Torres Strait Management Scenario Evaluation: Effects of Trawling
Final Report

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PREFACE

This document contains the final report for the CRC Torres Strait project task T3.3: Integrated ecosystem modelling for evaluating multiple-use management strategies. This report also indicates that the final milestone ‘Implementation of MSE models and production and reporting on management strategies’ has been reached.
This project set out with two main purposes: firstly, to develop, build and apply a model that allows us to estimate the direct effects of trawling on benthic biota under various scenarios of prawn fisheries management, and secondly to examine how to extend such a model to include the follow-on effects of trawling on the wider ecosystem.

To meet the first objective, an ‘effects of trawl’ model for the Torres Strait prawn trawl fisheries was constructed and populated with the available data for the region. This model allows us to estimate the amount of benthic biomass removed by applying the large scale distribution of trawling effort (in time and space) on the distribution of benthic biota, combined with our existing knowledge of the vulnerability of such biota to trawling. The distribution of benthic biota was a product from the Torres Straits benthic surveys in TS-CRC task T2.1.

The ‘effects of trawl’ model is embedded in a larger set of models that allow the application of various scenarios for the management of the prawn trawling activities (management strategy evaluation or MSE). Four scenarios of management actions for the Torres Strait fisheries were defined by the stakeholders (AFMA, TSRA and the people of Masig): (1) closing buffer zones around reefs/islands to prawn trawling, (2) fisheries closure four days either side of full moon, (3) three Total Allowable Effort (TAE) quota options and (4) the closure of an area of 10 km around Masig for the recovery of benthic biota. A fifth scenario that maintained the status quo was defined to allow the relative comparison of the results of management actions to a fictitious ‘no-change’ scenario. Apart from the four scenarios, stakeholders also defined five measures to represent the performance of each management scenario: (i) changes in benthic biomass, (ii) benthic impact status for a set of vulnerability classes, (iii) maps of effort change, (iv) total expected change in prawn catch and CPUE and (v) percentage effort reallocated by management actions.

The results of the evaluations can be summarised as follows:

- in general: the more spatially concentrated the trawl effort is, the less impact (in the form of the overall removal of benthic biomass) it is expected to have. Spatial concentration may be in the form of aggregated trawling patterns or of concentration in certain areas of the region.
- if an area has already been depleted or has relatively little benthic biota, closing such an area to trawling without appropriate effort reduction is likely to result in the increase of trawl impact with respect to overall removed benthic biomass. The reason for this is that the trawl effort will be displaced into areas with relatively...
high density of benthic biota.

- We did not find that closing relatively small areas around reefs and islands would result in a considerable decrease in overall benthic biomass. This may partly be caused by the spatial resolution of the trawl effort data.

- Within the range of scenarios available, effort reduction was found to be the main mechanism to decrease the effects of trawling on benthic biota.

- Closing the prawn fishery four days either side of full moon concentrates the fishing effort to times where the CPUE is higher. The overall effect of moon closures increases the CPUE a little less than 3%.

- Spatial closures, such as a closure around Masig or the reefs and islands, make very little difference to the effects of trawling on overall benthic biomass if the effort is displaced (instead of removed).

- Using comparative performance measure of scenarios against a ‘no-change’ (status quo) scenario increases the robustness of the performance measures considerably.

- The main source of uncertainties presented in the scenario results is caused by the inter-annual variation of trawling effort that is sampled by the stochastic scheme to re-run the same scenarios. These uncertainties are underestimated as they do not include process variances e.g. uncertainties in the recovery and depletion rates or pre-trawl biomass estimates. Future work is needed to get better estimates of those uncertainties.

This project has resulted in a well-structured and functional model to estimate the effects of trawling in the Torres Strait. The four scenarios that were evaluated have given us some insights of how management actions would alter the effects of trawling on the benthic biota.

Looking into the future of the effects of trawling model, the improvement of the estimation of uncertainties would have a high priority but presumes the availability of data to better estimate process uncertainties. Also, to get better estimates of the effects of various management scenarios on the fishing industry, the effects of trawling model should ideally include a spatially explicit prawn population model and at least a basic notion of economic factors that drive the industry.

The second objective was to examine ways to extend our ‘effects of trawl’ model to include follow-on effects of trawling on the wider ecosystem to prepare ourselves for a follow-on project. We implemented a simple ecosystem model of the Torres Strait region. This preliminary ecosystem model includes a range of trophic interactions between functional and taxonomic groups for which we have some data. The ecosystem model and some of its
preliminary results are presented in this report.

We also performed a literature study into the wider effects of trawling (apart from biomass removal) on the ecosystem to create an inventory of mechanisms through which trawling could affect the ecosystem. Such an inventory would allow us to include more comprehensive effects of trawling on the wider ecosystem, needed for ecosystem-based fisheries management. It also would allow us to better specify the data needs to populate these model extensions.

Looking into the future for the integration of the trawl effect model, the findings from the literature study need to be incorporated into the preliminary ecosystem model. The resulting (extended) ecosystem model needs to be linked to the ‘effects of trawl’ model. This would allow us to better estimate the effects on the wider ecosystem of selectively removing species or functional groups in the future.

We hope and expect that the results of the four management scenarios presented in this report only form a starting point, awaiting active feedback from the stakeholders in the Torres Strait prawn fishing. It is the iteration between stakeholders and MSE results that generate many of the synergies needed to responsibly manage the use our natural resources.
1. PROJECT DESCRIPTION AND BRIEF

1.1. Introduction

This report describes the results of task T3.3 of the Torres Strait CRC: Integrated ecosystem modelling for evaluating multiple-use management scenarios.

The process of Management Strategy Evaluation (MSE, Sainsbury et al. 1992) aims to present the trade-offs resulting from (combinations of) options available to the natural resource manager. To facilitate the generation and presentation of these trade-offs, mathematical models are often used to imitate or predict the responses of the resource to alternative management activities. MSE models include these management activities as a part of the system to be modelled.

Management Strategy Evaluation aims to compare various options for management of a natural resource. This CRC task sets out to develop and implement a system to evaluate various options for the management of trawling effort in the Torres Strait. It uses the changes in benthic biota under different trawl management scenarios as its main response.

The first section, the Research Summary, will briefly outline the MSE specification, software development and results to date. The second section (Outcomes and achievements against objectives) describes in more detail the activities and products for each of the project objectives, where available. It starts off with a short introduction into Management Strategy Evaluation.

1.2. Project objectives

The stated project objectives are as follows:

- **Objective 1**: Consult with stakeholders involved in the use and management of Torres Strait natural resources to ensure credible development of performance measures and strategies for evaluations.

- **Objective 2**: Develop and implement an operating model of the Torres Strait ecosystem that will synthesise multi-disciplinary information and provide a foundation for developing multiple-use management strategy evaluation.
Objective 3: Evaluate the ecological effects of trawling, its interactions, and the performance of different options for managing the fishery to achieve environmental objectives.

Objective 4: Where appropriate, conduct preliminary modelling and evaluation of other ecosystem issues, e.g. sea-grass dieback consequences, as appropriate to the state of ecosystem understanding and modelling.

1.3. Consultation

The consultation to date has been mainly directed towards the development of the trawl model and the framework software. Various visits of the CRC Task Staff have been made to Torres Strait to liaise with stakeholders, particularly the people of Masig, Torres Strait Regional Authority and AFMA Staff. The main objectives of these discussions were to explain the methods and goals of MSE and discuss its specification for the Torres Strait prawn trawl fishery. These opportunities were also used to meet people working in the prawn trawling industry.

A presentation on the progress of the Task was given to managers, industry representatives and science providers during the alternative management strategies workshop in Cairns. A recent meeting with AFMA Staff was convened to inform on the project status and to elicit feedback on the proposed work to be undertaken and completed around mid-2007.

1.4. MSE Specification

The management strategy evaluation of the effects of trawling (EoT-MSE) for the Torres Strait is based on a specification by stakeholders. In collaboration with the Torres Strait Regional Authority (TSRA), the Australian Fisheries Management Authority (TS-AFMA) Staff and in discussion with the people of Masig, the following (fixed-rule) scenarios were proposed:

- Reefs/islands buffer zones, closed to prawn trawling.
- Time closure: moon phase closure, 4 days either side of full moon.
- Three different Total Allowable Effort (TAE) quota: currently 16,750 (13,396*1.25) boat days allocated (+3 TS licences). TAE down from 9000 to 5000 boat days in 3 linear annual steps of 1,300 days.
- Demo option: close an area of 10 km around Masig for the recovery of benthic biota.

The performance measures that were asked to be reported (against status quo, 2003...
standard):

- Benthic biomass changes
- Benthic impact status for 3x3 vulnerability classes
- Effort change maps.
- Total expected catch/CPUE (total prawn biomass) change.
- Percentage reallocated effort

This is expected to be the first of a number of iterations where the scenarios can be adapted on request. To be able to address these scenarios, the data were collected from a range of sources. Natural resource management often needs to incorporate a range of issues pertaining to system response in respect to human usage and management efficacy.

Management Scenario Evaluation and its adaptive counterpart Management Strategy Evaluation (MSE) are modelling approaches that support the management of such resources by testing alternative options available and reporting the results in terms of a set of specified performance measures. In the case of management scenario evaluation, a set of concrete management actions is evaluated against specified performance measures.

In an MSE one or more sets of alternative management rules are tested, possibly within an adaptive management scheme. Rules are essentially recipes for what management actions should be taken given a set of assessment results. Chapter 2 describes the MSE process in more detail. For the Torres Strait trawl MSE project, the operating model represents the ‘best available knowledge’ against which we aim to test different management scenarios. This report describes the trawl model in more detail in Chapter 3 and discusses the results of the MSE task activities in Chapter 4.
2. MANAGEMENT STRATEGY EVALUATION

2.1. Introduction

Management strategy evaluation (MSE) evaluates alternative management strategies by predicting the values of performance measures for each strategy. The performance measures are based on operational management objectives. A management strategy includes a monitoring program, scientific assessment methods (for calculating indicators), decision rules and mechanisms for implementing management decisions.

An MSE operating model is a simulation model of the managed system, which includes the ecosystem and human activity in the ecosystem (Figure 2.1). The model predicts the state of the system under the operation of a given management strategy. The predicted state is used for two purposes—strategy evaluation and simulation of management decisions. For strategy evaluation, the actual predicted state is used for calculation of performance measures. In contrast, simulated management decisions use information equivalent to that available to a real-world manager: they use data sampled from the system by a simulated monitoring program. The monitoring program is simulated by the observation model.

The operating model also includes a management model, which simulates the implementation of management decisions, including imperfection in management operations (e.g. delays in implementing management decisions or non-compliance with management decisions).

An MSE system may consider multiple management objectives or include multiple decision rules, and may include multiple indicators or performance measures. Evaluation of a given management strategy may involve a number of different operating models that represent different hypotheses about how the system operates.
Figure 2.1 A management strategy evaluation system supports the planning of adaptive management. The role of the system is to predict the effects of management actions on performance measures. Note that there may be multiple submodels for each step in the cycle to support different decision rules and performance measures.

In an MSE system the indicators are typically generated within the response models, which represent our best knowledge of how the real world reacts to management actions. Performance measures are assessments of the effects of management actions as simulated by the response models. The performance measures form the basis of the adaptive learning and are used to base the next set of management action on. For example, if our performance measure was simply the total benthic biomass, then the best model would be the one that best predicts the effects of different trawl management strategies on benthic biomass. However, if we added a performance measure for community composition (e.g. a diversity index), then we would need a (probably more complex) model that also predicted community composition.

For each submodel there is a choice of model type (e.g. simulation model, statistical model, analytical model) and design, and a given MSE system may include a mix of model types.

The basic MSE framework (Figure 2.1) can be modified to suit different situations. For example, in Figure 2.1 the management strategy is non-adaptive because the decision rule does not change (as in a constant reference point). For an adaptive management strategy, an
additional decision rule would modify the basic decision rule within each management cycle or some longer interval (e.g. a reference point might be reviewed every five years). Alternatively, the framework may be simplified. For example, if the management strategy was to specify a total number of fishing licenses, and not regularly review that number of licenses, then there would be no decision rule and therefore no management cycle. The operating model would provide data for strategy evaluation only, but not for management decisions.

2.2. MSE Implementation

The main components of a resource management evaluation system are presented in Figure 2.1. A core feature of an MSE simulation system is the need to simulate aspects of the process of resource management itself. However, depending on the actual management system in use, it may or may not simulate an adaptive approach. In an adaptive management approach the resource management actions adapt in response to (simulated) monitoring of management performance using a (pre-agreed) set of management rules.

The management decisions result in resource management activities that entail various tools to manage the use of a natural resource. In the case of fisheries management, these activities often include various levels of spatial and/or temporal restrictions of fisheries access, individual or fleet-wide. The management of a resource is often (but not exclusively e.g. wild stock enhancement programs) through the management of human exploitation of the resource.

The response models represent the response of the ‘real world’ to the exploitation (and its management) based on our best knowledge. They may include models of the ecosystem (biology, environment and their interactions with human use), economy, stock etc.. The most challenging and time-consuming part of the MSE is often the development of an adequate representation of the system’s responses to exploitation. The descriptions may be at different levels of detail, from broad-brush spatio-temporal fisheries effort allocation based on statistical interpretation of historical data down to simulating detailed decision making processes on board of individual vessels.

The observation module simulates the way we probe or monitor the ‘real world’, typically through field programs. Such programs may be used to extract information about the status of parts of the system under investigation but also to assess efficacy of management. As these programs are costly to implement and maintain, it is important to properly design the spatial and temporal characteristics of their field programs and carefully choose the indicators they collect. For realistic simulation of decision rules, we may want to constrain the observations from the operating models to those that can be feasibly collected through a monitoring program. On top
of monitoring the status of the resource, additional observations may need to be collected to assess and report the efficacy of the management itself.

The **assessment and reporting** phase contains (often statistical) methods for processing the collected indicators into management performance measures. This may be as simple as drawing some summary statistics out of the measured indicators or a complex as a full-blown fisheries assessment. The implementation of this function often needs a range of data extraction modules (e.g. SQL-queries, multi-dimensional data cubes), access to statistical capabilities (e.g. Splus, Matlab) and report writing modules (e.g. Crystal reports). The next paragraph discusses some of the software requirements.

The **learning** model looks at the discrepancies between the expected results of management actions and the actual results after they have been applied to the response model. There are two modes of learning: the passive learning aims to update the importance of each management action after an iteration through the adaptive management loop. The active learning mode may schedule management activities purely aimed at gaining information about the response system or the efficacy of management actions.

Appendix D describes the software implementation of the Torres Strait MSE software in more detail.
3. EFFECTS OF TRAWLING MODEL

3.1. Introduction

This chapter discusses the model we developed to estimate the primary effects of trawling for the Torres Straits on specific benthic (ground-dwelling) biota (flora and fauna) and the data that were used to constrain the model. In this model, space and time are explicitly represented.

The core activity of the effects of trawl MSE project is evaluate a range of options for the management of trawling in the Torres Strait with respect to the effects on benthic biota.

3.2. Modelling the effects of trawling on benthic biota

The effects of trawling (EoT) model estimates the primary effect of repeated trawling on the biomass of bottom-dwelling (benthic) organisms (primary effects), ignoring longer-term consequences of that removal to the ecosystem (secondary effects). The model incorporates four main processes (see Figure 3.1): (i) the spatio-temporal distribution of trawl effort, (ii) the estimation of effects of (repeated) trawling on the depletion of the biomass of a range of benthic species, (iii) the recovery of each of the species that are under consideration and (iv) the estimation of the pre-trawl distribution of benthic species.

At the core of the EoT model is the biomass depletion and recovery model:

\[
\text{Change in biomass} = \text{growth of biomass} - \text{biomass depletion}. \tag{3.0}
\]

Figure 3.1 gives an overview of the factors that are being taken into account.
The Effects of Trawling model is based on the biomass depletion and recovery of species that are affected by trawling. Its parameters are derived from the four areas indicated in the diagram.

In the centre of Figure 3.1 the biomass equation of a species (or other taxonomic unit) that is affected by trawling, is written as the biomass at a time $t + \delta$, given the biomass $B_i$ at time $t$, some recovery rate $r$, the pre-trawl biomass $B_0$, the trawl effort $E$ and some large-scale depletion rate $\lambda$:

$$B_{t+\delta} = B_t + (rB_t(1 - B_t / B_0) - B_t.E \lambda)\delta$$  \hspace{1cm} (3-1)

In this representation, the pre-trawling biomass $B_0$ is also assumed to be at the carrying capacity of that species or taxonomic unit. Instead of solving Eq. (3-1) by stepping through time, an analytic expression is found that describes the biomass after a relatively large time step (e.g. after a week or a year). See section: Analytical solution.

The modelling process is restricted to a geographic region of interest, such as the Torres Strait, which is divided into rectangular cells (Figure 3.2). The EoT model in Eq. (3-1) is applied to each of those cells. In the next sections, each of the factors included in the EoT model is discussed.
3.3. **Spatial distribution of the benthic biomass**

To allow us to estimate the effects of trawling on benthic flora and fauna (biota), one of the core pieces of information we need is the spatial distribution of the benthic biota in the region, for instance on a grid as shown in Figure 3.2. The Torres Strait mapping project (Torres Strait CRC Task T2.1) has surveyed the Torres Strait and estimated the biomass at a range of locations; a summary map of their findings is shown in Figure 3.3.
Figure 3.3 One of the results of surveys in the Torres Strait is a summary map of benthic flora and fauna at a range of sample sites. This map shows the results using the benthic sled sampling device (Courtesy Pitcher).

The summary map in Figure 3.3 shows the benthic biota at discrete sample locations at the time of the survey (between January 2004 and April 2005). It is of importance to realise that the distribution of biota on these maps also includes the effects of decades of trawling in the region. There are two more steps of processing that need to occur on this data before it is suitable to be used in the trawl model: estimating the pre-trawl biomasses at the survey sites (using a regression technique) and estimating the biomasses on a grid such as shown in Figure 3.2 (using a prediction from the regression model). The last step means that we need to interpolate the benthic biota between the survey sites as given in Figure 3.3.

