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Evaluating ecosystem-based management options: Effects of trawling in Torres Strait, Australia

Nick Ellis [a,*], Francis Pantus [a], Andrzej Welna [a], Alan Butler [b]

[a] CSIRO Marine and Atmospheric Research, 233 Middle Street, Cleveland, Qld. 4163, Australia
[b] CSIRO Marine and Atmospheric Research, Castray Esplanade, GPO Box 1538, Hobart, Tasmania 7001, Australia

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ABSTRACT

A suite of management options for a prawn trawl fishery in Torres Strait, Australia was assessed for impacts on the benthic fauna using a dynamic management strategy evaluation approach. The specification of the management options was gained through consultation with stakeholders. Data for the model was drawn from several sources: the fleet data from fishery logbooks and satellite vessel monitoring systems, benthic depletion rates from trawl-down experiments, benthic recovery rates from post-experiment recovery monitoring studies, and benthic distribution from large-scale benthic surveys. Although there were large uncertainties in the resulting indicators, robust measures relevant to management were obtained by taking ratios relative to the status quo. The management control with the biggest effect was total effort; reducing trawl effort always led to increases in benthic faunal density of up to 10%. Spatial closures had a smaller benefit of up to 2%. The effect of closing a set of buffer zones around reefs to trawling was indistinguishable from the status quo option. Closing a larger area, however, was largely beneficial especially for sea cucumbers. When the spatial distributions of fauna prior to fishing were accounted for, fauna with distributions positively correlated with effort improved relative to those negatively correlated. The reduction in prawn catch under effort reduction scenarios could be ameliorated by introducing temporal closures over the full-moon period.

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1. Introduction

Commercial trawling is a large-scale activity conducted over most of the world's continental shelves at varying levels of intensity (Watling and Norse, 1998). The activity poses a threat to the sea-bed biota, both directly, by removal of biomass (e.g., Ellis and Pantus, 2001), and indirectly, by likely habitat destruction and possible domino effects on the ecological relationships and food webs (Pauly et al., 1998) by the selective removal of some benthic species.

The direct effect of trawling on the benthos has been demonstrated using experimental trawls (Van Dolah et al., 1991; Lindeboom and de Groot, 1998; Poiner et al., 1998; Bergman and van Santbrink, 2000; Burridge et al., 2003). Trawling on an industrial scale leads to large-scale impacts, which, though often difficult to detect (Poiner et al., 1998; Lindegarth et al., 2000; Kenchington et al., 2001; Drabsch et al., 2001), may lead to long-term depletion of vulnerable species (Bergman and Hup, 1992), depletion of invertebrates generally (Philippart, 1998; Engel and Kvitek, 1998) and reduction of biodiversity (Thrush et al., 1998; McConnaughey et al., 2000). Collie et al. (2000), using a meta-analysis of dozens of studies, have shown that the large-scale effects of trawling are complex and depend on many factors including habitat, benthic composition, trawl intensity and trawl gear types. See Jennings and Kaiser (1998) for a review of the extensive literature. Recently, Hiddink et al. (2006a) have shown using a size-based competitive biomass model (Duplisea et al., 2002) that trawling in the North Sea has substantially reduced both biomass and production.

The trawling industry is now under pressure from community groups, local legislation and international certification, to ensure the conservation of the ecosystem as a whole. Furthermore, there is now increasing recognition from both the fishing industry and the fisheries managers that sustainable harvesting requires the preservation of both the target stocks and the ecosystem in which these resources are embedded. There is therefore a need for managers to understand the effects of their management actions on both the target stocks and the benthos.

The framework used to structure the evaluation of the various trawl management options is known as management strategy evaluation or MSE (Smith, 1994; Sainsbury et al., 2000). MSE is both an approach and a set of tools to support decision making under uncertainty for complex natural resource and environmental management systems. It involves assessing the

* Corresponding author. Tel.: +61 7 38267260.
E-mail address: nick.ellis@csiro.au (N. Ellis).

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consequences of a range of management strategies or options and presenting the results in a way that lays bare the tradeoffs in performance across a range of management objectives.

