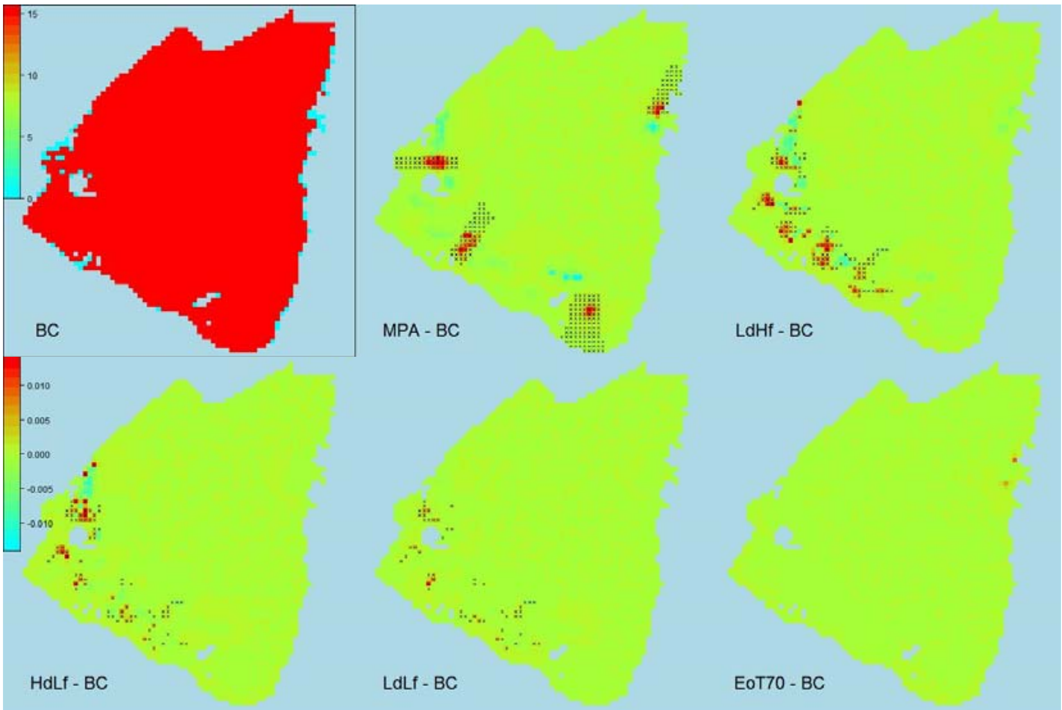
Tiger prawn adults



Tiger Prawn juvenile



Turtles



Zooplankton

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