3.4. Estimating pre-trawl benthic biomass

The Torres Strait mapping survey visited around 400 sites according to a stratified design. At each site a benthic sled was deployed and at about 150 of these sites a prawn trawl net was also deployed. All benthic fauna caught in these devices were weighed and identified to an operational taxonomic unit (OTU). Mostly, an OTU equates to a species. As well as biotic data, physical data such as sediment content and depth were collected at each site. Other physical
data available were also gathered for each site; these were bathymetry data (slope, aspect), water column physical and chemical attributes (temperature, salinity, oxygen, silicate, phosphate, nitrate) from the CSIRO Atlas of Regional Seas database, ocean colour (chlorophyll-a, K490 attenuation, benthic irradiance) from the SeaWifs satellite data, modelled benthic stress and time-averaged trawl effort.

The biotic data were then modelled in terms of the physical data (the predictors) and the other variables (the co-variables), such as time of day, time of year, phase of moon. Each OTU was fit to a statistical model that was a product of two generalized linear models: the first fit the expected probability of presence and the second fit the expected biomass given presence. The product of these two is the expected biomass.

A further correction was made to provide the optimal density of the OTU, meaning the density one would measure under the most favourable co-variable settings. For instance, some species might have a higher catch rate at night, but were actually surveyed at some sites in day time and other sites at night. The optimal density would then be predicted at the night-time setting for this species. The statistical analysis is described in the Appendix B, Single Species Biophysical Models and Prediction and more fully in Pitcher et al. (2007b).

Using these models one can predict the optimal density of each OTU at any point for which one has predictor values. In particular this means one can draw a map of the optimal density on a regular grid. This predicted density is actually relative to the unknown catch rates, which is assumed to be constant. An example is shown in Figure 3.4 for the soft coral genus Dendronephthya spp.

These models can also be used to estimate the pristine density over the area of the fishery. This is done by predicting using the optimal co-variable settings as before and setting the trawl effort predictor to zero. The result is then the predicted density taking into account the physical covariates but in the absence of trawling, i.e. the pristine pre-fishery density. The estimation of pristine density at a trawled site is based on the observed density at sites away from trawled areas having similar physical attributes.
Figure 3.4 Predicted optimal density for a soft coral genus (*Dendronephthya* spp). The colour scale is given in Figure 3.5. For this taxon bottom slope, trawl effort and K490 attenuation influenced the probability of presence, whereas mud and chlorophyll-a influenced the magnitude of the density.

Figure 3.5 Colour scale for Figure 3.4 and Figure 3.6. Density is indicated by hue and precision by saturation. Washed out colours tending to grey imply higher uncertainty in the estimates.
3.5. **Estimating benthic biomass on the regular grid.**

Just as we can estimate the predicted pristine biomass on the pixels of the map in Figure 3.4, so we can provide it on the grid of the trawl model in Figure 3.2. The trawl model propagates the density forward from its pristine value to the value at the end of the simulation after trawling has been applied. The resulting mean density is the average of the simulated density over all grid cells. An example is shown in Figure 3.6 for sea cucumbers (Holothuroidea) restricted to the trawling grounds. The pristine map on the left is the overlay of all holothuroid pristine density predictions summed at each grid cell. The right-hand side is the result of propagating the pristine density forward in time under the status quo scenario. The mean density at any time is found by averaging over all the grid cells the density at that time.

![Figure 3.6. Density (g/Ha) of Holothuroidea (left) in the pristine state and (right) in 2020 under the StatusQuo scenario. The colour scale is given in Figure 3.5.](image)

3.6. **Spatial and Temporal Distribution of Trawl Effort**

The second important piece of information needed to estimate the effects of trawling is the...
estimation of the trawl effort to which benthic biota are exposed. There are a couple of questions to be addressed:

1. scaling the trawl effort: given large-scale trawl effort, given by the trawl fisheries logbook data, how does that translate to effort to which a benthic organism is exposed?
2. re-allocating trawl effort: if we close a certain area to trawl, what do we expect to happen to the displaced trawl effort?

To answer the first question requires us not only to derive a consistent measure of effort on our prediction grid, but also to include patterns of trawling within the grid. Section 3.7 discusses two issues: how do we derive a consistent estimate of trawl effort in each of the grid cells shown in Figure 3.2 and how does trawl effort translate to an estimation of the number of net strikes a benthic organism can expect, given a large scale amount of trawling.

The answer to the second question depends on how we expect fishermen to react to spatial or temporal closures of fishing grounds.

### 3.7. Process large-scale trawling effort data into organism-scale impacts

#### 3.7.1. Converting large scale effort to small scale effort

Depending on the available data, the approach to converting trawling effort into effects on biota would be different. In the case of the Torres Strait trawl, trawl effort data is available through the Australian Fisheries Management Authority (AFMA) fisheries logbook data and via the satellite vessel monitoring system (VMS). The AFMA logbook data is mainly given as trawling effort (often aggregated by day) on a pre-specified 30 minute (about 1700 km$^2$ grid) or 6 minute grid (about 125 km$^2$ grid). The VMS data consist of vessel positions with GPS resolution. However, as the vessel positions are only interrogated (polled) at relatively long intervals (typically in the order of six hours) its operational resolution is much less.

First, we need to decide on a scale of spatial and temporal aggregation at which to operate the model. With management scenario modelling we are mostly interested in the impacts on moderately vulnerable species over time scales of decades. We believe, therefore, that it is adequate to use effort aggregated within each year. However, as the moon closure management scenario acts on a weekly scale, we need to run the trawl effects model also on a weekly scale and consequently we chose our effort resolution to be on a weekly scale. That is, we assume that for each grid the total effort over one week is uniformly distributed. Effort data in fact exist
at daily temporal resolution and it is used by aggregating it up to weekly intervals. However, the weekly time step has been ignored in the seasonality of organism growths (in the recovery model) for lack of information.

On the spatial scale, the effort was defined at a resolution of 1 nautical mile. For 2004 this was achieved by analysing the VMS data. Each VMS event represents a location of effort and the duration of the effort, being the time interval to the next event. The events are binned into the 1-nautical mile grid and durations are summed within each bin. Events corresponding to non-trawling are filtered out. The resulting data set represents the annual effort at 1-nautical mile resolution in units of hours for 2004.

For other years only the logbook records were available. These consist of daily effort in 6-nautical mile grid cells. These data were converted to 1-nautical mile resolution by distributing the total effort in a 6-nautical mile grid cell among all the sub cells in proportion to the pattern within that cell in 2004.

### 3.7.2. Trawling patterns: converting grid-scale effort to organism scale

The question we need to address is what direct pressure is put upon a benthic organism, given an amount of trawl effort (often in boat days) in a prediction grid. Elements that contribute to answering this question are: effective width (on the ground) of nets/chains used in trawling, the speed of the vessels during trawl, number of trawling hours in a boat day, the trawl pattern, and the amount of trawl effort. Some of the parameters used in the modelling are given in Table 3.1.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Average value</th>
<th>Unit</th>
</tr>
</thead>
<tbody>
<tr>
<td>Trawl speed</td>
<td>3</td>
<td>kts</td>
</tr>
<tr>
<td>Effective net width</td>
<td>28</td>
<td>m</td>
</tr>
<tr>
<td>Effort in a boat day</td>
<td>10.7</td>
<td>hrs</td>
</tr>
<tr>
<td>VMS polling frequency</td>
<td>variable</td>
<td>hrs</td>
</tr>
</tbody>
</table>

The more interesting question is how trawl patterns may affect benthic organisms. Trawling patterns fall into three classes: aggregated, uniform and random, examples of the three classes are shown in Figure 3.7.
Figure 3.7 The three graphs in left column show an example of simulated trawl patterns on a 133 by 133 pixel grid. They represent the results of uniform, random and aggregated trawling respectively. The right-hand column shows the histograms of the probability (Y axis) of a pixel being struck a number of times (X axis) by a trawl net.

Uniform trawling corresponds to patterns that avoid previously trawled ground; an extreme case is trawling in non-overlapping swathes, like a lawn mower. Given an amount of effort, the histogram corresponding to the uniform trawling pattern shows that an organism in this case has a chance of a bit less than 50% of not being affected and bit more of being struck by trawling once.

A higher degree of aggregation is random trawling. Random trawling is simulated by trawling in a straight line in a random direction from a random starting point. Given the same effort as in the uniform trawling, the histogram of the random trawling shows that the chance of not being passed over by a net at all has increased to about 60%, but that an organism now also has a chance of being struck more than once.

Aggregated (or clustered) trawling corresponds to particular patches being targeted by multiple tows from one or more boats. The histogram shows a probability of around 75% of not being hit, given the same trawl effort as in the previous two trawl patterns. However, now there...
are increased probabilities that an organism is passed over a number of times.

Instead of having these discrete classes of trawl aggregation, we have modelled these classes by a family of statistical distributions known as the negative-binomial distribution. The level of aggregation is characterised by a single aggregation parameter $\beta$. This aggregation parameter allows us to model the level of trawl aggregation on a continuous scale instead of the three discrete classes. For specific values of $\beta$, we can identify the three discrete cases we discussed previously. The three cases are described by the negative binomial ($\beta > 0$), binomial ($\beta < 0$), and Poisson ($\beta = 0$) distributions, respectively. The nature of these distributions can also be appreciated from the histograms in Figure 3.7, corresponding to the three trawl patterns. The uniform case is clearly binomial (with $N = 1$ and $p$ slightly greater than $\frac{1}{2}$), the random trawling histogram follows almost exactly Poisson, and the aggregated case (right) is like the random case but with a larger variance, and it is reasonably well approximated by a negative binomial with $\beta = 1.24$. We see that the aggregated case has considerably more zero counts and somewhat more high counts than the random case. The parameter $\beta$ provides the bridge between depletion on the small scale and depletion on the large scale, as discussed in the next section.

Having established the model to estimate the probability of the number of strikes, given a level of trawl aggregation $\beta$ and amount of trawl effort in a grid cell, the next step is to estimate values for $\beta$. We have two approaches to estimating $\beta$: the first uses the vessel positioning data from the Torres Strait; the second uses simulation from a trawling model.

The first approach to estimating the aggregation parameter is by using regularly polled VMS data and exact vessel tracks from GPS plotter. Hall et al. (1999) coupled these two data sets in order to simulate the fine-scale pattern of trawling over an entire grid cell. From the resulting distribution of effort we can obtain the most appropriate aggregation model. Hall et al found values of $\beta$ roughly in the range 2–5 (see their Table 3 for values of $P$, which is their notation for $\beta$).

The second approach is through simulation of the trawling process. We can model a single trawl as a rectangular sweep across the seabed, whose position and orientation can be varied. We can simulate the reported number of boat days of effort by 'trawling' until a chosen level of effort (swept area) is reached. We can introduce aggregation by making multiple sweeps over the same patch, attributable to repeated tows by the same vessel or group of vessels. We can also control the amount of overlap in the multiple sweeps by specifying the size of the turning circle and the relative bearing of the return tow. We generated the trawling patterns in Figure 3.7 using this approach.
3.8. Effort allocation

In the historical period of the MSE simulations, the actual historical effort is used to drive the model. In the projection part of the model, where the various scenarios come into effect, the spatial and temporal distribution of effort has to be predicted somehow.

We do not attempt to model the detailed processes leading to the spatio-temporal patterns of effort over a season. Instead we use the entire season from a particular historical year $y^*$ as a prototype for the pattern in the projection year $y$. This prototype is adjusted depending on the management actions in place for year $y$. For each projection year a prototype year is chosen at random from a range of historical years considered as representative of future patterns.

The effort is projected first by allocating the total effort $E_{y}^{alloc}$ for year $y$ thus

$$E_{y}^{alloc} = E_{y}^{cap} \varepsilon_{y}. \quad (3-2)$$

Here $E_{y}^{cap}$ is the effort cap set by management and $\varepsilon_{y}$ is a lognormal variate with mean 1 and coefficient of variation $\sigma$. This random term represents the implementation error, that is, the ratio of actual effort to desired effort.

Next, the effort is allocated to each spatial cell $g$ in proportion to the spatial effort $E_{y}^{*g}$ for the representative year $y^*$:

$$E_{y}^{alloc} = E_{y}^{alloc} \frac{S_{g} E_{y}^{*g}}{\sum_{g^{\prime}} S_{g^{\prime}} E_{y}^{*g^{\prime}}}. \quad (3-3)$$

Here $S_{g}$ is the spatial closure term taking value 1 if the cell is open and 0 if the cell is closed.

An analogous procedure is applied for temporal closures in those scenarios where effort is resolved on a within-season weekly time scale. For week $w$ the allocated effort is proportional to the weekly effort $E_{y}^{*w}$ for the representative year $y^*$:

$$E_{y}^{alloc} = E_{y}^{cap} \varepsilon_{y} S_{w} E_{y}^{*w} \frac{\sum_{w^{\prime}} S_{w^{\prime}} E_{y}^{*w^{\prime}}}{\sum_{w^{\prime}} S_{w^{\prime}} E_{y}^{*w^{\prime}}}. \quad (3-4)$$

Here $S_{w}$ is the temporal closure term: it is the proportion of the week that was open to trawling. So, outside the season it takes value 0, inside it takes value 1, but in a week close to either the beginning or end of the season or full moon (if a full moon closure is in operation) it takes a value between 0 and 1.

The spatial allocation is then performed similarly to the annual time step case, except that the totals are now weekly totals. The allocated effort to each grid $g$ for each week $w$ is proportional to the spatial weekly effort $E_{y}^{*wg}$ for the representative year $y^*$:
This formulation has provision for time varying spatial closures, $S_{yw}$, although these were not used in our simulations (i.e. $S_{yw} = 1$).

### 3.9. Depletion

We model the impacts of trawling at the scale of an effort grid. Since most of the fishery’s effort data is recorded on either the 30’ grid or the 6’ grid, we may choose either one of these grids as our modelling scale. To describe depletion at such a scale we use a parameter $\lambda$, called the large-scale depletion rate. This parameter is actually derived from the small-scale aspects of trawling, namely, the depletion rate per tow $d$ and the pattern of trawling within the grid. Knowledge of $\lambda$ therefore depends on knowledge of these small-scale aspects. The small-scale depletion model is illustrated in Figure 3.8; it is simply removal of a proportion $d$ of the existing biomass by one pass of the trawl gear. Estimates of the depletion rate per tow are available from the repeat-trawl experiment in Poiner et al (1998). Trawling patterns can be obtained from VMS data augmented by simulation (Hall et al, 1999).

Linking the depletion to the trawl pattern results, the parameter $\beta$ provides the bridge between depletion on the small scale and depletion on the large scale. Remarkably, the two depletion rates $\lambda$ and $d$ are identical for random trawling. Aggregated trawling tends to reduce the large-scale depletion rate, whereas uniform trawling tends to increase it, as Figure 3.9 shows. This is easy to understand since, in aggregated trawling, subsequent trawls over an area remove less than the first trawl, whereas, in uniform trawling, this effect of ‘diminishing returns’ is avoided because of the tendency to trawl previously non-trawled areas.

It may seem counter-intuitive that $\lambda$ should not equal $d$ for the uniform case, since it appears that a proportion $dE$ of the benthos is being removed per unit time. However, this is only true initially. As time proceeds and the total benthos becomes depleted, the proportion of remaining benthos removed per unit time steadily increases, even though the absolute amount removed per unit time remains the same. Therefore $\lambda$, which is defined as the rate of change of biomass per unit effort rate per unit biomass, should be greater than $d$.

To consider the implications of Figure 3.9, suppose we have a sessile epi-benthic species that is removed at the rate of 30% per tow. If trawling is aggregated, with $\beta = 1.24$, then the species is removed on the large scale at rate 26% per unit effort per year (see figure). For low depletion rates (10% in the figure), aggregation has only a small effect, since $\lambda$ and $d$ are almost the same.
The effect of trawling depends on the product of effort and depletion rate per unit effort. Therefore we are completely at liberty to define convenient units for effort, provided we use related units for depletion. For our purposes, the most convenient definition of effort is this:

\[
\text{effort} = \frac{\text{swept area}}{\text{grid area}}
\]

Here, swept area is the area swept out by the net irrespective of whether or not it traverses the same ground. This defines effort relative to the area it is exercised in.

Figure 3.8 A simple model of depletion. The depletion rate per tow is 0.2

Figure 3.9 Large-scale vs small-scale depletion for the uniform, random and aggregated
fishing patterns. The $\beta$ values are –0.51, 0.00 and 1.24, respectively.

The relationship between the large-scale depletion rate, $\lambda$, and the small-scale depletion rate, $d$, is

$$\lambda = \begin{cases} \log(1 + \beta d) / \beta & \beta \neq 0 \\ d & \beta = 0 \end{cases}$$

(3-6)

3.10. Recovery

3.10.1. The small-scale recovery model

At the scale of the tow, i.e. on the order of metres, we assume that recovery follows a sigmoid growth curve. An example is shown in Figure 3.10: if the biomass is depleted to, say, 5% capacity, then, in the absence of any further depletion, the biomass would take about 30 years to reach 50% capacity and a further 30 years to reach 95% capacity. The shape of the curve is governed by the small-scale recovery rate $r_s$, which in this case is 0.1 year$^{-1}$. The curve has maximum slope equal to $\frac{1}{4} r_s$ at the time when the biomass reaches 50% capacity. We do not distinguish between recovery through population growth (i.e. increase in number of individuals) and recovery through individual growth (i.e. increase in size). We assume that both forms of recovery can be subsumed in a single sigmoidal curve parameterised by $r_s$.

It may be helpful to interpret the recovery rate in terms of time scales. We can regard $2/r_s$ as a nominal recovery time: it is the time the biomass would take to recover completely from 50% capacity if it kept on growing linearly at its maximum rate along the straight line in Figure 3.10. Alternatively, $3/r_s$ is the actual time taken for the biomass to recover from 50% to 95% capacity.
3.10.2. The large-scale recovery model

Consider now the recovery of the biomass over an entire grid. If the relative biomass in every small-scale patch within the grid were the same, then the total biomass would follow the sigmoid recovery curve with parameter $r_s$. That is the large-scale recovery curve would be the same as the small-scale recovery curve. However, the relative biomass will more likely be different from one patch to the next, owing to differing amounts of depletion over each patch. In this case the total biomass will again approximately follow a sigmoid curve, but now it will have a smaller recovery rate, $r$.

The fact that $r$ is less than $r_s$ arises from the concavity of biomass as a function of biomass at an earlier time. This fact is most easily understood for the case where the relative total biomass is 50%. The rate of change of relative total biomass is the average of the rates of change of relative biomass over all patches. In patches where the biomass is 50% the rate of change of relative biomass is at the maximum, $\frac{1}{4} r_s$. But, for patches with relative biomass either less than or greater than 50%, the rate of change of relative biomass is less than the maxi-
mum. Therefore the rate of change of relative total biomass is less than $\frac{1}{4} r_s$, and so $r$ must be less than $r_s$.