MSE studies have been reported by several authors. Gribble (2003) used a 25-component ecosystem model to simulate an effort reduction scenario in the trawl and line fishery in the Great Barrier Reef. Fujioka (2006) assessed the changes in affected habitat over a range of recovery rates under various spatial closure scenarios for the Bering Sea trawl fishery. Hiddink et al. (2006b) using their size-based competitive biomass model assessed the impact of combinations of closures and effort restrictions for the cod and plaice fisheries in the North Sea. And Hiddink et al. (2007) applied the same model to spatial closures based on habitat sensitivity.

Torres Strait, between Cape York in North-Eastern Australia and Papua New Guinea, supports a range of fisheries, notably prawn trawling, managed by the Australian Fisheries Management Authority (AFMA). The Torres Strait Regional Authority (TSRA) has a mandate to protect the marine ecosystem in the region.

The Torres Strait Cooperative Research Centre of which the Commonwealth Scientific and Industrial Research Organisation (CSIRO) is a partner was tasked to examine the effects of prawn trawling under alternative management scenarios in the region in collaboration with the TSRA and AFMA. In 2004, representatives from TSRA, AFMA, CSIRO and the people of Masig, an island in the northern parts of Torres Strait, drew up a set of management options to be evaluated using effects-of-trawling models to be developed by CSIRO. Masig is used by the trawl fleet as a logistical hub in the region and as such is intimately connected to the trawl fishery. The proposed options are outlined in the next section, followed by an account of the model developed, and its use to evaluate them.

2. Methods

MSE emulates the key processes of an adaptive management system (Holling, 1978) and a diagram with its main activities is presented in Fig. 1. Management strategies (or rules) describe what actions to take given a result of an assessment. The dashed line in Fig. 1 indicates that explicit management strategies may or may not be available. In the case that management strategies are available, management actions can adapt dynamically to the changes in the managed system through the results of an assessment. In case such management rules are not available, a fixed set of sequential management actions has to take their place. A fixed set of management actions is referred to as a management scenario. In the Torres Strait effects of trawling MSE described in this paper, several of such scenarios were defined by the stakeholders. A detailed description of the MSE process can be found in Pantus et al. (2007).

The following section describes the management scenarios that were used to evaluate the effects of trawling on benthic biota (flora and fauna) for the Torres Strait prawn trawl fishery.

3. Management scenarios

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

The stakeholder group proposed four alternative options for evaluation:

1. *Reef buffer*: establish buffer zones that excluded trawling around reefs and islands.
2. *Moon closure*: close the whole fishery 4 days either side of full moon.
3. *Effort reduction*: reduce total allowable effort (TAE) from 9000 to 5000 boat days in 3 linear annual steps of 1333 days.
4. *Masig closure*: close an area of 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 0.5, 1 and 2 km.

The *moon closure* option entails closing the prawn fishery around the time of full moon. The fact that lunar phase influences catch has been known for a long time for fish species (e.g., Savage and Hodgson, 1934) and more recently for prawn species (e.g., Vance et al., 1994; Wassenberg and Hill, 1994; Courtney et al., 1996; Griffiths, 1999). In Torres Strait, fishers know that prawns are much harder to catch and of lower quality (molting) around full moon. Therefore, a management strategy that is sometimes suggested is to close the fishery around full moon in order to increase the overall productivity. This 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 times the fishery was open 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 quo* option was used.

The status quo was defined to be the effort distribution and spatial and temporal closures as they existed in Torres Strait in 2003 (Kung et al., 2004).

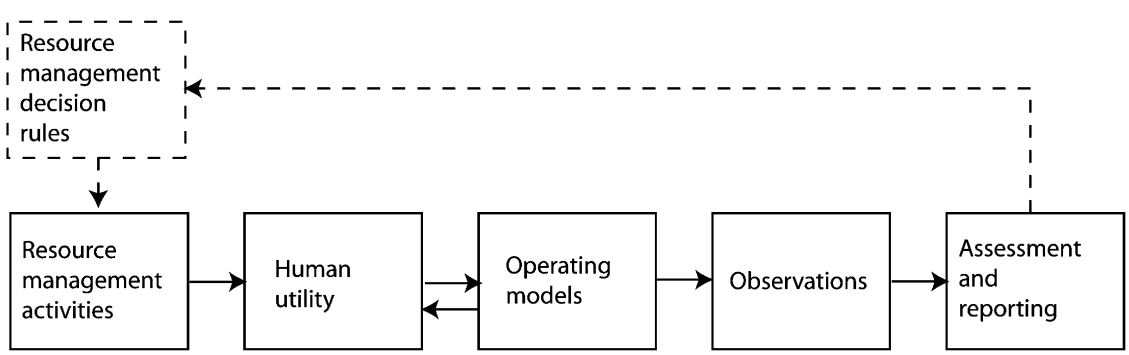

Fig. 1. Overview of TS MSE system components and the major information flows. The dashed line indicates the adaptive management extension (Pantus et al., 2006).

4. Operating models

There are ranges of models and data sources used to evaluate the four management scenarios and an overview is given in Fig. 2.

In the centre of Fig. 2, 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_t at time t, some recovery rate r, the pre-trawl biomass B_0, the trawl effort E and some large-scale depletion rate λ:

$$B_{t+\delta} = B_t + \left(rB_t(1 - B_t/B_0) - B_t \times E \times \lambda\right)\delta. \quad (1)$$

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. (1) by stepping through time, we integrate the corresponding differential equation (in the limit $\delta \to 0$) to obtain an analytic expression for the biomass after an arbitrary interval (e.g., after a week or a year).

The modelling process is spatially explicit and restricted to a geographic region of interest, the Torres Strait in this case, which is divided into rectangular one-nautical mile cells. The model in Eq. (1) is applied to each of those cells.

The effects of trawling model is also applied to a set of benthic groups (generally higher taxa—e.g., families, classes or phyla) defined within the spatial grid on which trawling effort is applied both over the historical period of the fishery (starting in 1980) and over a projection period (until 2020). The main task of the model is to compute the change in benthic biomass for each of the benthic groups in each cell over time. Each cell is self-contained and does not interact with any other cell. The model also produces catch, which is based on an empirical catch per unit effort (CPUE) relationship.

In the historical period, all scenarios should give identical results, because they represent the results of the management that already has been supplied to the fishery. In the projection period, however, the simulations may differ because the management options differ. For scenarios 1, 3 and 4, the projections are continued on an annual time step; scenario 2, however, is run on a weekly time step in order that the temporal effects of the moon closure can be resolved.

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 the temporal CPUE pattern (especially its decline over the season) is related to the temporal effort pattern, we also use the CPUE pattern from the same year y^*.

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

$$E_y^{\text{alloc}} = E_y^{\text{cap}} \varepsilon_y. \quad (2)$$

Here E^{cap} is the effort cap set by management and ε_y is a lognormal variate with mean 1 and coefficient of variation σ. This random term represents the implementation error that is, the ratio of actual effort to desired effort.

Next, for options on the annual time step, the effort is allocated to each spatial cell g in proportion to the spatial effort E_{y^*g} for the representative year y^*:

$$E_{yg}^{\text{alloc}} = \frac{E_y^{\text{alloc}} S_g E_{y^*g}}{\sum_{g'} S_{g'} E_{y^*g'}}, \quad (3)$$

where S_g is the spatial closure term (options 1 and 4) taking value 1 if the cell is open and 0 if the cell is closed.

For options on the weekly time step (option 2), we first allocate to weekly effort and then distribute that effort spatially. For week