The amount by which $r$ is less than $r_s$ depends on the variance of relative biomass among small-scale patches. Indeed, if $b$ is the mean relative biomass (i.e. the large-scale relative biomass) and the variance is $\phi b(1 - b)$, then $r = r_s(1 - \phi)$. The quantity $\phi$, which lies between 0 and 1, depends on the details of the small-scale history of trawling within the grid. By approximating $\phi$ with a constant (Ellis and Pantus, 2001), we can show that the large-scale recovery $r$ is approximately related to the small-scale recovery $r_s$ as follows

$$r = r_s(1 - \frac{1}{2}d)\frac{\lambda}{d}$$  \hspace{1cm} (3-7)

It is this recovery rate that is used in the main dynamical equation (3-1). Note that, in the absence of depletion, $r = r_s$, but when depletion is present the recovery is reduced by a factor $1 - \frac{1}{2}d$. The effect of aggregation is to further reduce the recovery by a factor $\frac{\lambda}{d}$. See Part Two of Ellis and Pantus (2001) for further details on the derivation of this formula.

### 3.11. Data collection

The following data were collected and processed:

- Fine-scale spatio-temporal distribution of fishing effort from the 2004 Vessel Monitoring System (VMS) data
- Spatial distribution of benthic biota
- Benthic biota depletion and recovery data
- Geographic map layers of TS

VMS high-definition spatio-temporal effort data for 2004 was processed to fit the needs of the Effects of Trawling (EoT) model. All data sets were checked and adjusted to suit the project’s needs. All data has been entered in the appropriately structured databases. Restrictions apply to which level of detail this data may be used in public.

### 3.12. References


4. RESULTS OF MANAGEMENT SCENARIOS EVALUATION

4.1. Introduction

This chapter presents and discusses the results of the Torres Straits effects of trawl management scenario evaluations.

The effects of trawling on the benthic ecosystem in the Torres Strait are of concern to conservationists, the local community and the fishing industry alike. The Torres Strait Regional Authority (TSRA) has a mandate to protect the marine ecosystem in the region; and the Australian Fisheries Management Authority (AFMA) has a responsibility to ensure that fishing is operated in an ecologically sustainable manner. In June 2004, representatives from TSRA, AFMA and the people of Masig drew up a set of management scenarios and performance measures for evaluation, as specified in Chapter 1. The results of these management scenarios are presented in this chapter.

4.2. Management scenarios

The Torres Strait prawn fishery is managed using input controls. The basic kinds of management scenarios are restrictions on where, when and how much trawling can occur. To assess the effect of management scenarios one first needs a status quo option, which is continuation of management settings currently in place.

The stakeholder group proposed four alternative options for evaluation:

- *reef buffer*: establish buffer zones around reefs and islands that are closed to trawling
- *moon closure*: close the whole fishery 4 days either side of full moon
- *effort reduction*: reduce total allowable effort (TAE) from 9000 to 5000 boat days in 3 linear annual steps of 1,333 days.
- *Masig closure*: close an area 10 km around Masig for the recovery of benthic biota.

The *reef buffer* option had three sub-options, in which the buffer width was fixed at 0.5km, 1km and 2km. The *moon closure* option had two sub-options: one in which the effort during the temporal closure was lost from the fishery, and a second in which the effort was displaced to the open times by drawing on latent effort. Similarly, for the *Masig closure* one option allowed displacement of effort out of the closed area while a second option assumed the effort was removed from the fishery. In options where total effort was unaffected, the TAE of the *status*
This management scenario evaluation is a special case of management strategy evaluation (MSE). In MSE there is usually an assessment step (typically a stock assessment) that feeds back into the management process. The management decisions thus adapt to the current state of the system, as estimated from the assessment. In management scenario evaluation the management strategies are fixed at the outset and do not adapt to changes in the target stock or the ecosystem.

4.3. Model

The dynamic biomass model is a set of Schaefer-like models operating independently in each spatial cell:

\[
\frac{dB_{sx}}{dt} = r_s B_{sx}(t)(1 - B_{sx}(t)/K_{sx}) - d_s E_x(t)B_{sx}(t)
\]

where \(B_{sx}(t)\) is the biomass at time \(t\) of benthic species (or taxon) \(s\) in cell \(x\), \(K_{sx}\) is the carrying capacity of species \(s\) in cell \(x\), \(E_x(t)\) is the effort rate at time \(t\) in units of swept area per unit time, \(r_s\) is the recovery rate of species \(s\), and \(d_s\) is the depletion rate per tow of species \(s\). The equation simplifies to

\[
\frac{db_{sx}}{dt} = r_s b_{sx}(t)(1 - b_{sx}(t)) - d_s E_x(t)b_{sx}(t)
\]

where \(b_{sx}(t) = B_{sx}(t)/K_{sx}\) is the relative biomass. This has the practical consequence that the biomass distribution can be split into two components, one, \(b_{sx}\), depending only on the vulnerability pair \((r_s, d_s)\) and the other, \(K_{sx}\), depending only on survey data. Each component can then be computed independently and combined later. To provide an initial condition for \(b_{sx}\), we assume the pre-fishery biomass (at time \(t_0\)) was at the carrying capacity, i.e. \(b_{sx}(t_0) = 1\).

To model the effect of moon phase on target catch, we fit a periodic series to fishery catch-effort data over an extended period. Details are provided in the Appendix E.

4.4. Data

Data for the model was drawn from several sources: the fleet data from AFMA fishery logbooks and satellite vessel monitoring systems; the benthic depletion rates from the repeated trawl experiments (Poiner et al, 1998); the benthic biota recovery rates used were developed for the Northern Prawn Fisheries Surrogates project (Hill et al. 2002); and benthic distribution from large-scale benthic surveys (Pitcher et al. 2007b).

4.4.1. Fleet data

Logbook data have been collected in the Torres Strait fishery since 1980 until 2002. In the later
years trawl effort was reported daily by each vessel at the resolution of 6-minute cells. In 2003 a satellite vessel monitoring system (VMS) was introduced which allowed vessels to be located with high accuracy at any time of day. It is therefore possible to capture the pattern of trawling at very fine spatial and temporal resolution.

For this study we aggregated the trawl effort to 1-minute spatial cells (Haywood, REF) and to annual or weekly temporal bins, depending on the management scenario. For historical effort available at 6-minute resolution, we subdivided the effort into 1-minute cells proportionally to the average VMS pattern within each 6-minute cell.

### 4.4.2. Depletion and recovery rates

Pitcher et al (2004) obtained parameters describing the depletion and recovery dynamics of a set of benthic taxa in the Far Northern section of the Great Barrier Reef. However, the type of dynamics allowed by their parameterization was more flexible than a simple Schaefer model. We therefore chose the parameter pair \((r_s, d_i)\) to minimize the discrepancy of the dynamics predicted by the Schaefer model from the dynamics predicted by the Pitcher et al (2004) model. The method is described fully in Pitcher et al (2007a).

In the absence of parameters at fine taxonomic resolution in Pitcher et al (2004) we used the coarser taxonomic groupings reported by Poiner et al (1998) with recovery rates obtained using the categorical method of Hill et al (2002). Table 4.2 summarizes the benthic groups, their \((r_s, d_i)\) values and the sources.

### 4.4.3. Pristine biomass

A large-scale benthic survey was carried out over the Torres Strait region (Pitcher et al, 2007b). A total of 400 sites were visited according to a stratified design. At each site a benthic sled was deployed and at 150 of these sites a prawn trawl net was also deployed. All benthic fauna caught in these devices were weighed and identified to an operational taxonomic unit (OTU). As well as biotic data, physical data such as sediment content and depth were collected at each site. Other physical data available were also gathered for each site; these were bathymetry data (slope, aspect), water column physical and chemical attributes (temperature, salinity, oxygen, silicate, phosphate, nitrate) from the CSIRO Atlas of Regional Seas database, ocean colour (chlorophyll-a, K490 attenuation, benthic irradiance) from the SeaWifs satellite data, modelled benthic stress and time-averaged trawl effort.

The biotic data were then modelled in terms of the physical data (the predictors) and the other nuisance variables such as time of day, time of year, phase of moon (the co-variables). Each OTU was fitted to a statistical model that was a product of two generalized linear models:
the first fitted the expected probability of presence and the second fitted the expected biomass given presence. The product of these two is the expected biomass. The statistical analysis is described fully in Pitcher et al (2007b).

Using these models one can predict the optimal density of each OTU at any point for which one has predictor values. In particular this means one can draw a map of the optimal density. Here ‘optimal’ means the density one would measure under the most favourable co-variable settings. For instance, some species might have higher catch rates at night, but were surveyed at some sites in daytime and other sites at night. The predicted density would then use the night-time setting for this species. This predicted density is actually relative to the unknown catch rates, which is assumed to be constant.

For the purposes of this report, these models can also be used to predict the pristine density over the area of the fishery. This is done by predicting using the optimal covariable settings as before and setting the trawl effort predictor to zero. The result is then the predicted density taking into account the physical covariates but in the absence of trawling, i.e. the pristine pre-fishery density. The prediction of pristine density at a trawled site is based on the observed density at sites away from trawled areas having similar physical attributes.

4.5. Results

A key output of an MSE is a decision support table. The decision support table brings out the trade-offs between various management scenarios in terms of performance measures. The performance measures that were agreed to be assessed were as follows:

- Percentage reallocated effort
- Total expected catch/CPUE (total prawn biomass) change
- Benthic biomass changes
- Benthic impact status for 9 vulnerability classes
- Effort change maps

With regards to the first performance measure, we have taken the straightforward approach of considering two types of scenario: scenarios where effort is displaced (reallocated) either temporally or spatially; and scenarios where that effort is instead removed. The percentage reallocated effort performance measure can then be simply assessed by comparison of the total effort performance measure between ‘displaced’ and ‘removed’ scenarios.

Table 3.1 shows the decision table for all 8 scenarios and the first three performance measures above. The decision support table is the highest level of results for an MSE and from there, drilling down to more detailed views helps to maintain a top-down approach to the various scenarios and their implications. Performance measures 4 and 5 are examples of more
The first 5 scenarios in the decision table are on the annual time step and the last 3 are on the weekly time step. ‘Status Quo, Annual’ is taken as the baseline scenario. In subsequent figures and tables, this is simply referred to as \textit{Status Quo}. It is against this scenario that all others should be compared. ‘Status Quo, Seasonal’ (subsequently referred to as \textit{Seasonal}) is the equivalent of \textit{Status Quo} implemented on the weekly time step, and so its performance measures should be similar. The larger standard deviations arise because extra weekly variability is included in the stochastic part of the effort model.

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Total effort ('000 hours)</th>
<th>Total catch (tonnes)</th>
<th>CPUE (kg hour\textsuperscript{-1})</th>
<th>Mean benthic biomass (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Status Quo, Annual</td>
<td>100 ± 12</td>
<td>1748 ± 246</td>
<td>17.48 ± 1.60</td>
<td>54.2 ± 0.8</td>
</tr>
<tr>
<td>Reef Buffers, Effort Displaced</td>
<td>100 ± 11</td>
<td>1742 ± 267</td>
<td>17.42 ± 1.61</td>
<td>54.4 ± 0.9</td>
</tr>
<tr>
<td>Masig Closure, Effort Displaced</td>
<td>100 ± 13</td>
<td>1785 ± 300</td>
<td>17.85 ± 1.68</td>
<td>54.8 ± 1.1</td>
</tr>
<tr>
<td>Masig Closure, Effort Removed</td>
<td>84 ± 10</td>
<td>1502 ± 234</td>
<td>17.88 ± 1.59</td>
<td>58.1 ± 1.0</td>
</tr>
<tr>
<td>Effort Reduction</td>
<td>55 ± 7</td>
<td>980 ± 160</td>
<td>17.62 ± 1.74</td>
<td>65.9 ± 1.1</td>
</tr>
<tr>
<td>Status Quo, Seasonal</td>
<td>99 ± 20</td>
<td>1728 ± 391</td>
<td>17.41 ± 2.39</td>
<td>54.6 ± 1.0</td>
</tr>
<tr>
<td>Full-moon Closure, Effort Displaced</td>
<td>99 ± 19</td>
<td>1774 ± 403</td>
<td>17.91 ± 2.45</td>
<td>54.7 ± 1.3</td>
</tr>
<tr>
<td>Full-moon Closure, Effort Removed</td>
<td>73 ± 14</td>
<td>1311 ± 295</td>
<td>17.97 ± 2.45</td>
<td>61.2 ± 0.9</td>
</tr>
</tbody>
</table>

Table 4.1. Performance measures ± standard deviations for each scenario. Effort and catch are the annual totals averaged, and CPUE the average annual CPUE, over the period 2006–2020. Biomass is the relative biomass in 2020 averaged over 11 vulnerability groups. The standard deviations are over 20 replicate simulations.

4.5.1. Percentage reallocated effort

The average amount of effort displaced by the Masig spatial closure is 16% (16,000 hours) and by the full-moon closure is 26% (26,000 hours). We did not consider the effort displaced by the reef closures here; however, earlier work showed that this was about 700 hours.

4.5.2. Total expected catch/CPUE (total prawn biomass) change

The full-moon closures should give a slightly higher CPUE than all other scenarios because effort is expended at times when CPUE is higher than average. This is indeed the case, the increase relative to the \textit{Seasonal} scenario being a little less than 3%. Figure 4.1 shows how this works for the year 2005 for the \textit{FullMoon/EffortDisplaced} scenario. The model operates on a weekly time step, and the points represent weekly totals for catch and effort. In 2005 the full moon fell in week 12 and the 8-day closure entirely spanned that week so there was zero effort.
and zero catch for that week (*black squares*). At other times the 8-day closure straddled two weeks (e.g. weeks 25 and 26) so the effort and catch was somewhat reduced in both weeks. During the rest of the month however the effort was elevated relative to the seasonal scenario. This was also the period where CPUE was higher and so the overall CPUE was higher. Since the total effort is the same for both scenarios the catch is higher for the closure scenario. For the *FullMoon/EffortRemoved* scenario, the effort, and hence catch, are lower than the *Seasonal* scenario. However, since the distribution of effort is the same as for the other full-moon scenario, the CPUE is also higher.

![Figure 4.1. Weekly catch (left) and effort (right) in 2005 averaged over 20 replicate simulations for the Seasonal and FullMoon/EffortDisplaced scenarios.](image)

It is possible that the benefits to CPUE of the full-moon scenarios are underestimated because the modelling does not take into account the dynamics of the target species. For, if fishing is absent in one week, we should expect the CPUE to be higher in subsequent weeks than if fishing had been present. A proper treatment of this would require a biomass-dynamic model taking into account mortality, growth, recruitment and migration; such a treatment would be beyond the scope of this study.

A further refinement to the full-moon scenario would be to optimize the timing of the closure. There is a suggestion from Figure 4.6 in section E.1 ‘The CPUE model’ that the minimum CPUE actually occurs a day or two *after* full moon. It is therefore possible that lagging the closure by a day or so would slightly increase the overall CPUE. It would be prudent to obtain alternative data to find out whether this lag effect is real.

Although the mean CPUE is higher for the full-moon scenarios, the inter-annual variation is quite large and the size of the effect can be reversed. For instance, in our simulations the mean CPUE in 2019 was slightly higher for the *Seasonal* scenario than for either full-moon scenario (Figure 4.2). Such variation may be related to the timing of full moon within the
season, but the relationship, if one exists, is not straightforward. One the other hand it may simply be random variation, since the standard deviations themselves are much larger than the difference in the means.

![Graph showing annual catch, effort, and CPUE]

**Figure 4.2.** Annual catch (top), effort (bottom left) and CPUE (bottom right) with error bars representing standard deviation over 20 replicate simulations for the Seasonal (blue), FullMoon/EffortDisplaced (black) and FullMoon/EffortRemoved (red) scenarios.

### 4.5.3. Benthic biomass changes

With respect to performance measure 3, the scenarios in which effort is displaced, either spatially (Reef Buffers, Effort Displaced and Masig Closure, Effort Displaced) or temporally (Full Moon Closure, Effort Displaced) have little effect. However, the reduction scenarios (Masig Closure, Effort Removed; Effort Reduction; and Full Moon Closure, Effort Removed) all have a significant effect, the size depending on the degree of effort removal.

From Table 4.1 it is clear that the overall impact on benthic fauna depends mainly on total effort and very slightly on spatial distribution of effort. This indicator is the mean biomass of 12 benthic groups relative to their initial biomass under the assumption that in their pristine state they were uniformly distributed. Figure 4.3 shows the separate indicators for each benthic group. The overall resilience of a benthic group is strongly related to the ratio \( r/d \). This ratio is the *extinction coverage*: it is the minimum number of tows per year over the same patch of ground to make the benthic group go locally extinct. The higher the extinction coverage the more resilient the benthic group.
Figure 4.3. Relative biomass in 2020 of 12 benthic groups for each scenario. The values have been jittered slightly to prevent overplotting. The scenarios are ordered by overall relative biomass. Standard deviations (omitted for clarity) are around 1%.

From Figure 4.4 we see that Asteroidea have a fairly high extinction coverage of 10, and so Asteroidea have a relatively high final relative biomass (around 85%). At the other extreme is Gastropoda with an extinction coverage of 2; Gastropoda have a relatively low final relative biomass (around 60%). Some groups have the same or very similar extinction coverage (Porifera and Bivalvia; Ophiuroidea, Crinoidea and Hydrozoa) and so they have very similar final relative biomass.

With respect to the scenarios themselves, the message is similar to the summary performance measure in Figure 4.3: the bottom 5 scenarios, with roughly the same amount of effort (100,000 hours per year), are very similar to one another whereas the top 3 scenarios have increasingly more benign effect of the benthic groups (the top scenario having only 55,000 hours effort per year).
Figure 4.4. Recovery and depletion parameter estimates from Poiner et al (1998) and Hill et al (2002). Extinction coverage is the minimum number of times per year a patch would be trawled to make the benthic group go locally extinct.