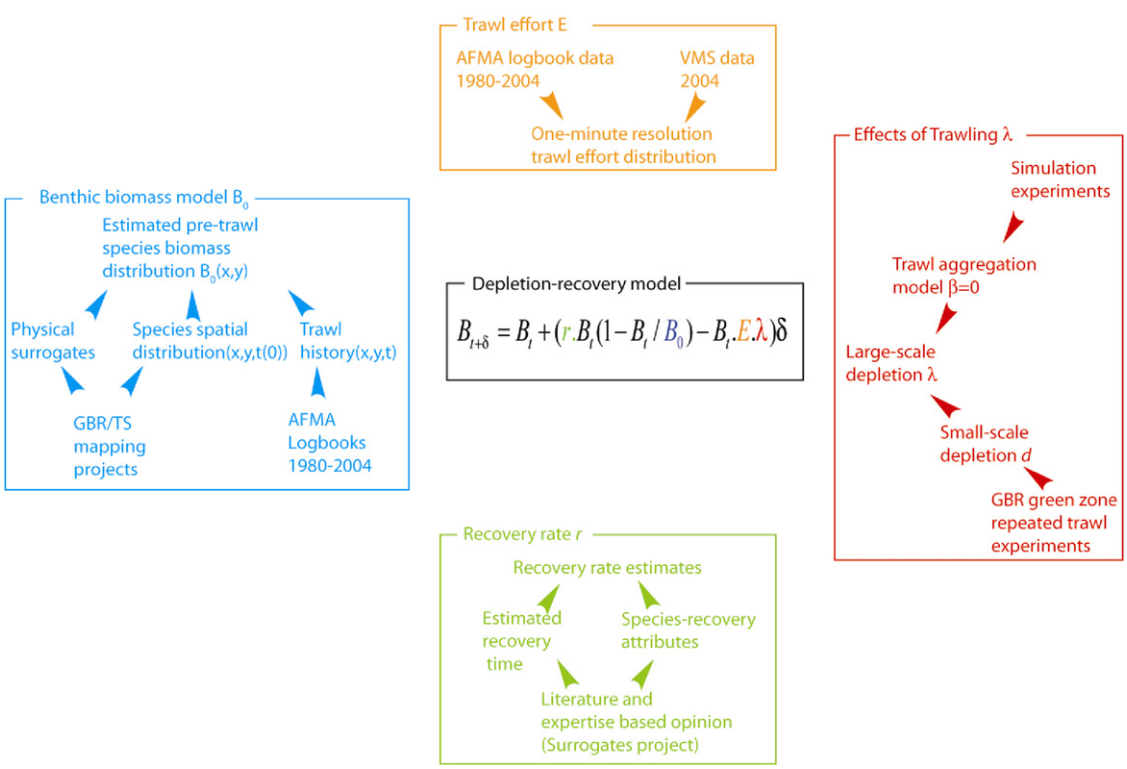

Fig. 2. The effects of trawling model is based on the biomass depletion and recovery of species that are affected by trawling. Its parameters are derived from the four areas indicated in the diagram (Pantus et al., 2007).

w, the allocated effort is proportional to the weekly effort E_{y^*w} for the representative year y^*:

$$E_{yw}^{\text{alloc}} = \frac{E_y^{\text{cap}} \varepsilon_y S_{yw} E_{y^*w}}{\sum_{w'} S_{yw'} E_{y^*w'}}, \tag{4}$$

where S_{yw} 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 it takes a value between 0 and 1.

The spatial allocation is 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^*:

$$E_{ywg}^{\text{alloc}} = \frac{E_{yw}^{\text{alloc}} E_{y^*wg}}{\sum_{g'} E_{y^*wg'}}. \tag{5}$$

We did not use spatial closures for option 2, although such closures could be implemented in a similar way to Eqs. (3) and (4).

4.1. Catch

For all historical effort, we also had CPUE U at the same temporal resolution. For options 1, 3 and 4, we simply calculated annual catch C_y as

$$C_y = U_{y^*} E_y^{\text{alloc}}. \tag{6}$$

For option 2, we calculated weekly catch C_{yw} as

$$C_{yw} = T_{y^*w} M_{yw} E_{yw}^{\text{alloc}}, \tag{7}$$

where T_{y^*w} is the seasonal CPUE trend for week w in year y^*, and M_{yw} is the multiplicative moon effect on CPUE for week w in year y (normalised to have maximum 1). The log trend was modelled as separate 5th-degree polynomials in day for each year; and the log moon effect was modelled as a Fourier series up to the 5th harmonic with fundamental period equal to the synodic month (29.53059 days). The model was fit to daily CPUE data over 1995–2005 by weighted linear regression with weights proportional to effort. Fig. 3 shows the daily CPUE data.

As well as a general decline over the season, there is a definite monthly oscillation with a quite sharp trough near full moon. This can be seen more clearly by dividing the CPUE by the seasonal

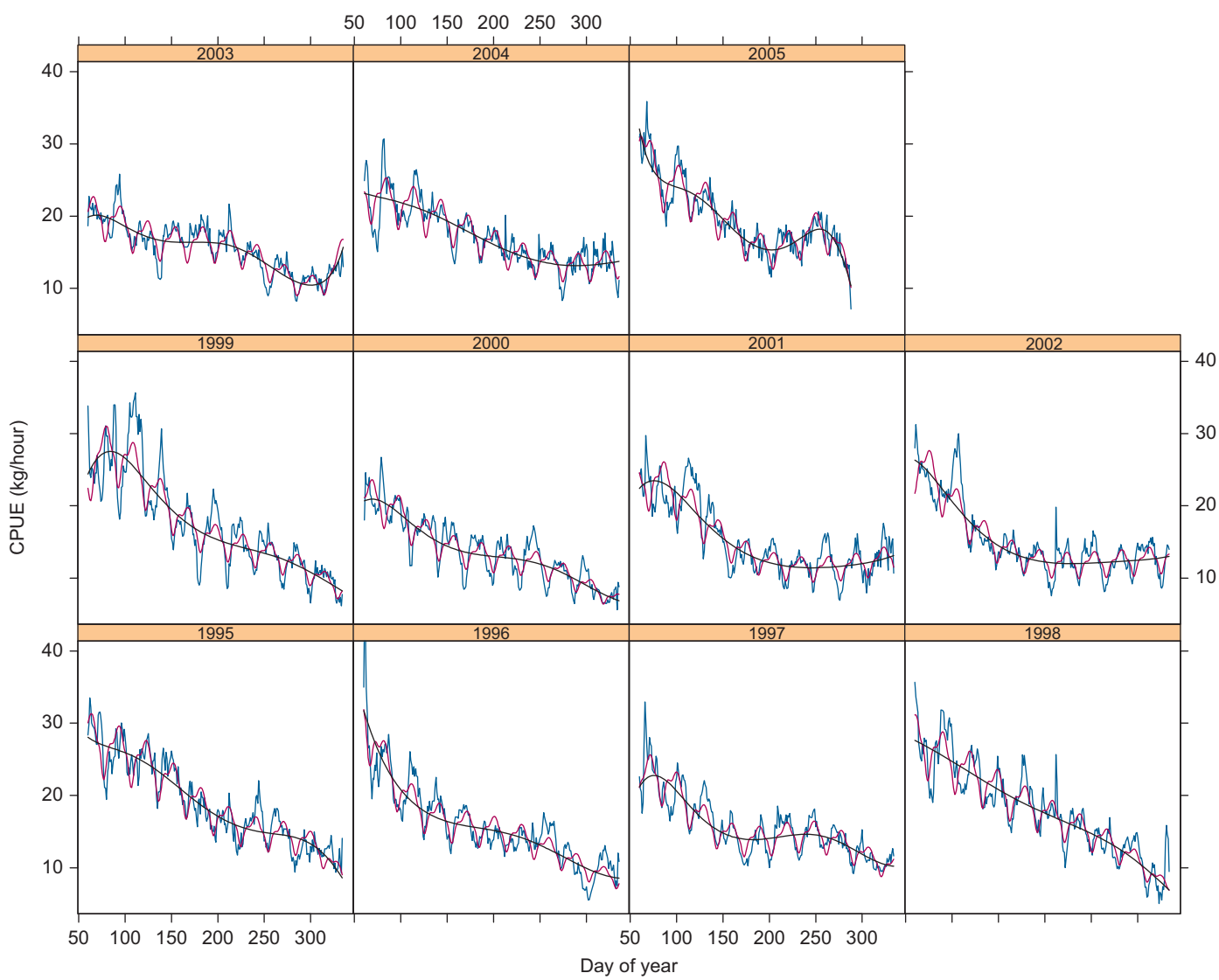

Fig. 3. Daily CPUE (*blue line*) and fitted model (*red line*). The black line is the fitted model without the moon-phase component.