If we take into account the spatial distribution of the benthic groups, two effects occur: first, the overall resilience of the benthic groups changes; and second, some differentiation occurs among the spatially explicit scenarios. Figure 4.5 shows the overall density in 2020 relative to the pristine density, taking into account the estimated spatial distribution. This is in contrast to Figure 4.3 which shows the same ratio, but under the assumption of uniform spatial distribution. This shows the effect of the interaction of biomass distribution with effort distribution. Where there is a positive correlation between effort and biomass the density ratio falls (Echinoidea, Holothuroidea). If the correlation is negative (large effort in low biomass areas) the density ratio rises (Crinoidea, Hydrozoa).
Figure 4.5. Ratio of mean density in 2020 to mean pristine density of 11 benthic groups for each scenario. The scenarios are ordered by overall relative biomass as in Figure 4.3.

Figure 4.5 also shows the second effect of differentiation among the spatially explicit scenarios. On average, the bottom 5 scenarios should have equal impact on the benthic groups. However, for Porifera the Masig/EffortDisplaced scenario has a greater impact than the StatusQuo scenario. This is because the pristine density is on average higher outside the Masig closure than inside. The effort that is displaced from within the Masig closure therefore has a greater impact on Porifera than if it had not been displaced. On the other hand Holothuroidea (and perhaps Bivalvia) benefit from the Masig closure, because they have higher pristine densities inside the closure.

This is shown in even more detail in Figure 4.6, which allows us to see the standard deviations as well. The variation across replicate simulations is such that the bottom 5 scenarios are largely indistinguishable. Also it may be that the spatial effect for Porifera and Bivalvia is merely random variation, but the effect for Holothuroidea appears real. In fact
Table 4.2. Parameters $r$ and $d$ for benthic groups; $d$ comes from Poiner et al (1998) and $r$ from Hill et al (2002).

<table>
<thead>
<tr>
<th>Benthic group</th>
<th>$r$ (year$^{-1}$)</th>
<th>$d$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Asteroidea</td>
<td>0.97</td>
<td>0.10</td>
</tr>
<tr>
<td>Bivalvia</td>
<td>0.52</td>
<td>0.09</td>
</tr>
<tr>
<td>Bryozoa</td>
<td>0.40</td>
<td>0.09</td>
</tr>
<tr>
<td>Crinoidea</td>
<td>0.56</td>
<td>0.08</td>
</tr>
<tr>
<td>Crustacea</td>
<td>0.52</td>
<td>0.13</td>
</tr>
<tr>
<td>Echinoidea</td>
<td>0.40</td>
<td>0.14</td>
</tr>
<tr>
<td>Gastropoda</td>
<td>0.41</td>
<td>0.20</td>
</tr>
<tr>
<td>Holothuroidea</td>
<td>0.56</td>
<td>0.11</td>
</tr>
<tr>
<td>Hydrozoa</td>
<td>0.56</td>
<td>0.08</td>
</tr>
<tr>
<td>Ophiuroidea</td>
<td>0.63</td>
<td>0.09</td>
</tr>
<tr>
<td>Porifera</td>
<td>0.71</td>
<td>0.12</td>
</tr>
</tbody>
</table>

Figure 4.6. Overall density in 2020 of 12 functional groups for each scenario. The scenarios are ordered by overall relative biomass as in Figure 4.3. Standard deviations over 20 replicates are indicated by the horizontal bars.
Figure 4.7 shows that the highest density region for Holothuroidea is a band that encloses the Masig region. (Masig is the larger island at position 143.5°E, 9.75°S.)

The depiction of process error as represented by the error bars in Figure 4.6 can be misleading. This is because the errors themselves are strongly correlated with the effort used in the simulation. For simulations with lower than average effort, the biomass indicator will be higher than average, and vice versa. That is, the errors in each scenario are correlated with effort. This means that differences between scenarios in Figure 4.6 are more significant than they appear from the error bars.

Figure 4.7. Density (g/Ha) of sea cucumbers (Holothuroidea) (left) in the pristine state and (right) in 2020 under the StatusQuo scenario. The colour scale is given in Figure 3.5.

4.5.4. Effort change maps

The spatial distribution of effort in the various scenarios can be understood from Figure 4.8. These maps are not actual effort but the change in effort from StatusQuo. (The scenarios on a weekly time-step are not shown here because spatially they are similar to StatusQuo.) In the Masig Closure, Effort Displaced scenario all effort is removed from the closure and displaced to the rest of the trawl grounds. Hence the effort is slightly bigger than StatusQuo outside the Masig closure. In contrast the Masig Closure, Effort Removed scenario is very similar to
StatusQuo outside the closure. (The slight discrepancies are due to random variation). The Effort Reduction scenario is everywhere less than StatusQuo. The Reef Closure, Effort Displaced scenario is similar to the Masig Closure, Effort Displaced scenario in that everywhere effort is increased slightly, except close to reefs mainly in the north east.

It is important to realize that the definition of the study area necessarily comprises a buffer region in which no effort has ever been recorded. The model assumes that this region will remain unfished and that the benthos will remain in a pristine state. The effect of this region is to dilute the apparent impact on the benthos. Of the 2524 1-minute grid cells in the study area, 924 are in the buffer. If $b_{SA}$ is the relative biomass over the study area and $b_{TC}$ is the relative biomass only over the trawled cells then $b_{TC} = b_{SA} - \frac{924}{(2524-924)}(1- b_{SA})$. For example, for StatusQuo $b_{SA} = 54.2\%$, but $b_{TC} = 28.2\%$; and indeed there are some highly trawled cells in which the relative biomass is very small indeed. This diluting effect, being linear (or, more correctly, affine), is less important when comparing biomass indicators across scenarios; but it should be remembered when using the indicator value itself as an indicator of ecosystem health for instance.
Figure 4.8. Change in effort with respect to StatusQuo for the annual time-step scenarios in 2020. The units of effort are hours per 1-minute grid cell. The weekly time-step scenarios are very similar to the StatusQuo.

4.5.5. Uncertainty

The results of MSE simulations are subject to various sources of uncertainty. These can be classified as process variation and observation error. Process variation is the variation in the natural processes that are being simulated. If we have an idea of the magnitudes and sources of such variation, process variation can in principle be incorporated in the simulation. The following lists some examples of process variation:

1. the depletion per tow may vary from the proportional case
2. recovery may be slower or faster than logistic recovery
3. biomass may be affected by recruitment from or migration to nearboiring cells
4. effort within a grid cell may be aggregated
5. the spatial pattern of effort among grid cells may vary
6. the temporal pattern of effort within season may vary
7. total effort may vary among years

The MSE model attempts to address items 5 and 6 by drawing on empirical distributions of
effort patterns. Item 7 is modelled explicitly by incorporating the lognormal variate with coefficient of variation estimated from variation in total effort over representative years. The degree of aggregation at sub 1-nautical mile scale is assumed random from other studies (Rijnsdorp et al., 1998). With item 1, although the model does not explicitly allow variable depletion, the parameter \(d\) itself was estimated under such a model (Burridge et al., 2003). The \(d\) parameter is the mean depletion rate under that model. The logistic recovery curve (item 2) really represents a distillation of many complex processes, such as those in item 3, which are captured in a single parameter \(r\). There are many more sophisticated alternatives to such a treatment, but they all require more knowledge of the processes involved. Although the variation of the benthic dynamic process is ignored in the simulation, the results for averaged quantities, like mean biomass over the region, should be fairly robust to such variation.

Observation error applies to parameters of the model that have been estimated with uncertainty. These include:

1. aggregation parameter \(\beta\)
2. depletion rate \(d\)
3. recovery rate \(r\)
4. pristine biomass \(B_0\)

Uncertainty in \(\beta\) is probably of lesser importance because the effect of \(\beta\) is an adjustment to \(d\). For studies such as this one with very fine spatial resolution the approximation of random trawling is fairly good. If effort is known only at very coarse resolution however, it is important to have a good estimate of \(\beta\). The depletion rates reported by Burridge et al. (2003) are reasonably well estimated. However, the recovery rate is very uncertain and results from Pitcher et al. (2007a) suggest even wider variation than used here. For \(B_0\), the generalized linear models do provide estimates of uncertainty: typically the standard error is the same order of magnitude as the mean.

### 4.6. Conclusions

The effect of measurement errors is generally much larger than the process error in the simulation. For instance, in Figure 4.6, the error bars represent process error. However, the measurement error due to uncertainty in \(B_0\) is large. We stated earlier that the main effect of the biomass model was through the ratio \(r/d\) (extinction coverage). Uncertainty in \(r\) therefore has a large effect. We can get a feel for the size of the effect by considering the average effort under StatusQuo, which is 100,000 hours over 2524 1-nautical mile cells or 1.8 tows per cell per year. For a biota that is uniformly distributed, the relative biomass would tend to \(100(1 – 1.8d/r) \%\). (or \(0\%\), whichever is larger). That is, the deficit from \(100\%\) is inversely proportional to \(r\).
Uncertainty in $r$ is especially important when $r$ is small.

The conclusion is that uncertainty in biomass indicators for a particular scenario is large. However, the ratios of indicators relative to the StatusQuo, say, are much more robust. To be concrete: we may be highly uncertain about the absolute value of the relative density of Echinoidea in 2020 under the Effort Reduction scenario; however, we can be confident that this value is larger than the corresponding value for the StatusQuo scenario.

The robustness of the indicator ratios is evident in Figure 4.9. For the relative biomass ratio, there is very little difference across species. The main effect is the change in effort among scenarios. The uncertainties due to observation error are not important here. The slight differentiation among species for the Masig scenarios is probably due to a transient temporal effect. Greater differentiation is evident once the spatial distribution is taken into account (Figure 4.9, bottom), especially for Holothuroidea. But this is a secondary effect after the main effect due to total effort.

The main message of these simulations is that total effort is the principal driver of the benthic performance measures. Although the final density depends principally on the joint effect of $r$, $d$ and effort, managers only have control over effort. Their main tool is to control the total amount of effort.
RESULTS OF MANAGEMENT SCENARIOS EVALUATION

Figure 4.9. (top) Ratio of relative biomass to StatusQuo relative biomass in 2020. (bottom) Ratio of relative density to StatusQuo relative density in 2020.

The spatial distribution of effort plays a lesser role. Its effect is really an adjustment to the main effect of the total effort. The reason why spatial effects are not very strong is probably a consequence of aggregating different benthic species under a coarse taxonomic grouping. Some species’ densities may be positively correlated with effort, and others’ negatively correlated. By combining the species, the depletion effect of the positively correlated species may be diluted by the effect of the others. The spatial effects will probably be stronger at species resolution.

For instance, according to the detailed decision table for some individual species (Figure 4.10), Dichotella sp1, Alertigorgia orientalis and Subergorgia suberosa are all impacted more by the Masig closure scenarios. This is because these species are restricted to the south-west part of the trawling area, away from the Masig closure (Figure 4.11). In this region, the trawl effort is slightly more intense than under StatusQuo.
Figure 4.10. Overall density in 2020 of 7 individual species for each scenario.
Figure 4.11. Density (g/Ha) of a soft coral (*Dichotella* sp1) in 2020 (*left*) in the pristine state and (*right*) in 2020 under the Masig/EffortDisplaced scenario. The colour scale is given in Figure 3.5.

We have seen that the full-moon closure option does have an effect on the catch, increasing the CPUE by about 2.5%. This efficiency is probably underestimated, owing to the simplistic modelling of the target species through a CPUE relationship. A combination of effort reduction and full-moon closure would be beneficial to the benthic system. For instance, if effort can be displaced temporarily (i.e. there is latency in the fleet), then, in principle, effort could be reduced by 2.5% and a full-moon closure introduced. Then the catch would remain the same and the benthos would benefit from reduced trawling pressure.
4.7. References


Results of Management Scenarios Evaluation


5. ECOSIM EXAMINATION

5.1. Introduction

It is well known that fisheries can have severe effects not only on target stocks, but also on non-target components of ecosystems through indirect and cascading effects (Jones 1992, Dayton et al. 1995, Goni 1998, Pauly et al. 1998, Watling and Norse 1998, Jackson et al. 2001, Jennings et al. 2001, NRC 2002, Thrush and Dayton 2002). This ecological and stock degradation often has severe impacts on human economies and communities as well (e.g. Harris 1998, Hennessey and Healey 2000). Many countries and jurisdictions around the world are therefore very interested in improving the sustainability of their fisheries from a whole ecosystem perspective, but this has been viewed as a great challenge because of the complexity and intractability of ecosystems.

A very promising approach to assessing and projecting the impacts of fisheries (and other disturbances) on stocks, biological communities, and broader social and economic values of ecosystems is the Ecopath with Ecosim trophodynamic (food web) modelling approach. This approach has been in refinement for more than 20 years, and it is now a global standard for marine fisheries and ecosystem modelling with over 2400 registered users in over 120 countries.

*Ecopath* trophic models were first developed by Polovina (1984) to describe the state of energy (i.e. biomass) flows in a coral reefs food web at French Frigate Shoals. Since then, a variety of dynamic capabilities have been added (e.g., Christensen and Pauly 1992, Walters et al. 1997, Walters et al. 1999, Christensen et al. 2000, Pauly et al. 2000). Scores of applications of *Ecopath with Ecosim* can be found at: [http://www.ecopath.org/](http://www.ecopath.org/), along with the freely distributed software and documentation. Although the formulations and basic concepts are accessible in these venues, the general approach is summarized here.

5.2. Methods

The main approach to this component of the project was to adapt Gribble’s (2003) Northern Great Barrier Reef Model to the Torres Strait Ecosystem. This was accomplished by adding six functional groups to Gribble’s GBR model that are considered important in the Torres Strait ecosystem, by re-estimating many of the biomasses of the existing functional groups to represent the defined Torres Strait ecosystem, and by defining and specifying the Torres Strait fisheries fleets and catches. The area that was defined as the Torres Strait for the purpose of this
exercise corresponds with the area where most prawn trawling effort is concentrated. Parameters were calculated for the Torres Strait where data were available, as documented in Appendix A. All other parameters were taken from the Northern Great Barrier Reef model and other tropical reef models.

The Ecopath master equation, Eq. (5-1), 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.

\[ B_i (P/B_i) \cdot EE_i = Y_i + \sum_{j=1}^{N_{pred}} B_j (Q/B_j) \cdot DC_{ji} + BA_i + NM_i \]  

(5-1)

\( B_i \) 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 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 modeling 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 (see 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 (Eq.
\[
dB_i \over dt = g_i \sum_j Q_{ji} - \sum_j Q_{ji} + I_i - (M_{0i} + F_i + e_i) \cdot B_i \tag{5-2}
\]

\(dB_i/dt\) 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, \(M_{0i}\) 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 9-2), 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 Eq. (5-3) includes the prey vulnerability parameters.

\[
Q_{ji} = \frac{v_{ij} \cdot a_{ij} \cdot B_i \cdot B_j \cdot T_i \cdot T_j \cdot S_{ij} \cdot M_{ij} / D_j}{v_{ij} + v'_{ij} \cdot T_i \cdot M_{ij} + a_{ij} \cdot M_{ij} \cdot B_j \cdot S_{ij} \cdot T_j / D_j} \tag{5-3}
\]

where \(a_{ij}\) 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 (Christensen et al. 2004) for more information.

Parameters \(v_{ij}\) and \(v'_{ij}\) represent prey vulnerabilities, or the rate of exchange of biomass between two prey behavioral 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 behavioral 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. 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 (see 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 or as a 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, e.g., predator-prey interactions.

5.3. Results

The basic parameters for the Torres Strait Ecopath model are shown in Table 5.1. The estimated diet compositions and fisheries information is found in Appendix B.
### Table 5.1 Basic parameters for the Torres Strait food web using Ecopath

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

* Functional groups added to the Northern Great Barrier Reef model relevant to Torres Strait
Figure 5.1 The results of the relative biomass dynamics (Y axis) over 20 years (X axis) as simulated by the Ecosim/Ecopath model of the Torres Strait.

Figure 5.1 shows the dynamics of the populations for which data was available. According to the model, *Metapenaeus endeavouri* (endeavour prawns), seabirds and large sharks and rays and *Penaeus longistylus* (red spot king prawn) will decline below 50% of their initial biomass after about 5 years. Large gropers, large carnivore fish, sessile animals and phytoplankton will all have exceeded their initial (standardised) biomasses.

### 5.4. Discussion

This Ecopath model of the Torres Strait can be used in dynamic simulations to estimate the direct and indirect impacts of trawling on this ecosystem. Although the model is quantitative, initial results of such simulations using this preliminary model will need to be interpreted in qualitative rather than a quantitative way because of the preliminary nature of the specification of this model. This model has not gone through any evaluation with respect to its validity for the particular question at hand. The value of this iteration of the model is as a starting point for assembling the quality information that will enable model refinement so that simulations are useful for management in the future.

Modifying the Great Barrier Reef prawn Ecopath model for the Torres Strait area resulted in utilising many parameters calculated specifically for the Great Barrier Reef. More site-
specific data is obviously preferred for future refinements. Fortunately, much more such data are becoming available for the Torres Strait, so there will be good potential to refine the present model and resulting simulations in a meaningful way in the near future.

5.5. Conclusion

The purpose of the present exercise was to initiate a framework for the future refinement of a simple Torres Strait Ecopath model so that the estimations of direct impacts of trawling made in other components of this project could eventually be utilised in a broader community and ecosystem context. The model in its present form should not be used in simulations for exploring the temporal or spatial dynamics of the Torres Strait or components therein because it needs thorough refinement by one or more ecologists who can coordinate teams of experts in addition to reviewing and synthesizing existing information.

5.6. Acknowledgements

We wish to express our gratitude to Dr. Neil Gribble, of the Queensland Department of Primary Industries and Fisheries for his approval and encouragement in the use of his Northern Great Barrier Reef Ecopath model as the basis for the development of the present straw-man model of the Torres Strait.

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6. BENTHIC-PELAGIC COUPLING AND TRAWL IMPACTS

6.1. Introduction

The aim of the ecosystem model developed for this CRC task was to predict the primary effects of trawling on benthic biota. Primary effects were defined to be the effect of trawling on the abundance and species composition of benthic biota as determined by species-specific rates of depletion by trawling and recovery following trawling.

At present the ecosystem model does not consider secondary effects of trawling, but may be enhanced to consider such effects in a follow-up project. Secondary effects include disruption of the food web (e.g. less food for predators whose prey are depleted by trawling) and disruption of ecosystem processes (e.g. trawling disturbance may affect the exchange of nutrients and oxygen with the sediment).