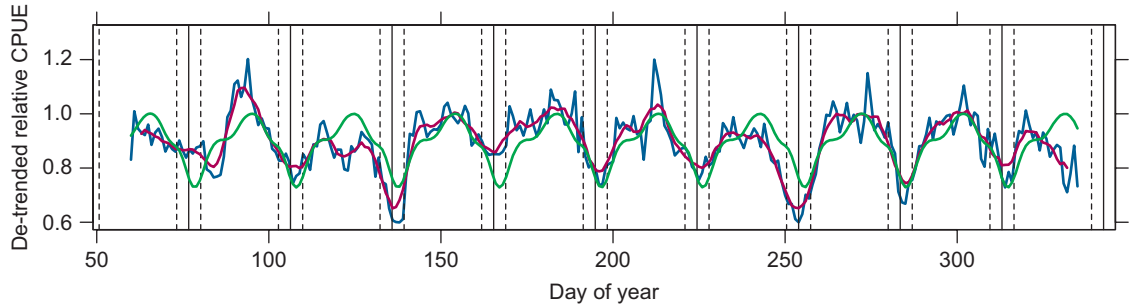

Fig. 4. Daily de-trended relative CPUE ($C_{20,003}/T_{2003}$) for 2003 with seasonal trend removed (*blue*), 7-day moving average (*red*) and periodic component of model M_{2003} (*green*). M_{2003} is normalised to have maximum 1. Also shown are the timings of full moon (*vertical solid lines*) and 4 days either side of full moon (*dashed lines*).

trend T (where $T \times M$ is the model fit with M normalised to have maximum 1) as shown for 2003 in Fig. 4. The residual moon effect (*blue line*) is modelled by the periodic component M (*green line*). The trough occurs just after full moon.

4.2. Benthic impact

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

$$\frac{dB_{sg}}{dt} = r_s B_{sg}(t)\big(1 - B_{sg}(t)/K_{sg}\big) - d_s E_g(t) B_{sg}(t), \tag{8}$$

where $B_{sg}(t)$ is the biomass at time t of benthic species (or taxon) s in cell g, K_{sg} is the carrying capacity of species s in cell g, $E_g(t)$ is the effort rate at time t in units of relative swept area (i.e., proportion of cell area swept) per unit time, d_s is the relative depletion rate per tow of species s, and r_s is the relative recovery rate of species s. (Note that r_s has units of time^{-1} and d_s is dimensionless.) Here the large-scale rate λ (Eq. 1) is equal to the depletion rate per tow d_s under the assumption of random sub-cell trawling (Ellis and Pantus, 2001). The equation simplifies to

$$\frac{db_{sg}}{dt} = r_s b_{sg}(t)(1 - b_{sg}(t)) - d_s E_g(t) b_{sg}(t)$$
$$b_{sg} = \frac{B_{sg}}{K_{sg}}, \tag{9}$$

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

An important quantity is the ratio r_s/d_s. In a hypothetical cell with constant annual effort (coverage) C, the long-term relative biomass will approach the value max $(0, 1-C/(r_s/d_s))$. The value of C that just drives the biomass to local extinction is r_s/d_s: we call this quantity the *extinction coverage*. The higher the extinction coverage the more resilient the benthic species. The results of simulations tend to depend on r_s and d_s principally through the ratio r_s/d_s.

Inter-cell interaction is partially accounted for through the pre-fishery biomass (where global trends are explicitly modelled) and the effort distribution (which is taken from actual spatially correlated distributions). However, the relative biomass component assumes that the dynamic equation within each cell is independent of its neighbours. This assumption has at least two implications. First, the way the model allows biomass to attenuate indefinitely in cells with high effort might be artificial because the potential for recruitment from outside the cell is ignored. This would imply that r needs to be increased. Second, the consequence of allowing recruitment from outside the cell is to deplete effectively the neighbouring cells. This is equivalent to having a less aggregated trawling pattern which is known to lead to an increase in the effective depletion rate (see Ellis and Pantus, 2001). These two effects, increasing r and increasing d, tend to oppose each other. However, it is difficult to know the effect of ignoring neighbour interaction without actually simulating such interaction.

Fujioka (2006) used a dynamic equation very similar to Eq. (9) with relative biomass replaced by proportion of unaffected habitat H as the dependent variable, and rb in the recovery term replaced by an absolute recovery rate ρ (see his Eq. (A.3)). In a hypothetical cell with constant annual effort (coverage) C, the long-term value of H approaches the value $1-C/(C+\rho/d)$, which, when ρ is large or C or d are small, is very similar to the behaviour of b in our model. This is to be expected because under these conditions, b stays close to 1 so that the term rb is close to constant, as in Fujioka's form.

The two models differ when the dependent variable is close to zero. In our model, the recovery is small because of the rb term, whereas in Fujioka's model the recovery is greatest when $H = 0$. Unlike in our model, where species can be trawled to local extinction, Fujioka's model allows habitats recover to some extent, and there is no corresponding extinction coverage. There may be species for which it is appropriate to use Fujioka's model instead of the logistic recovery model; for example, a weedlike, invasive species that prefers an empty habitat.

Duplisea et al. (2002) developed a refinement to the biomass dynamic Eq. (8) that reduced the carrying capacity of a species by an amount proportional to the biomass of its competitors. They used a parsimonous scheme to reduce the number of parameters needed to specify the inter-species interaction. The model was used by Hiddink et al. (2006a, b), (2007) in the North Sea trawl fishery. They also reported on productivity and species richness as well as biomass. Such models are intermediate in complexity between our model and a full ecosystem model.

4.3. Handling of 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 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 neighbouring cells.
4. The aggregation of effort within a grid cell.
5. The spatial pattern of effort among grid cells.
6. The temporal pattern of effort within season.
7. The variation of total effort 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 one-nautical mile scale is assumed random from other studies (Rijnsdorp et al., 1998). With depletion per tow (item 1), although the model does not explicitly allow variable depletion, the parameter d 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 β,
2. depletion rate d,
3. recovery rate r,
4. pristine biomass B_0.

Uncertainty in β is probably of lesser importance because the effect of β is an adjustment to d ($\lambda = \log(1+\beta d)/\beta$ from Ellis and Pantus (2001)). 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 β. The depletion rates reported by Burridge et al. (2003) are reasonably well estimated. However, the recovery rate is very uncertain (standard errors are typically around the same size as the mean (C.R. Pitcher, personal communication)) and results from Pitcher et al. (2007a) suggest even wider variation than used here, with r ranging from 0 to 4 (see their Figs. 2–45). For B_0, the generalised linear models do provide estimates of uncertainty: typically the standard error is the same order of magnitude as the mean.

A sensible way to address the uncertainty in parameters is to carry out sensitivity analysis that is, to look at the variation in outputs over a feasible range of parameter settings. We shall see in the results that the relative effect of trawling among scenarios is most strongly determined by the overall effort. The dependence on spatial distributions is smaller, though not negligible. The absolute effect is also strongly sensitive to the extinction coverage r/d; however, this sensitivity is much weaker for *relative* effects, being similar in magnitude to the spatial dependence. Because the benthic groups in this study cover a reasonable range of (r, d) values, the results for those groups provide a sensitivity analysis that can be applied to a wider selection of biota.

5. Data

Data for the model was drawn from several sources: the fleet data from fishery logbooks and satellite vessel monitoring systems (VMS), the benthic depletion rates from trawl-down experiments (Poiner et al., 1998), the benthic recovery rates from the categorical method of Hill et al. (2002) and benthic distribution from large-scale benthic surveys (Pitcher et al., 2007a, b). Fig. 2 gives an overview of how the various data fit into the model.

5.1. Fleet data

Logbook data have been collected in the Torres Strait fishery from 1980 until 2002. In the later years, trawl effort was reported daily by each vessel at the resolution of 6 min cells. In 2003, a satellite 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 available trawl effort to 1 min spatial cells and to annual or weekly temporal bins, depending on the management option. For historical effort available at 6 min resolution, we subdivided the effort into 1 min cells proportionally to the average VMS pattern within each 6 min cell.

5.2. Depletion and recovery rates

We concentrated on broad taxonomic groupings in this paper. The depletion rates were those reported by Poiner et al. (1998) and Burridge et al. (2003) with recovery rates obtained using the categorical method of Hill et al. (2002). We also used these model results to further assess seven species for which pre-trawling biomass models were available (Pitcher et al., 2007b). We used the trawl model results whose (r_s, d_s) values were closest to the (r_s, d_s) values reported for these species in Pitcher et al (2007a, Tables 2–15 and 2–16). Table 1 summarises the taxonomic units and their (r_s, d_s) values.