To help scope a potential follow-up project, we performed a preliminary evaluation of benthic-pelagic coupling processes that could be added to the ecosystem model to predict secondary effects of trawling on the benthic ecosystem. Benthic-pelagic coupling processes are processes that link the seafloor and the water column in some way, for example by the movement of animals, plants or nutrients. A benthic-pelagic coupling process may mediate secondary effects of trawling in three ways. First, the process itself may be disrupted by trawling (e.g. removal of benthic filter feeders disrupts the transfer of plankton and microbes from the water column to the seafloor). Second, the process may facilitate recovery from trawling (e.g. depleted crabs will be replaced by crab larvae settling from the water column). Finally, the process may communicate an effect on the seafloor to the water column (e.g. if crabs are deleted, then fewer crab larvae will be released to the water column as plankton).

6.2. Methods

To evaluate benthic-pelagic coupling processes for inclusion in the ecosystem model, we aggregated processes identified in the literature into 13 generic processes (see Results). We then scored each generic process according to four criteria:

1. Is the process affected by trawling?
2. Does the process affect recovery?
3. Would knowledge about the process or its effects be relevant to decision making?
4. Can the process be modelled practically and accurately enough to support decision making?

The scores presented here are preliminary because criteria 1, 2 and 4 require detailed knowledge of processes for complete assessment and criterion 3 requires knowledge of the particular decisions to be made. However, they are sufficiently indicative to provide a priority listing of processes for further investigation.

6.3. Results

The 13 generic benthic-pelagic coupling processes and their evaluations are summarised in Table 6.1. The details of the evaluations are presented in the following subsections.

Table 6.1 Summary of benthic-pelagic coupling process evaluations. The priority column indicates the priority for further investigation.

<table>
<thead>
<tr>
<th>Process</th>
<th>Affected by trawling</th>
<th>Affects recovery</th>
<th>Affects decision</th>
<th>Can be modelled</th>
<th>Priority</th>
</tr>
</thead>
<tbody>
<tr>
<td>Consumption of benthos by pelagic animals</td>
<td>yes</td>
<td>no</td>
<td>no</td>
<td>yes</td>
<td>*</td>
</tr>
<tr>
<td>Commuting between the benthos and the water column</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>*</td>
</tr>
<tr>
<td>Feeding by benthic filter feeders</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>***</td>
</tr>
<tr>
<td>Input of larvae and propagules from the water column</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>***</td>
</tr>
<tr>
<td>Return from resting sessile stages to the water</td>
<td>yes</td>
<td>no</td>
<td>no</td>
<td>yes</td>
<td>*</td>
</tr>
<tr>
<td>Fall of non-living organic matter</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>**</td>
</tr>
<tr>
<td>Consumption by scavengers and detritus feeders</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>**</td>
</tr>
<tr>
<td>Decomposition of detritus</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>no</td>
<td>*</td>
</tr>
<tr>
<td>Transfer of oxygen from the water column to the sediment</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>*</td>
</tr>
<tr>
<td>Transfer of nutrients from the sediment to the water column</td>
<td>yes</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>*</td>
</tr>
<tr>
<td>Bioturbation</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>*</td>
</tr>
<tr>
<td>Resuspension of sediment</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>yes</td>
<td>**</td>
</tr>
<tr>
<td>Deposition of sediment</td>
<td>yes</td>
<td>yes</td>
<td>no</td>
<td>no</td>
<td>*</td>
</tr>
</tbody>
</table>
6.4. Consumption of benthos by pelagic animals

6.4.1. Description

Pelagic predators and herbivores feed on benthic animals and plants, and transfer energy from the benthos to the water column.

6.4.2. Effect of trawling

Affected by trawling: yes
Food for pelagic predators and herbivores will be reduced by benthic biomass removal. Predation may be decreased by removal of predators (Jennings and Kaiser 1998). Predation may be more efficient if trawling disrupts habitat structure that provides prey refuges (Ruth and Lindholm 2002, Thrush and Dayton 2002).

6.4.3. Effect on recovery

Affects recovery: no
If the abundance of pelagic predators or herbivores is reduced, then recovery may be affected (e.g. enhanced); however, such an affect is less likely in the open Torres Strait system. Refuge removal could affect recovery—but is refuge removal significant on soft bottom sediments?

6.4.4. Affects decision

Affects decision: no
We are only managing impacts on benthic species and habitats.

6.4.5. Can be modelled

Can be modelled: yes
The effect could be modelled with Ecopath based on depletion estimates from the trawl model. However, the results are likely to be more uncertain than the basic trawl model outputs (depending on the sensitivity of the Ecopath model the trawl model outputs).
6.5. Commuting between the benthos and the water column

6.5.1. Description

Many animals make diurnal vertical movements between the benthos and the water column. This can facilitate the transfer of organic material to the benthos if, for example, an animal feeds in the water column and is eaten or deposits faeces on the seafloor (Raffaelli et al. 2003).

6.5.2. Effect of trawling

Affected by trawling: yes
Removal of commuters by trawling will disrupt associated benthic-pelagic coupling processes such as transfer of material to the benthos.

6.5.3. Effect on recovery

Affects recovery: yes
If trawling removes commuters, then the amount or form of energy reaching the benthos will be altered. What do we mean by form? For example, if commuters were removed, energy previously arriving as live commuters and their faeces might instead arrive as falling detritus from unconsumed primary producers.

6.5.4. Affects decision

Affects decision: yes
The process directly affects benthos recovery. However, is the effect large enough to matter?

6.5.5. Can be modelled

Can be modelled: no
The process is likely to be highly variable and difficult to parameterise.

6.6. Feeding by benthic filter feeders
6.6.1. Description

Benthic filter feeders filter plankton and microbes from the water column (Thrush and Dayton 2002). They include sponges, ascidians, bryozoans, phoronids, brachiopods, fan worms, scallops, clams, oysters, mussels and barnacles. Corals (including soft corals) perform the same function, although they are not often referred to as filter feeders.

6.6.2. Effect of trawling

Affected by trawling: yes
Trawling induced mortality of filter feeders will disrupt the transfer of energy from the water column to the benthos.

6.6.3. Effect on recovery

Affects recovery: yes
Recovery rates may be reduced by the reduced transfer of energy from the water column to the benthos.

6.6.4. Affects decision

Affects decision: yes
The process directly affects recovery. How important is it in Torres Strait?

6.6.5. Can be modelled

Can be modelled: probably yes
We really don't know whether this can be modelled or not, but will investigate further if the process appears to be important.

6.7. Input of larvae and propagules from the water column

6.7.1. Description

Larvae and propagules of benthic animals and plants settle from the water column (Raffaelli et
al. 2003). They are a source of recruits and food. Note that recruitment may also occur by other means such as movement of adults and juveniles across the seabed (Ruth and Lindholm 2002) or, for modular organisms, dispersal of fragments. In some species, the benthic lifestage is active (e.g. jellyfish polyps), whereas in others it is dormant (e.g. eggs) (Raffaelli et al. 2003).

6.7.2. Effect of trawling

Affected by trawling: yes
Recruitment decisions by infauna can be affected by changes in sediment chemistry that result from disturbance (Marinelli and Woodin 2002).

6.7.3. Effect on recovery

Affects recovery: yes
Larvae settling from the water column provide recruits for recovery (Ruth and Lindholm 2002). For some species, an alternative source of recruits is advection of post-settlement lifestages (juveniles and adults) across the seabed (Ruth and Lindholm 2002). The relative importance of settling larvae versus advecting post-settlement lifestages will increase with the size of the area trawled (Ruth and Lindholm 2002).

6.7.4. Affects decision

Affects decision: yes
Recruitment from the water column directly affects recovery.

6.7.5. Can be modelled

Can be modelled: yes
Movement is very difficult to model because it is difficult to parameterise and highly variable in space and time. Recruitment from the water column will need to be modelled—even if only as a constant or seasonal supply. If the regional population of a species is being modelled, then movement models assuming uniform dispersal are often as good as more explicit models when incertitude and variability are high. One exception is where strong, directional source–sink relationships apply, which may be the case in Torres Strait where currents are strong. Ruth and Lindholm (2002) describe a model of recolonisation, but do not validate the model.
Effects of trawling on recruitment decisions cannot be modelled because current research is still concerned with detecting the effects rather than parameterising the effect for a range of taxa.

6.8. **Return from resting/sessile stages to the water**

6.8.1. **Description**

Animals hatch from resting stages (spores, cysts, resting eggs) and sessile stages (polyps) and return to the water column (Raffaelli et al. 2003).

6.8.2. **Effect of trawling**

Affected by trawling: yes

Plankton communities are often dominated by invertebrate larvae (Jennings and Kaiser 1998). Trawling induced changes in benthic species composition appear to affect plankton composition (Jennings and Kaiser 1998).

6.8.3. **Effect on recovery**

Affects recovery: no

In the open Torres Strait system the plankton community will be determined by regional rather than local inputs.

6.8.4. **Affects decision**

Affects decision: no

We are only managing benthic species and habitats.

6.8.5. **Can be modelled**

Can be modelled: yes

The impact of trawling on the abundance of sessile stages such as jellyfish polyps would be a direct output of the trawl model. The impact of trawling on the abundance of spores, cysts, eggs...
6.9. Fall of non-living organic matter

6.9.1. Description

Non-living organic matter falls from the water column to the benthos in such forms as dead organisms and faeces (Raffaelli et al. 2003).

6.9.2. Effect of trawling

Affected by trawling: yes

The input of organic matter can potentially decrease if trawling disrupts nutrient cycling and therefore primary production; however, that affect is likely to be small in the open Torres Strait system. Alternatively, input of organic matter may increase due to bycatch discards. The sediment plume created by trawling may increase detritus and nutrient inputs to adjacent untrawled areas. The rate or pattern of detritus deposition may change if filter feeders are removed. Filter feeding molluscs don't just remove ingested material from the water column: they package unsuitable material in mucus and deposit that too (Thrush and Dayton 2002).

6.9.3. Effect on recovery

Affects recovery: yes

Non-living organic matter from the water column provides energy and nutrients for recovery. Any trawling induced changes to the fall of organic matter will affect the rate or pattern of recovery.

6.9.4. Affects decision

Affects decision: yes

A substantial change to the fall of non-living organic matter would directly affect recovery.

6.9.5. Can be modelled

Can be modelled: yes
The effect of removal of filter feeders on detritus deposition could potentially be important, although it may be difficult to parameterise. Effects of sediment plumes are worth looking at. Effects of nutrient cycling changes on primary production will be small in the open Torres Strait system. Effects of bycatch discards would be localised and highly variable.

6.10. **Consumption by scavengers and detritus feeders**

6.10.1. **Description**
Dead plant and animal material, some of which falls from the water column, is consumed by scavengers and detritus feeders.

6.10.2. **Effect of trawling**
Affected by trawling: yes
Detritus may accumulate if trawling removes scavengers and detritus feeders and reduces the consumption of primary production (Jennings et al. 2001). However, changes in community composition or function may cause consumption to remain constant by increasing the consumption to biomass ratio (Jennings et al. 2001).

6.10.3. **Effect on recovery**
Affects recovery: yes
Any trawling induced change in scavenger and detritus feeder abundance or activity will affect the rate or pattern of recovery. For example, if abundance was reduced there would be less food for predators. If activity was reduced, decomposition rates might also be reduced, thus slowing the release of nutrients for plants.

6.10.4. **Affects decision**
Affects decision: yes
Changes to consumption by scavengers and detritus feeders directly and indirectly affect recovery.
6.10.5. Can be modelled

Can be modelled: yes
Ecopath could model the direct and indirect affects on recovery. Results may be sensitive to trawling model estimates of the direct effects of trawling on the abundances of different species.

6.11. Decomposition of detritus

6.11.1. Description
Bacteria and fungi decompose detritus to release energy and nutrients.

6.11.2. Effect of trawling
Affected by trawling: yes
Trawling disturbance may increase contact between bacteria and detritus (Jennings et al. 2001).

6.11.3. Effect on recovery
Affects recovery: yes
Any trawling induced change to decomposition rates may affect the release of nutrients for plants.

6.11.4. Affects decision
Affects decision: no
The changes may be short-term.

6.11.5. Can be modelled
Can be modelled: no
The effect is likely to be small and impossible to parameterise.
6.12. Transfer of oxygen from the water column to the sediment

6.12.1. Description
Dissolved oxygen is carried by diffusion or water flow from the water column to the sediment.

6.12.2. Effect of trawling
Affected by trawling: yes
Trawling disturbance may increase the depth of the aerobic layer (Jennings et al. 2001). Trawling may decrease exchange due to decreased porewater advection due to disruption of burrow mounds and ripples (Raffaelli et al. 2003).

6.12.3. Effect on recovery
Affects recovery: yes
The increased depth of the aerobic layer may increase the production of meiofauna (Jennings et al. 2001).

6.12.4. Affects decision
Affects decision: yes
If the effect is significant, it will directly affect recovery.

6.12.5. Can be modelled
Can be modelled: no
Affects of sediment topography on porewater advection have been modelled; however, in the Torres Strait effect is likely to be small and impossible to parameterise.
6.13. Transfer of nutrients from the sediment to the water column

6.13.1. Description

Dissolved nutrients such as nitrate and phosphate are carried by diffusion or water flow from the sediment to the water column (Raffaelli et al. 2003). Often, the nutrients are the product of demineralisation of organic material.

6.13.2. Effect of trawling

Affected by trawling: yes
Trawling causes a pulsed release of nutrients from the sediment to the water column (Thrush and Dayton 2002). Trawling may increase cycling due to altered redox due to disturbance (Jennings et al. 2001). Trawling may decrease exchange due to reduction of surface area for exchange due to disruption of burrows (Raffaelli et al. 2003). Trawling may decrease exchange due to decreased porewater advection due to disruption of burrow mounds and ripples (Raffaelli et al. 2003).

6.13.3. Effect on recovery

Affects recovery: no
In the open Torres Strait system, nutrient levels in the water column will be determined by regional rather than local inputs.

6.13.4. Affects decision

Affects decision: no
We are only managing effects on benthic species and habitats.

6.13.5. Can be modelled

Can be modelled: no
The effect is likely to be small and difficult to parameterise.
6.14. **Bioturbation**

6.14.1. **Description**

Burrowing animals mix the sediment, facilitating the movement of organic material, oxygen and nutrients (Rosenberg 2001).

6.14.2. **Effect of trawling**

Affected by trawling: yes
Trawling causes mortality of bioturbators.

6.14.3. **Effect on recovery**

Affects recovery: yes
Bioturbation is an important mechanism for exchange of detritus and nutrients within the sediment (Thrush and Dayton 2002, Rosenberg 2001).

6.14.4. **Affects decision**

Affects decision: yes
Mortality of bioturbators could affect recovery significantly and for a significant time.

6.14.5. **Can be modelled**

Can be modelled: no
Although the mortality and recovery of bioturbators could be modelled, affects on and of bioturbation may be impossible to parameterise.

6.15. **Resuspension of sediment**
6.15.1. Description

Sediment is resuspended in the water column by natural water movement (including during storms) or trawling.

6.15.2. Effect of trawling

Affected by trawling: yes
Trawling resuspends sediment (Jennings and Kaiser 1998). Trawling may destabilise sediment by e.g. disrupting worm tube mats or seagrass rhizomes (Jennings and Kaiser 1998).

6.15.3. Effect on recovery

Affects recovery: yes
Resuspended sediment may clog the feeding apparatus of filter feeders (Jennings and Kaiser 1998, Thrush and Dayton 2002). Resuspended sediment may reduce light availability (Jennings and Kaiser 1998).

6.15.4. Affects decision

Affects decision: yes
Affects on filter feeders or seagrass would directly affect recovery.

6.15.5. Can be modelled

Can be modelled: yes
Directly modelling affects on and of resuspension of sediment is unlikely to be practical. However, mortality associated with sediment resuspension could be modelled through mortality parameters.

6.16. Deposition of sediment
6.16.1. Description
Suspended sediment falls onto the seabed.

6.16.2. Effect of trawling
Affected by trawling: yes
Deposition of sediment may be increased by deposition of sediment resuspended by trawling (Jennings and Kaiser 1998).

6.16.3. Effect on recovery
Affects recovery: yes
Sediment deposition inhibits the settlement and growth of species such as oysters and scallops (Jennings and Kaiser 1998).

6.16.4. Affects decision
Affects decision: no
The effect is unlikely to be important in Torres Strait, where sediment movement is naturally high.

6.16.5. Can be modelled
Can be modelled: no
The process would be very difficult to parameterise.

6.17. Discussion
Key benthic-pelagic coupling processes identified for further investigation were feeding by benthic filter feeders and input of larvae and propagules from the water column. Feeding by benthic filter feeders is potentially important for transferring organic material from the water column to the seabed, and is disrupted by trawling (Thrush and Dayton 2002). The input of larvae and propagules from the water column is a source of recruits for recovery following trawling (Ruth and Lindholm 2002). The importance of this process relative to other methods of
recruitment (e.g. movement across the seabed) varies among species, in turn varying recovery patterns among species.

Further investigation of whether each benthic-pelagic coupling process should be added to the ecosystem model will in particular address (a) how important the process is within the particular environment of the Torres Strait and (b) what methods can be used to model the process.

6.18. References


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APPENDIX A: A TORRES STRAIT ECOPATH MODEL

A.1. SUMMARY

This Appendix describes a first attempt to construct an ecosystem model for the Torres Strait and is based on a model for the Northern Great Barrier Reef model (Gribble, 2003).

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

The purpose of the preliminary food web of the Torres Strait is to provide a means for examining the complexity of modelling secondary effects of benthic biomass removal on the ecosystem. The area of interest corresponds with the area where most prawn trawling effort is concentrated. Modifying the Great Barrier Reef prawn Ecopath model for the Torres Strait area results in utilising many parameters already calculated for the Great Barrier Reef, although site-specific data would be preferred. The functional groups in the Great Barrier Reef model were used as a basis for the Torres Strait food web, with additional groups relevant to the area added (Table A.1). Parameters were calculated for the Torres Strait where data was available, as documented in the following sections, with all other parameters taken from the Northern Great Barrier Reef model and other tropical reef models (Table A.1).
A.2. MODEL INPUTS

A.2.1. Primary Producers and Detritus

Benthic autotrophic organisms

Benthic autotrophic biota represent the seagrass, algae and coral algal symbiotic populations of the Northern Great Barrier Reef, with their combined average biomass being the largest of any other functional group in the area (Gribble, 2003). The Torres Strait food web however, has seagrass as a separate functional group, which is included in more detail below, due to Dugong dependence on seagrass for food and their importance as prawn nursery grounds. Benthic autotrophic biota in the Torres Strait food web therefore, represent algae and coral algal symbiotic populations. Common algae in the Torres Strait include those from the genre Halimeda, Caulerpa and Sargassum, with the Halimeda genre widespread across the entire Torres Strait showing little or no substrate preference (Pitcher et al., 1992). The Caulerpa genre shows some preference for habitat with rubble, whereas Sargassum appears restricted to hard substrata due to the need of a solid attachment for the holdfast (Pitcher et al., 1992). As well as providing food and shelter for many organisms, as with the other algae genre, Halimeda also has the potential to contribute substantially to sediment carbonate due to abundant skeletons (Pitcher et al., 1992).
Table A.1. Basic parameters for the Torres Strait food web using Ecopath

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

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

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

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

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

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

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

<table>
<thead>
<tr>
<th>Bycatch species</th>
<th>Percent of total catch</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bony fish</td>
<td>52-69</td>
</tr>
<tr>
<td>Crabs</td>
<td>8</td>
</tr>
<tr>
<td>Scallops</td>
<td>3</td>
</tr>
<tr>
<td>Sharks and rays</td>
<td>1-3</td>
</tr>
<tr>
<td>Squid, tropical rock lobster, Moreton bay bugs, sponges and turtles</td>
<td>1</td>
</tr>
<tr>
<td>Sea snakes</td>
<td>0.1-0.2</td>
</tr>
</tbody>
</table>

Source: TSPZJA (2005)

A.2.2. Microbes

Decomposer/microfauna

The decomposer/microfauna functional group, incorporating single celled heterotrophs such as bacteria, ciliates and foraminiferans, is important in reintroducing dead organic matter back into the trophic cycle of reef systems (Opitz 1996). Biomass (6 t/km²), production (120/year) and consumption (400/year) rates for this functional group were taken directly from the Great Barrier Reef model (Gribble, 2001; 2003), although the model did not state how these values were calculated.
A.2.3. Plankton

Zooplankton

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

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

<table>
<thead>
<tr>
<th>Prey</th>
<th>Proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td>Phytoplankton</td>
<td>0.200</td>
</tr>
<tr>
<td>Benthic autotrophic biota</td>
<td>0.800</td>
</tr>
<tr>
<td>SUM</td>
<td>1.000</td>
</tr>
</tbody>
</table>

Source: Gribble (2003)

A.2.4. Macro-invertebrates

Pearl shell

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

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

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

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

<table>
<thead>
<tr>
<th>Prey</th>
<th>Proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td>Phytoplankton</td>
<td>0.290</td>
</tr>
<tr>
<td>Benthic autotrophs</td>
<td>0.300</td>
</tr>
<tr>
<td>Detritus/discards</td>
<td>0.190</td>
</tr>
<tr>
<td>Detritus</td>
<td>0.200</td>
</tr>
<tr>
<td>SUM</td>
<td>1.000</td>
</tr>
</tbody>
</table>

Source: Okey and Mahmoudi (2002)

**Benthic molluscs/worms**

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

<table>
<thead>
<tr>
<th>Prey</th>
<th>Proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td>Crustaceans</td>
<td>0.010</td>
</tr>
<tr>
<td>Echinoderms</td>
<td>0.002</td>
</tr>
<tr>
<td>Benthic molluscs/worms</td>
<td>0.065</td>
</tr>
<tr>
<td>Zooplankton</td>
<td>0.013</td>
</tr>
<tr>
<td>Sessile animals</td>
<td>0.050</td>
</tr>
<tr>
<td>Decomposer/microfauna</td>
<td>0.170</td>
</tr>
<tr>
<td>Phytoplankton</td>
<td>0.086</td>
</tr>
<tr>
<td>Benthic autotrophs</td>
<td>0.179</td>
</tr>
<tr>
<td>Detritus</td>
<td>0.425</td>
</tr>
<tr>
<td>SUM</td>
<td>1.000</td>
</tr>
</tbody>
</table>

Source: Gribble (2003)

A.2.5. Mega-invertebrates

Sessile animals

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

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

<table>
<thead>
<tr>
<th>Prey</th>
<th>Proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td>Zooplankton</td>
<td>0.025</td>
</tr>
<tr>
<td>Decomposer/microfauna</td>
<td>0.019</td>
</tr>
<tr>
<td>Phytoplankton</td>
<td>0.220</td>
</tr>
<tr>
<td>Benthic autotrophs</td>
<td>0.651</td>
</tr>
<tr>
<td>Detritus</td>
<td>0.085</td>
</tr>
<tr>
<td>SUM</td>
<td>1.000</td>
</tr>
</tbody>
</table>

Source: (Gribble, 2003)
**Echinoderms**

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

**Table A.7. Diet composition of echinoderms in northern GBR Ecopath model**

<table>
<thead>
<tr>
<th>Prey</th>
<th>Proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cephalopods</td>
<td>0.003</td>
</tr>
<tr>
<td>Crustaceans</td>
<td>0.002</td>
</tr>
<tr>
<td>Echinoderms</td>
<td>0.059</td>
</tr>
<tr>
<td>Benthic molluscs/worms</td>
<td>0.020</td>
</tr>
<tr>
<td>Zooplankton</td>
<td>0.053</td>
</tr>
<tr>
<td>Sessile animals</td>
<td>0.050</td>
</tr>
<tr>
<td>Decomposer/microfauna</td>
<td>0.149</td>
</tr>
<tr>
<td>Phytoplankton</td>
<td>0.003</td>
</tr>
<tr>
<td>Benthic autotrophs</td>
<td>0.554</td>
</tr>
<tr>
<td>Detritus</td>
<td>0.107</td>
</tr>
<tr>
<td><strong>SUM</strong></td>
<td><strong>1.000</strong></td>
</tr>
</tbody>
</table>

Source: (Gribble, 2003)

**Holothurians**

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

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

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

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

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

<table>
<thead>
<tr>
<th>Prey</th>
<th>Proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td>Phytoplankton</td>
<td>0.01</td>
</tr>
<tr>
<td>Benthic autotrophs</td>
<td>0.13</td>
</tr>
<tr>
<td>Detritus</td>
<td>0.86</td>
</tr>
<tr>
<td>SUM</td>
<td>1.00</td>
</tr>
</tbody>
</table>

Source: Hudson et al. (2004)

Trochus

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

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

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

<table>
<thead>
<tr>
<th>Reef</th>
<th>Total numbers</th>
<th>Total Weight (Tones)</th>
<th>weight of an individual (tones)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ashmore</td>
<td>21767.00</td>
<td>4.40</td>
<td>0.0002</td>
</tr>
<tr>
<td>Browse</td>
<td>0.00</td>
<td>0.00</td>
<td>0.0000</td>
</tr>
<tr>
<td>Cartier</td>
<td>222.00</td>
<td>0.00</td>
<td>n/a</td>
</tr>
<tr>
<td>Hibernia</td>
<td>0.00</td>
<td>0.00</td>
<td>0.0000</td>
</tr>
<tr>
<td>Scott Nth</td>
<td>0.00</td>
<td>0.00</td>
<td>0.0000</td>
</tr>
<tr>
<td>Scott Sth</td>
<td>718.00</td>
<td>0.10</td>
<td>0.0002</td>
</tr>
<tr>
<td>Seringapatam</td>
<td>0.00</td>
<td>0.00</td>
<td>0.0000</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>22706.00</strong></td>
<td><strong>4.60</strong></td>
<td><strong>Average =0.0002</strong></td>
</tr>
</tbody>
</table>

Source: adapted from CMR (1999)

### Table A.10. Diet composition of small gastropods in the Galapagos Ecopath model

<table>
<thead>
<tr>
<th>Prey</th>
<th>Proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td>Echinoderms</td>
<td>0.033</td>
</tr>
<tr>
<td>Benthic molluscs/worms</td>
<td>0.039</td>
</tr>
<tr>
<td>Sessile animals</td>
<td>0.002</td>
</tr>
<tr>
<td>Benthic autotrophs</td>
<td>0.456</td>
</tr>
<tr>
<td>Detritus</td>
<td>0.470</td>
</tr>
<tr>
<td><strong>SUM</strong></td>
<td><strong>1.000</strong></td>
</tr>
</tbody>
</table>

Source: Okey et al. (2004)

**Crustaceans**

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

**Table A.11. Diet composition of crustaceans in the northern GBR Ecopath model**

<table>
<thead>
<tr>
<th>Prey</th>
<th>Proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cephalopods</td>
<td>0.002</td>
</tr>
<tr>
<td>Small schooling fish</td>
<td>0.005</td>
</tr>
<tr>
<td>Small fish omnivore</td>
<td>0.002</td>
</tr>
<tr>
<td>Crustaceans</td>
<td>0.030</td>
</tr>
<tr>
<td>Echinoderms</td>
<td>0.030</td>
</tr>
<tr>
<td>Benthic molluscs/worms</td>
<td>0.050</td>
</tr>
<tr>
<td>Zooplankton</td>
<td>0.120</td>
</tr>
<tr>
<td>Sessile animals</td>
<td>0.095</td>
</tr>
<tr>
<td>Fish herbivore</td>
<td>0.001</td>
</tr>
<tr>
<td>Decomposer/microfauna</td>
<td>0.055</td>
</tr>
<tr>
<td>Phytoplankton</td>
<td>0.113</td>
</tr>
<tr>
<td>Benthic autotrophs</td>
<td>0.200</td>
</tr>
<tr>
<td>Detritus</td>
<td>0.297</td>
</tr>
<tr>
<td><strong>SUM</strong></td>
<td><strong>1.000</strong></td>
</tr>
</tbody>
</table>

Source: Gribble (2003)

**Prawns**

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

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

Source: Gribble (2001)

### Rock lobster

Rock lobsters are a separate functional group in the Torres Strait food web due to the important commercial value of the fishery in the area. Site specific data for rock lobsters of Torres Strait was only available for the calculation of biomass, with production and consumption rates taken from the Spiny lobster of Caribbean coral reefs (Opitz, 1996), and diet composition (Table A.13) from the lobster fisheries of the Galapagos (Martinez, 2000). Biomass calculations of rock lobsters within the study area came from Ye et al. (2004). Two populations of rock lobster were identified, the recruiting 1+ year class and the fished 2+ year class, each making up a different proportion of the entire population (Ye et al., 2004). Rock lobsters have not been separated into two-year classes in the model, therefore biomass estimates are averaged across the population. Within a 19045 km² area, the report estimates an average 7.25 million 1+ lobsters and 1.75 million 2+ lobsters.

Ye et al., (2004) surveyed rock lobsters in the western Torres Strait in May/June of 2002 collecting measurements of total weight, tail length and equivalent carapace length. Each
frequency distribution for the measurements displayed two distinct peaks that appear to represent the two age-classes. Comparing the size-frequency distribution of carapace length with a graph of spatial and temporal differences in the size of each year class confirms that the peaks are depicting age-classes. The peak of each age class in the total weight frequency distribution provided average weights which, when combined with the average number of lobsters of each age class, allowed total biomass to be calculated at 0.68 t/km².

Table A.13. Diet composition of lobsters in the Galapagos Ecopath model

<table>
<thead>
<tr>
<th>Prey</th>
<th>proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td>P.longistylus</td>
<td>0.090</td>
</tr>
<tr>
<td>Other prawns</td>
<td>0.013</td>
</tr>
<tr>
<td>P.esculentus</td>
<td>0.078</td>
</tr>
<tr>
<td>M.endeavouri</td>
<td>0.060</td>
</tr>
<tr>
<td>Echinoderms</td>
<td>0.038</td>
</tr>
<tr>
<td>Pearl shell</td>
<td>0.002</td>
</tr>
<tr>
<td>Benthic molluscs/worms</td>
<td>0.055</td>
</tr>
<tr>
<td>Sessile animals</td>
<td>0.300</td>
</tr>
<tr>
<td>Benthic autotrophs</td>
<td>0.103</td>
</tr>
<tr>
<td>Detritus</td>
<td>0.261</td>
</tr>
<tr>
<td>SUM</td>
<td>1.000</td>
</tr>
</tbody>
</table>

Source: Martinez (2000)

Cephalopods

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

<table>
<thead>
<tr>
<th>prey</th>
<th>proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cephalopods</td>
<td>0.100</td>
</tr>
<tr>
<td>Small schooling fish</td>
<td>0.065</td>
</tr>
<tr>
<td>large fish carnivore</td>
<td>0.020</td>
</tr>
<tr>
<td>Large schooling fish</td>
<td>0.030</td>
</tr>
<tr>
<td>Other prawns</td>
<td>0.010</td>
</tr>
<tr>
<td>Small fish omnivore</td>
<td>0.100</td>
</tr>
<tr>
<td>Crustaceans</td>
<td>0.090</td>
</tr>
<tr>
<td>Benthic molluscs/worms</td>
<td>0.310</td>
</tr>
<tr>
<td>Zooplankton</td>
<td>0.245</td>
</tr>
<tr>
<td>Detritus/discards</td>
<td>0.030</td>
</tr>
<tr>
<td><strong>SUM</strong></td>
<td><strong>1.000</strong></td>
</tr>
</tbody>
</table>

Source: Gribble (2003)

A.2.6. Fish

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

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

Source: Gribble (2001)

### A.2.7. Seabirds

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

### Table A.16. Diet composition of seabirds in the northern GBR Ecopath model

<table>
<thead>
<tr>
<th>prey</th>
<th>proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td>Scombrids/jacks</td>
<td>0.100</td>
</tr>
<tr>
<td>Small schooling fish</td>
<td>0.200</td>
</tr>
<tr>
<td>large fish carnivore</td>
<td>0.090</td>
</tr>
<tr>
<td>Large schooling fish</td>
<td>0.150</td>
</tr>
<tr>
<td><em>P. longistyli</em></td>
<td>0.005</td>
</tr>
<tr>
<td>Other prawns</td>
<td>0.007</td>
</tr>
<tr>
<td><em>P. esculentus</em></td>
<td>0.020</td>
</tr>
<tr>
<td>Small fish omnivore</td>
<td>0.041</td>
</tr>
<tr>
<td><em>M. endeavouri</em></td>
<td>0.030</td>
</tr>
<tr>
<td>Zooplankton</td>
<td>0.050</td>
</tr>
<tr>
<td>Fish herbivore</td>
<td>0.002</td>
</tr>
<tr>
<td>Detritus/discard</td>
<td>0.305</td>
</tr>
<tr>
<td><strong>SUM</strong></td>
<td>1.000</td>
</tr>
</tbody>
</table>

Source: Gribble (2001)

### A.2.8. Reptiles

**Sea turtles**

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

<table>
<thead>
<tr>
<th>prey</th>
<th>proportion of diet</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>P. esculentus</em></td>
<td>0.010</td>
</tr>
<tr>
<td>Crustaceans</td>
<td>0.050</td>
</tr>
<tr>
<td><em>M. endeavouri</em></td>
<td>0.055</td>
</tr>
<tr>
<td>Echinoderms</td>
<td>0.030</td>
</tr>
<tr>
<td>Benthic molluscs/worms</td>
<td>0.040</td>
</tr>
<tr>
<td>Sessile animals</td>
<td>0.010</td>
</tr>
<tr>
<td>Benthic autotrophs</td>
<td>0.805</td>
</tr>
<tr>
<td>SUM</td>
<td>1.000</td>
</tr>
</tbody>
</table>

Source: Gribble (2001)

A.2.9. Mammals

Dugong

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

Marsh et al., (2004a) carried out aerial surveys of most of Torres Strait to assess Dugong distribution and abundance, gathering information used to estimate biomass for the model. The survey broke the region up into “blocks” for assessment, each covering different areas and assessed at varying intensities. Using Arcview 3.1, the area surveyed in the ‘2B survey block’ overlapping with the area of interest in the model was calculated at 9.22 km$^2$. The entire area of interest is however 9.716 km$^2$, meaning that some of it was unsurveyed by Marsh et al., (2004a, Figure A.18).
With only 8.43% of survey block 2B surveyed; it is likely that the two dugong sighting in the overlap area may only be a fraction of what was there. The estimate of dugong abundance below may be more accurate for the overlap area.

\[2 \text{ dugong} \times 0.0843 \text{ (% of 2B surveyed)} = 23.72 \text{ dugong}\]

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

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

### A.2.10. Fisheries

**Prawn trawling**

The Torres Strait prawn fishery operates out of the eastern part of the Torres Strait and is considered the most valuable commercial fishery in the area. Commercial species of the fishery include *Metapenaeus endeavouri*, *Penaeus esculentus*, and *P. longistylus*, with the area of interest for the food web based on areas where effort for this fishery is greatest. Fishery landings for the Torres Strait food web were calculated from final catch data of 2002-2003 (Table A.18) in the area of 15 655 km² stipulated by Skewes et al., (2004) to be the eastern Torres Strait. Given that the area of interest is based on distribution of effort within the fishery, data more specific to that area is likely to be available and should replace these initial estimations.

**Table A.18. Total catch and catch in t/km²/year for commercial prawn species in the Torres Strait**

<table>
<thead>
<tr>
<th>Prawn species</th>
<th>2002-2003 final catch (t)</th>
<th>Fishery landings in eastern Torres Strait (t·km⁻²·year⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Metapenaeus endeavouri</em></td>
<td>750</td>
<td>0.048</td>
</tr>
<tr>
<td><em>Penaeus esculentus</em></td>
<td>665</td>
<td>0.043</td>
</tr>
<tr>
<td><em>Penaeus longistylus</em></td>
<td>121</td>
<td>0.008</td>
</tr>
<tr>
<td>Other prawns*</td>
<td>5</td>
<td>0.0003</td>
</tr>
</tbody>
</table>

Source: Adapted from TSPZJA (2006). Final catches have been divided by 15655 km² to obtain fishery landings in the eastern Torres Strait. *ABARE (2003)

**Finfish**

The finfish, or line, fishery of the Torres Strait is restricted to the eastern and central strait, with no line fishing allowed west of 142°30’19”E. The fishery is split into islander and non-
islander commercial fishers, with only the non-islanders required to keep logbook records. Records of reef fish caught by islanders and then purchased by non-islander or community operated freezers are used to estimate islander commercial catches in the absence of logbooks (Mapstone et al., 2003). The total median commercial catch by islanders in the late 1990’s (Table A.19) was added to the 57.1 tonnes recorded in 2000 (Mapstone et al., 2003) by non-islander commercial fishermen to obtain an estimated overall fleet catch. No specific areas of finfish fishing was stipulated, so the east Torres Strait area stated by Skewes et al., (2004) of 15 655 km² was used to calculate the fleet landing of 0.0045 t/km²/year. Net fishing also occurs in the Strait, but in small numbers and is restricted to islanders fishing in the outside but near region.

### Table A.19. Catch of finfish in the Torres Strait in the late 1990’s

<table>
<thead>
<tr>
<th>place</th>
<th>Catch (tonnes)</th>
<th>Median Catch (tonnes)</th>
</tr>
</thead>
<tbody>
<tr>
<td>York Island</td>
<td>1-2</td>
<td>1.5</td>
</tr>
<tr>
<td>Darnley Island</td>
<td>4.5-5.5</td>
<td>3.5</td>
</tr>
<tr>
<td>Murray Island</td>
<td>8-10</td>
<td>9</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>14</strong></td>
<td></td>
</tr>
</tbody>
</table>

Source: adapted from Mapstone et al. (2003)

**Rock lobster**

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

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

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

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

**Trochus**

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

Records from the prime mother ship transporting trochus out of the Torres Strait show approximately 24 t harvested in 1999, 1.8 t in 1998, 17 t in 1997 and 9.35 t in 1996, all well below the 150 t total allowable catch set for the fishery (TSPZJA, 2006; D’Silva, 2001). These records are for trochus harvesting of the entire Torres Strait Protected Zone, an area of roughly 30 000 km² that gives an estimated 0.0008 t/km²/year when the 24 t catch of 1999 is used in the calculation.

**Pearl shell**

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

**Spanish mackerel**

Operating primarily in the eastern Torres Strait, the Spanish mackerel fishery targets the narrow-barred Spanish mackerel (*Scomberomorus commerson*) and, as of 1999, target species were expanded to include school mackerel (*Scomberomorus queenslandicus*), grey mackerel (*Scomberomorus semifasciatus*), spotted mackerel (*Scomberomorus munroii*) and shark mackerel (*Grammatorcynus bicarinatus*) (TSPZJA, 2006). Small numbers of commercial operators make the majority of Spanish mackerel landings in the strait using trolls, a baited line trailed behind a slowly moving boat. Islander participation within the fishery is unknown but considered low.

Catch of Spanish mackerel made per square kilometer in a year (0.177 t/km²/year) was calculated using the 278 t reported catch of 2000-2001 (ABARE, 2003) and the 15655 km² area of the east Torres Strait Fishery (Skewes et al., 2004).

**Dugong**

Under the Torres Strait Treaty between Australia and Papua New Guinea, Torres Strait islanders are entitled to hunt dugong as part of their traditional way of life and livelihood (Marsh et al., 2004) and as such, the fishery is restricted to catching of dugong for those purposes only, no dugong meat or dugong products may be sold. In 1994 approximately 860 dugong were taken in the Torres Strait Protected Zone, with the majority of those catches occurring in the western region (TSPZJA, 2006). Number of dugongs caught was converted to total catch weight by multiplying 860 and the 0.370 t average weight of a dugong. These calculations gave a catch weight of 318.2 t, which was then divided by the 30 000 km² area of the Torres Strait Protected Zone to convert catch into tonnes per square kilometre per year (0.012 t·km⁻²·year⁻¹).

**Turtle**

Turtles are taken in all areas of the Torres Strait Protected Zone, but landings are restricted to islanders for traditional purposes. Sale of turtle meat and products are strictly prohibited and only traditional fishing methods can be used. Islanders primarily catch green turtles with little exploitation of other turtle stocks. Calculations using the average weight of a green turtle (0.1 t), total landings in a year (2600 turtles) and the 30000 km² area of the Torres Strait Protected
Zone results in 0.0087 t/km²/year of turtles caught in the Torres Strait for traditional purposes (TSPZJA, 2006).

**A.3. CONSIDERATIONS**

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

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APPENDIX B: SINGLE SPECIES BIOPHYSICAL MODELS AND PREDICTION

This Appendix describes the methods used to generate estimates for pre-trawl benthic biota maps.

B.1. ASSUMPTIONS AND CHALLENGES

The distribution of each identified species was considered separately in relation to the available physical variables used as predictors in the study. This approach assumes that the observed geospatial distribution of each species may be adequately explained by an underlying physical gradient. Each species would be expected to have its own preferred habitat range and tolerance for various physical parameters, and the spatial variation in physical variables is thought to drive the observed spatial distributions of the taxa observed.

From the point of view of biophysical modelling, it is known that sampling in a marine environment is generally very susceptible to sampling variability. That is, for fixed levels of the biophysical parameters involved, the event of observing a species, and the biomass obtained given that the species is observed, are still liable to be highly variable from random influences, neither observable nor under the control of the sampling mechanism. Thus, the modelling approach applied needs to be conservative and to anticipate that much random variation will remain unexplained. Technically, the biological response variables may be described as having a zero-inflated log-normal distribution. That is, given that some biomass is observed, the samples are approximately Gaussian on a log-scale, with a mean depending linearly on the physical predictors. However, given that the estimated probability of observing any particular species is typically relatively low (< 10%), the very large number of zeros in any species’ site-records strongly suggests a two-phase approach: initially modelling the probability of observing a species at all and conditionally modelling the distribution of the log-biomass given that the species is observed.

B.2. MODEL FORMULATION

The two-stage model used to relate the observed biomass to the underlying physical variables may be described as follows: We use a logistic regression model for the chance of observing a
species in a sampling event. For any given species, $S$, the chance of observation is

$$p_S = \Pr(S) = \frac{e^{\beta^T x}}{1 + e^{\beta^T x}} \quad \text{or equivalently} \quad \logit(p_S) = \beta^T x$$

where $x$ is a collection of suitably chosen physical variables for that species and $\beta$ is the vector of coefficients multiplying them, to be estimated from the data. This first phase model is a standard logistic regression for presence absence. The first component of $x_0$ is often a constant predictor (unity) and its coefficient $\beta_0$ is called the intercept term.

The log-biomass of the species given the fact that it has been observed in a sampling event is then modelled as a normal linear regression. In equations:

$$\log B_S | S \sim N(\gamma^T z, \sigma^2) \quad \text{i.e.} \quad \mathbb{E}[\log B_S | S] = \gamma^T z, \quad \text{Var}[\log B_S | S] = \sigma^2$$

where again $z$ is a collection of suitably chosen physical predictors for species $S$ and $z_0 = 1$ making $\gamma_0$ the intercept term.

Note that for the logistic regression stage all site records can be used in the model calibration, but for the second stage only those site records for which there is a non-zero biomass contribute information that can be used in this calibration process. This sometimes limits the modelling strategies possible.

Combining both stages of the model, the estimate of expected biomass at a given sampling site is then

$$\hat{E}_s[B_S] = \hat{p}_s \exp\hat{E}[\log B_S | S] = \frac{\exp\left(\hat{\beta}^T x\right)}{1 + \exp\left(\hat{\beta}^T x\right)} \times \exp(\hat{\gamma}^T z)$$

where the circumflex accents denote quantities estimated from sample data. Technically this is the median rather than the mean but since for the lognormal distribution the median is proportional to the mean it produces the same relative picture of biomass distribution.

Approximate standard errors for the quantities involved can be derived by the usual delta method. The formulae are complex, but as the method is entirely standard and can largely be relegated to the computational procedures, we omit them here for simplicity.

### B.2.1. Explanatory physical variables, covariates, offsets and prediction

In the formulae above, the general-purpose explanatory variable $x_j$ (or $z_j$) represents any member of the combined set of:

- 28 physical explanatory variables (e.g. percent_mud).
- 2 spatial variables (‘across’ and ‘along’) representing in relative terms the distance
between shoreline and outer reef, and between northern- and southern-most points of the GBR.

- Squared terms of physical and spatial variates (e.g. \( \text{percent}_\text{mud}^2 \))
- Second degree interaction terms between physical or temporal explanatory variables, or both (e.g. \( \text{percent}_\text{mud} \times \text{benthic}_\text{irradiance} \))
- Harmonic terms in the temporal covariates: (described specifically below)
- The measurement method used (i.e. modified trawl net / benthic sled) as a factorial predictor,
- A weighted annual average of commercial trawl fishery effort local to the sampling site.

For the purposes of generating predictions on the Torres Strait study region, we are primarily interested in the relationship between the physical variables and observed biomass. It is, however, recognised that other factors would be expected to play a role in the observed biomass. For example, many species are known to have a strong diurnal or lunar behavioural cycle, or a seasonal abundance component. Also, when generating estimates on the physical grid for full coverage of the Torres Strait region, it is desirable that the estimated distributions are independent of sampling device, which we term the ‘survey type’. These variables are included in the models in order to ensure that these effects do not interfere with estimation of the effect of genuinely physical or spatial predictors. When predictions were later made from these fitted models on to the entire Torres Strait grid (see Chapter 3), the covariate predictors were set to values that represent the sampling disposition under which the estimated biomass would be largest. This is to promote maximum contrast in the predictions.

The temporal covariates included ‘Time of Day’, ‘Moon Phase’, and ‘Time of Year’ (or ‘season’). Since GLMs depend on the explanatory variables through a single linear function, it is appropriate to represent the effect of such temporal predictors through harmonic terms, that is

\[
x_a^{(k)}(t) = \sin \left( \frac{2p \pi}{T} \right) \quad x_b^{(k)}(t) = \cos \left( \frac{2p \pi}{T} \right) \quad k = 1, 2, K
\]

where \( T \) is the appropriate fundamental period. Including temporal covariates in this way ensures that the predictions will obey the natural periodicity with respect to such predictors. In order to simplify the modelling and interpretation, we selected harmonics that corresponded to temporal periods that were expected to correspond to a biological response. These were labelled; ‘Summer/Winter’, ‘Autumn/Spring’, ‘NewMoon’, ‘SpringMoon’, ‘Day/Night’, and ‘Dawn/Dusk’. The sampling design in the Torres Strait did not permit direct estimation of temporal variables from the data at hand. Therefore, temporal coefficients were estimated from the larger and temporally more balanced survey conducted in the Great Barrier Reef region.
It is has been mentioned that ‘survey type’: whether the measurement was made with the trawl or sled, was also included as a covariate factor in the analysis. If a species was observed a minimum number of times in both sampling devices, then all data was included in the model, with the ‘Survey Type’ factor included. However, if a species was observed very little or not at all with either one of the sampling devices, then the usefulness of a model incorporating both devices became very limited. In these cases the models were based on data from the most productive sampling device only, obviating the need for a ‘Survey Type’ covariate in the model.

The area swept by each device at each site was recorded, and although efforts were made to keep this constant, it did vary somewhat. The effect of trawling (or ‘sledding’) over a greater area is to increase the probability of a catch in some monotonic way, and to increase the amount of biomass caught, on average, proportionally. Thus, the recorded swept area on the log scale was included as a candidate predictor for the logistic regression models and as a fixed offset in the biomass models. Predictions were then standardized by setting swept area to 1 Ha.

**B.2.2. Model construction and variable selection**

Each species model included, at a minimum, ‘survey type’ and ‘log-swept area’, as explanatory variables (or offset). It has also been mentioned that temporal effects, if detected in the GBR survey, were also included in the model. It has been noted above that potentially included variables included 30 physical or spatial variables, as well as squared terms and interactions between physical explanatory variables (30 + 30 × 31/2 = 495 possible candidate predictors). Not all possible choices, however, are equally reasonable a priori. In practice we note that if a physical predictor is important either for the presence/absence or the conditional biomass of a taxon then the linear term will generally make this manifest and the additional to the contribution to the model due to higher order terms involving these will be much smaller, though often quite useful, of course. We adopted a two-stage variable screening method. At the first stage we allowed selections to occur only on the linear terms. Once these were found, we considered all possible squared and interaction terms involving the identified variables for additional inclusion in the model at the second stage. In this way we ensured that if a second order term was included in the model, its marginal linear terms were also included, which is generally regarded as a desirable feature of empirical statistical modelling.

The physical variables available are by their nature highly collinear (or confounded) and to counteract this we adopted a rigorous variable inclusion policy. Such a policy also has a good chance of ensuring some interpretability for the models as well as predictive effectiveness, although this has to be taken with some caution. The criterion used in these analyses was the ‘Bayesian Information Criteria’ due to Schwartz, 1978, defined as:

\[ BIC = -2 \log L + k \log n \]
with \( \log L \) denoting the log-likelihood of the optimized model, \( k \) the number of estimated parameters used in the model, and \( n \) being the number of observations. The BIC attempts to balance model performance in the training sample with a penalized measure of model complexity to ensure that the model will capture as much signal and as little noise as possible. The related Akaike Information Criterion (AIC) penalizes complexity by a factor of \( 2 \) rather than \( \log n \), leading to more complex models which in this context, from our experience seem to sacrifice interpretability for minimal gains in predictive performance, if any.

At each iteration of the stepwise search procedure, a term is added or removed from the working model if the inclusion or removal resulted in the greatest reduction in the BIC criterion. As mentioned earlier, this was done for the linear terms initially, and then for the second degree terms involving those predictors, with the linear terms chosen fixed in the model. This was done independently for the presence/absence and the conditional biomass models for any sampled species.

### B.2.3. Prediction on the Torres Strait grid

Based on the constructed models, estimates of expected biomass were generated for each species, for each grid location in the Torres Strait study region. This involves extrapolation away from the training set and also, as noted, above, a transform back from the log scale to the natural scale. Both of these operations can result in unrealistic estimates and in this case they will be much more likely to be unrealistically large than small. As a heuristic, it was determined that the largest standardized / Ha biomass actually caught for a particular species would be used to determine a ceiling on the largest confident estimate on the grid to be used. Final estimates were arbitrarily truncated at the largest observed standardized catch rate observed in the sample itself. This seems preferable to having unrealistic estimates dominating a graphical presentation, for example.

For plotting purposes, the coefficient of variation (CV = estimated SD/estimated mean), was then calculated at each grid point. In this case the CV offers a more suitable statistic to represent uncertainty than the SE because it does so in a proportional way. We used models for log biomass, for example, for the same reasons.

### APPENDIX C: ANALYTICAL SOLUTION OF BIOMASS EQUATION

This Appendix describes the analytical solution of the biomass dynamic equation for constant...
effort, as used in the effects of trawling management scenarios evaluation.

Consider the logistic recovery equation in the absence of depletion for relative biomass $b(t)$:

$$\frac{db}{dt} = rb(t)(1-b(t)/K). \quad (C-1)$$

Given the biomass at time $t$, the biomass at later time $b(t + \Delta t)$ is given by

$$b(t + \Delta t) = \frac{b(t)e^{r\Delta t}}{1 + b(t)(e^{r\Delta t} - 1)}. \quad (C-2)$$

When a depletion term is introduced thus

$$\frac{db}{dt} = rb(t)(1-b(t)/K) - e(t)b(t), \quad (C-3)$$

an analytical update equation similar to Eq. (C2) exists for the case of constant effort rate $e(t) = e$:

$$b(t + \Delta t) = \frac{(r-\lambda e)b(t)e^{(r-\lambda e)\Delta t}}{(r-\lambda e) + rb(t)(e^{(r-\lambda e)\Delta t} - 1)}. \quad (C-4)$$

For models on an annual time step, Eq. (C-4) is used with $\Delta t = 1$ year to propagate the biomass from the start of the year to the start of the next year. The propagation is exact if effort is uniform within the year. For weekly time-step models $\Delta t = 1$ week under the assumption that effort is applied uniformly over each week. Over one year the difference between uniform effort and the same total effort applied seasonally is slight.
APPENDIX D: SOFTWARE DEVELOPMENT

This Appendix is an extract from the TS-CRC T3.3 task report *Torres Strait Management Strategy Evaluation: software architecture and implementation* by Pantus F, Ellis N, Welna A, Rochester W (2006).

Management Strategy Evaluation aims to include the natural resource management activities explicitly as a part of the system to be modelled. This adds more elements to the already complex world of natural resource modelling. It is common in the development of complex software to adopt guidelines. These guidelines must be broad enough to cater for the breadth of activities the software must perform yet restrictive enough to ensure a consistent system of software, which is easy to extend and cheap to maintain.

D.1. DEVELOPMENT PHILOSOPHY

The core activities performed by the MSE system were discussed in the previous section. This section discusses some aspects of the software architecture needed to deliver those capabilities in an orderly fashion.

As MSE software tends to get complex, the aim is to manage this complexity from the start of the project by adopting a set of guidelines to help with consistent design and implementation. These guidelines are:

- Top-down approach (architecture design first)
- Iterative implementation trajectories (width-first, thin-layer implementation)
- Functional delegation and autonomous modules
- Flexibility through standardisation: communication (interfaces)
- Tiered functionality with MSE core functions and support via specialised utilities
- Where available, use ‘off the shelf’ functionality
- Keep options open with respect to web-based user interfaces and distributed computing

Each of these guidelines is discussed briefly in terms of what it means, why it is important and how it affects the actual activities.
D.1.1. Top-down approach

Top-down approach has meaning for two activities. For the design process it means to work your way from a system-wide intent down to a detailed level of functionality. It is not uncommon with such complex systems for us to find shortly before the planned delivery that crucial functionality is missing because too much time was spent in perfecting a particularly complex detail of a specific component (such as the operating model).

The practical consequences of this guideline are that a considerable amount of time during the initial design is devoted to the discussion of the overall project deliverables and trialling the various aspects of MSE phasing, software modules, interface standards and the like.

D.1.2. Iterative implementation trajectories

For the implementation it means to first implement a ‘thin layer’ of the whole software system, followed by iterations of putting more details in. The reason for this approach is to prevent the overall system’s functionality becoming skewed by some of the complexity of the details. By implementing in such an iterative fashion, we ensure that the communication infrastructure (both standards and software), which is a critical component for a modular, tiered and flexible approach, is implemented and tested before the complexity of the MSE detail is put on top of it. The consequence for the implementation process is that considerable project time is devoted to the implementation and testing of software infra-structure.

D.1.3. Functional delegation and autonomous modules

Functional delegation means that, where possible, there is a one-to-one relationship between a major MSE task and a software module. All expertise and data that are needed to perform a task are concentrated in one place and nowhere else. For instance, there would be no ‘knowledge’ of the observation task in an operating module, only in the observer module. Such a module is relatively autonomous in that it not only knows its own task but also where to get the relevant data and instructions from, and which other modules to communicate with.

The reason for consistently delegating expertise to single modules is manifold. For instance, it makes it much easier to replace such a module with another, without any changes to the rest of the software (i.e. the software becomes flexible). Delegation also makes it easier to trace where a certain task is done, and such an intuitive structure helps in testing, debugging and maintenance. For a system like MSE software, maintenance and debugging tasks grow exponentially with the complexity of the system, and a ‘divide and conquer’ strategy is needed.
to try to combat that tendency.

The consequence of this approach is that during the design phase ‘areas of expertise’ need to be recognised within the overall MSE task. To enable each autonomous module to interact with others, appropriate standards for communication need to be defined. Strict communication protocols form the backbone of MSE development and each MSE module needs to have basic functionality that deals with communication. Descriptions of inter-module communication standards and definitions of the information that can be transferred are called ‘interfaces’. The result of such interfaces is that all software modules implementing them can communicate with all other modules.

**D.1.4. Flexibility through standardisation**

‘Flexibility’ is used here to indicate that the hard-coded relationships between autonomous modules are kept to a minimum, allowing a degree of freedom of re-arranging and changing software modules based on the MSE definition.

The need for flexibility is inherent to the MSE process itself. The core of MSE is to vary not only parameters within a rigid model but also to examine the influence of much bigger changes, even to different ways of assessing, observing or representing the system. This may result in module and communication changes by changing the MSE specification itself.

The consequence of the need for flexibility is that the current software has only a minimal amount of hard-coded ‘framework’ and is geared to adapt its structure based on external specifications which are stored in a database instead of hard-coded in the software. It is this need for flexibility that also drives the need for autonomous modules and with it, function delegation.

**D.1.5. Tiered functionality**

In a software system like the MSE system, software modules can be classed in families of functionality. Some modules perform MSE-specific tasks like the Human activity and operating modules. Others perform tasks that link between the specific MSE tasks (such as intersection of spatial closures with trawling grounds) and others perform very general tasks, potentially needed by each of the MSE modules (such as general data storage and retrieval).
Figure D.1: The TS MSE tasks need various modules to be in place to perform ‘on request’ services. These services are written as reusable software modules and shown here as boxes with rounded corners. (mapping mispelt in spatial module)

Figure D.1 shows the three families, the MSE modules at the top, the data services to interface between the MSE and the data resources and the general function modules at the bottom right.

D.1.6. Off-the-shelf functionality

Off-the-shelf functionality is a term indicating pieces of functionality encapsulated in software modules, often commercially available. For instance, functionality like map-making and spatial overlays is one of the much-needed spatial functions of an MSE system. That functionality is commercially available as a GIS module which can be embedded in the overall system.

The advantages of such software objects are that they are already autonomous pieces of
software, delivering functions that would cost many times their price if developed in-house. These objects are often well tested and are maintained by the vendor.

As they lack our specific interface definitions, they need to be embedded inside a layer of software (called a wrapper) to adapt the object’s interface to the MSE interface specification. The MSE environment uses a range of such objects and the Windows/Studio.Net environment is very well suited to making use of externally developed objects. To date, modules for GIS (MapObjects), database interactions (ADO, DAO and ADO.Net), graphs (Teechart), statistics (S-Plus and R), spreadsheet (Farpoint), Matlab, AD model builder and many more are used within the system.

D.1.7. Keeping options open for web-based user interfaces and distributed computing

A web-based user interface allows anyone with a web browser like Internet Explorer to access services offered through a single-point access to some functionality.

Distributed computing is the ability to run a computing task on several computers at the same time, thus speeding up the completion of a task. It allows scaling up the computing power by adding more computers to process a certain task. For the MSE, you may think of the possibility of running different MSE simulations on different computers in a network.

Web-based access would open up functionality like the MSE system to potential users, especially stakeholders, anywhere in the world whilst the software and possibly the data reside under the control of the development team. This prevents a heavy burden of software distribution, maintenance and client liaison. This functionality is not available in the current version of the MSE software but could be developed, at least partially, without major changes to the current software.

The need for distributed computing is very real as the MSE system is expected to create very heavy computing loads. The option of distributing this load over a network of computers needs to be kept open, and the software development environment chosen for this project, Microsoft Visual Studio.Net, already has a range of services (called ‘remoting’) that supports distributed computing.
The normal operation of the trawl scenario model is on an annual time step. The model assumes that the effort distribution is uniform within the year. However, the full-moon scenarios, in which fishing is closed during a period around full moon, and the seasonal scenario, require that effort be unevenly distributed over the year.

![Figure E.1: Time series of relative biomass for Asteroids under the 8 scenarios averaged over 20 replicates. For the weekly scenarios the biomass is drawn weekly whereas for the annual time step biomass is drawn annually and joined by straight lines. Before 2005 the effort pattern is assumed uniform throughout the year for the weekly scenarios.](image)

For the benthic impact part of the model, the difference between coarse and fine temporal detail in the effort is probably quite small, except for fauna with very rapid recovery rates. The time series in Figure E.1 show the benthic impart part of the trawl model for Asteroids. Prior to 2005, when effort is assumed uniform in the weekly time-step scenarios all scenarios give identical results, as they should. This shows that the weekly and annual results are comparable at least for biomass. In the projection period the weekly scenarios show oscillations in biomass which correspond to the seasonal pattern of effort. The oscillations are quite large because
Asteroids have a rapid recovery rate. At the beginning of each year the biomass for the Seasonal, FullMoon/EffortDisplaced, Reefs/EffortDisplaced and StatusQuo are very similar. This confirms that the temporal detail is not very important for the benthic impact model.

For target species, however, the difference from the annual model can be quite marked, because the catch rates are substantially smaller around full moon.

E.1. THE CPUE MODEL

Figure E.2 shows the seasonal CPUE data for the years 1995–2005. There is a very marked moon-phase effect, with the CPUE dropping at full moon. There is also a generally decreasing trend, possibly to do with reduced availability or simply reduced biomass of the target species as the season progresses.

Figure E.2: Daily CPUE (blue line) and fitted model (red line). The black line is the fitted model without the moon-phase component.

In order to simulate the effect of a full moon closure we need to model the reduction in CPUE. We expect the moon effect to be multiplicative, and so we fit a linear model to log(CPUE). After some exploration we found the following was suitable:
log(CPUE) = factor(year) * Polynomial₅(day of year) 
+ \sum_{n=1,...,5} \cos(2\pi \frac{\text{julian day}}{\text{synodic month}}) 
+ \sum_{n=1,...,5} \sin(2\pi \frac{\text{julian day}}{\text{synodic month}}) + \text{error} 

(1)

This form has different 5th-order polynomials in day of year for each year, and a common moon effect being a 5th-order Fourier series periodic over the synodic month (29.53059 days). We fitted the model using weighted linear regression assuming uncorrelated Gaussian error, with weight equal to the daily effort.

The fitted model is also shown in Figure E.2. The seasonal and moon effects have been largely captured. Also, the minimum of the periodic component aligns very closely with the phase of the full moon, even though this was not constrained by the model itself. There are some departures from the model in fitting the peaks and troughs, especially for 1999. This may cause some bias when we use the model.

Figure E.3 shows the corresponding effort data. Although there is some evidence of periodicity, fitting the same model to effort gives a periodic component that is about ten times smaller than for CPUE. Moreover, even at its reduced amplitude this component does not exhibit marked departures at full moon. These facts imply that closing the fishery around full moon would have a bigger impact on effort than on catch so that the overall CPUE would increase.
Figure E.3: Daily effort by year from 1998 to 2003 (source: AFMA logbooks).

Figure E.4 shows the results of the model that estimates the daily CPUE. The line that diverges most from the seasonally adjusted blue line is the results of the mode.

Figure E.4: Daily relative CPUE for 2003 with seasonal trend removed (blue), 7-day moving average (red) and periodic component of model (green). Also shown are the timings of full moon (vertical solid lines) and a week either side of full moon (dashed lines).
E.2. THE WEEKLY TIME-STEP USING 7-DAY MOVING AVERAGES

Although these CPUE data are available on a daily time scale, it is not convenient to run the trawl model at such a fine scale. However, in order to capture the effect of the full moon closure, the model cannot be run at anything coarser than a monthly time scale. We therefore make a compromise by running at a weekly time step.

In aggregating from a daily to a weekly time scale we must guard against introducing biases. Bias typically occurs by neglecting a certain edge effect at the start (and end) of the season. For instance we would compute the total annual catch by summing the catch over all temporal grid points. Each grid point represents a 7-day window centred on that point. For points occurring earlier than 3.5 days before the beginning of the season, the catch will be zero. However, for points between this time and the start of the season, there will be some catch. This ‘edge effect’ is easily overlooked in implementation, and, because catches are highest at the start of the season, it should not be neglected.

The implementation is complicated further because we wish to separate the moon effect from the CPUE effect. We define the following functions of time: \( S(t) \) is the seasonal closure function taking value 1 when the season is open, and 0 when it is closed; \( U(t) \) is the CPUE that would be observed at time \( t \) if there were no moon; \( M(t) \) is the periodic moon effect and varies from a maximum of 1 around new moon to about 0.73 just after full moon; \( E(t) \) is the potential effort rate, that is the amount that would be applied in the absence of closures.

The catch rate \( C(t) \) is then simply \( C(t) = S(t)U(t)M(t)E(t) \) and the total catch and effort over the year are:

\[
C_{\text{tot}} = \int C(t) \, dt \quad (6-1)
\]

\[
E_{\text{tot}} = \int S(t)E(t) \, dt \quad (6-2)
\]

Since we are working on a 7-day grid, we can convert these integrals to a sum

\[
C_{\text{tot}} = \sum_{n=1 \ldots 52} C_{(7)}(t_n) \Delta w, \quad (6-3)
\]

and

\[
E_{\text{tot}} = \sum_{n=1 \ldots 52} C_{(7)}(t_n)E_{(7)}(t_n)\Delta w, \quad (6-4)
\]

where \( \Delta w \) is the interval of a week, \( t_n \) is the time of the \( n \)th week, and \( C_{(7)}(t) \) is the 7-day moving average of \( C(t) \) centred at time \( t \). We can approximate \( C_{(7)}(t) \) by the product of 7-day averages of the separate components thus:
\[ C_{(7)}(t) \approx S_{(7)}(t)U_{(7)}(t)M_{(7)}(t)E_{(7)}(t) \]  

Because the separate components are uncorrelated, this is quite a good approximation. Figure E5 shows this approximation in use: the blue curve is the exact (daily) \( C(t) \) (assuming a constant \( E(t) \)), and the dots are the product of the 7-day averages. The total catch computed by this approximation agrees with the exact result to 4 significant figures.

One detail in calculating this is that \( E(t) \) is the potential effort rate, meaning its (in this case constant) value is non-zero even outside the fishing season where \( S(t) = 0 \). It would be incorrect to set \( E(t) = 0 \) where \( S(t) = 0 \) because the effect of the season would be counted twice in the averaging, leading to a downward bias in the catch. The implication is that the potential effort rate must be extrapolated beyond the season in the projection phase of the trawl model. Notice in Figure 4 that the first non-zero orange point occurs just before the season begins where the 7-day average of \( S(t) \) is somewhere between 0 and 1.

![Figure E.5: Computing the yearly catch assuming constant effort rate. Daily modelled catch (blue) and 7-day average catch (orange) for year 2000. The unaffected (by moon) catch (green) is also plotted every 7 days with consecutive points joined by straight lines. All 3 quantities are modulated by the seasonal windowing function.](image)

The approximation mentioned above is also reasonable even when the individual terms are correlated. If the closure term \( S(t) \) also includes weekly closures about full moon, then \( S(t) \) is correlated with \( M(t) \). Nevertheless, the 7-day average of products can still be replaced by the
product of the 7-day averages.

This is confirmed in Figure E.6. The graphs show different ways the yearly catch could be calculated. The sum of daily catches is regarded as the exact result. Replacing these catches with modelled catches (as described above) yields practically identical values, showing the model is unbiased (top left). This is not surprising since the model is estimated to have zero bias over all years. The top right and bottom left graphs show the effect of using the approximation on actual catch and modelled catch. There are slight biases for some years, which we consider acceptable. The biases are slightly larger for the full moon scenario, as we expected. The bottom right compares the exact values with the 7-day average modelled catch using the approximation. The biases are still acceptably small; this is the calculation used in the MSE model.

Figure E.6: Comparison of yearly catches calculated by different methods: (top left) actual catch (computed by summing daily catches) vs modelled catch (computed by summing daily modelled catches); (top right) actual catch vs 7-day average actual catch (computed by summing 7-day averages using the approximation); (bottom left) as (top right) but using modelled catch instead of actual catch; (bottom right) actual catch vs modelled 7-day average catch using the approximation, the calculation used in the MSE model code.
E.3. THE MSE CATCH CALCULATION

During the historical phase both effort and CPUE are obtained from logbook data. However, in the projection phase these data are not available; they have to be imputed somehow. We do not attempt to model the detailed processes leading to the spatio-temporal patterns of effort over a season. Instead we use the entire season from a particular historical year $y^*$ as a prototype for the pattern in the projection year $y$. This prototype is adjusted depending on the management actions in place for year $y$. For each projection year a prototype year is chosen at random from a range of historical years considered as representative of future patterns. Since CPUE is probably correlated with effort (through targetting), we also use the CPUE pattern from the same year $y^*$.

In projection year $y$ the total effort depends on the management action in place. If the effort cap is $E_{\text{cap}}$ then we set the actual allocated effort to $E_{\text{cap}} \varepsilon_y$, where $\varepsilon_y$ is a lognormal variate with mean 1 and coefficient of variation $\sigma$. Currently $\sigma$ is taken as the CV of total effort for the historical prototype sample years. The allocated effort $E_{(7)}(y, w)$ in week $w$ of projection year $y$ is then simply $E_{(7)}(y, w) = \lambda_y S_{(7)}(y, w)E_{(7)}(y^*, w)$ where the scale factor $\lambda_y$ satisfies $\lambda_y = E_{\text{cap}} \varepsilon_y / \sum_{n=1}^{52} S_{(7)}(y, w)E_{(7)}(y^*, w) \Delta w$. Here $S_{(7)}(y, w)$ comes from the management model and $E_{(7)}(y^*, w)$ is the 7-day average of the (extrapolated) effort rate.

This calculation of the scale factor assumes latent resources in the fishery allowing displacement of effort from full moon closures to other times of the season. If this is not the case, then effort that would have occurred during full moon is lost. This means that the scale factor in (7) must be adjusted downwards. To do this, let us first decompose $S$ into a season (on-off) component $S_1$ and a full moon component $S_2$, so that $S = S_1 S_2$. Then the factor $S_{(7)}(y, w)$ in (7) must be replaced by $S_{1(7)}(y, w)$, the 7-day average of the season-only component. The equation (6) remains unchanged. In the MSE model we actually use a simpler approach, which is to find how much effort was displaced by the displacement scenario and rerun the simulation with the effort cap reduced by this amount. Finally, turning to the catch equation we have

$$C_{(7)}(y, w) \approx U_{(7)}(y^*, w)M_{(7)}(y, w)E_{(7)}(y, w). \quad (6-6)$$

Here $M_{(7)}(y, w)$ is the predictable part of the CPUE model and $U_{(7)}(y^*, w)$ is the 7-day average of the (extrapolated) modelled CPUE without the moon effect from historical year $y^*$; this corresponds to the green line in Figure A.8. (Simple nearest neighbour extrapolation of the underlying daily $U(t)$ beyond the season is sufficient.) The management closure $S$ is subsumed in $E$, as per equation (6).

The current implementation of the MSE model uses spatial effort patterns at weekly aggregation. The weekly pattern is scaled to agree with the allocated value $E_{(7)}(y, w)$. On the other hand the CPUE pattern is not spatially explicit; the catch during a week is determined by...
the CPUE for that week and the total effort for that week (as per equation (8)), but does not
depend on the spatial distribution of effort for that week.

E.4. EFFORT ALLOCATION IN THE MSE MODEL

We can describe effort allocation as a two-stage process. In the first stage we perform temporal
allocation by computing the weekly effort $E_{yw}^\text{alloc}$:

$$E_{yw}^\text{alloc} = E_y^\text{cap} S_{yw} \left(E_{yw}^*\right)^\delta / \sum_{w'} S_{yw'} \left(E_{yw'}^*\right)^\delta. \quad \text{(temporal, weekly)}$$

This is a simplification of what was stated in the previous section. For annual effort (i.e.
effort is not defined at weekly resolution) the temporal allocation is simply

$$E_y^\text{alloc} = E_y^\text{cap} E_y. \quad \text{(temporal, annual)}$$

The second stage is spatial allocation. For weekly allocation the spatial closure may vary
with week. The allocated effort to each grid $g$ for each week $w$ is

$$E_{ywg}^\text{alloc} = E_{yw}^\text{alloc} S_{ywg} \left(E_{ywg}^*\right)^\delta / \sum_{g'} S_{ywg} \left(E_{ywg'}^*\right)^\delta. \quad \text{(spatial, weekly)}$$

For annual allocation the equation is similar

$$E_{yg}^\text{alloc} = E_y^\text{alloc} S_g \left(E_{yg}^*\right)^\delta / \sum_{g'} S_{g'} \left(E_{yg'}^*\right)^\delta. \quad \text{(spatial, annual)}$$

Here $\delta$ is an adjustable parameter to control the degree of aggregation, which we have set
to 1 for this study.

These models assume effort is totally displaceable, so that any effort that would have been
expended in one place or at one time could be expended at all other places or all other times.
One way this assumption could fail is if the effort exceeded the maximum possible effort during
some time interval by having more boat days than there are boats in the fishery. This might
happen in the first weeks of the fishery when the effort is at a maximum.

This constraint could be accommodated in the reallocation formula like so

$$E_{yw}^\text{alloc} = \min \left(E_{yw}^{\text{max}}, \lambda_y S_{yw} \left(E_{yw}^*\right)^\gamma \right), \quad \text{(constrained)}$$

with a parameter $\lambda_y$ to be found. This formula says that allocated effort is proportional to a
power of the undisplaced effort except where it would exceed the capping value $E_{yw}^{\text{max}}$, e.g. the
number of boats. One would have to use non-linear root searching methods to find $\lambda_y$. One
could imagine spatial versions of this constrained allocation that would depend on the maximum
number of boats that could simultaneously fish in a grid cell.