Table 1
Recovery rate, trawl depletion rate and extinction coverage

Benthic group	Recovery rate (year^{-1})	Depletion rate (tow^{-1})	Extinction coverage
Gastropoda	0.41	0.20 (0.03)	2.1
Echinoidea	0.40	0.14 (0.09)	2.9
Crustacea	0.52	0.13 (0.05)	4.0
Alcyonaceae	0.40	0.09 (0.07)	4.4
Bryozoa	0.40	0.09 (0.06)	4.4
Holothuroidea	0.56	0.11 (0.11)	5.1
Bivalvia	0.52	0.09 (0.08)	5.8
Porifera	0.71	0.12 (0.04)	5.9
Ophiuroidea	0.63	0.09 (0.06)	7.0
Crinoidea	0.56	0.08 (0.06)	7.0
Hydrozoa	0.56	0.08 (0.06)	7.0
Asteroidea	0.97	0.10 (0.07)	9.7
Species			
Alertigorgia orientalis	0.97	0.10 (0.07)	9.7
Carijoa sp.1	0.97	0.10 (0.07)	9.7
Dendronephthya spp.	0.71	0.15 (0.05)	4.7
Dichotella sp.1	0.71	0.12 (0.04)	5.9
Ianthella flabelliformis	0.41	0.20 (0.03)	2.1
Ianthella quadrangulata	0.56	0.11 (0.11)	5.1
Subergorgia suberosa	0.41	0.20 (0.03)	2.1

Depletion rates d and standard errors are from Burridge et al. (2003). Recovery rates r (no standard errors available) are from Hill et al. (2002). Extinction coverage r/d is the number of times per year a patch must be trawled to make the benthic group go locally extinct. The benthic groups are ordered by extinction coverage.

5.3. Pre-trawling biomass

A large-scale benthic survey was carried out over the Torres Strait region (Pitcher et al., 2007b). A total of about 400 sites were visited according to a stratified design. At each site, a benthic sled was deployed and at around 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 (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 covariables). Each OTU was fit to a statistical model that was a product of two generalised 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. 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 covariable settings. For instance, some species might be more catchable at night, but were surveyed at some sites in daytime and other sites at night. The predicted density would then use the nighttime setting for this species. This predicted density is actually relative to the unknown catchability, which is assumed to be constant.

For the purposes of this paper, these models can also be used to predict the *pre-trawling* 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.

6. Results and discussion

A key output of an MSE is a decision support table. The decision support table brings out the tradeoffs between various management scenarios in terms of performance measures. The performance measures that were assessed were impacts on the benthic biomass and impacts on the target catch.

Table 2 shows the decision table for all eight scenarios and the relevant performance measures. 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. For instance, the performance measures for CPUE can be averaged over all years or analysed for patterns across years; and the impacts on the benthos can be compared among species, or spatially or temporally.

The first five scenarios in the decision table are on the annual time step and the last three 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 *StatusQuo*. It is against this scenario that all others should be compared. 'Status Quo, Seasonal' (subsequently referred to as *Seasonal*) is the equivalent of *StatusQuo* 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.

6.1. Catch performance measures

The full-moon closures are expected to 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 *Seasonal* scenario being a little less than 3%. Fig. 5 shows how this works for the year 2005 for the *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 (triangles). 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 thus 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.

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 1 week, we should expect the CPUE to be higher in

Table 2
Performance measures ± S.D. for each scenario

Scenario	Performance measure			
	Total effort ('000 h)	Total catch (tonnes)	CPUE (kg h^{-1})	Mean species biomass (%)
Status quo, annual	100 ± 12	1748 ± 246	17.48 ± 1.60	54.2 ± 0.8
Reef buffers, effort displaced	100 ± 11	1742 ± 267	17.42 ± 1.61	54.4 ± 0.9
Masig closure, effort displaced	100 ± 13	1785 ± 300	17.85 ± 1.68	54.8 ± 1.1
Masig closure, effort removed	84 ± 10	1502 ± 234	17.88 ± 1.59	58.1 ± 1.0
Effort reduction	55 ± 7	980 ± 160	17.62 ± 1.74	65.9 ± 1.1
Status quo, seasonal	99 ± 20	1728 ± 391	17.41 ± 2.39	54.6 ± 1.0
Full-moon closure, effort displaced	99 ± 19	1774 ± 403	17.91 ± 2.45	54.7 ± 1.3
Full-moon closure, effort removed	73 ± 14	1311 ± 295	17.97 ± 2.45	61.2 ± 0.9

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 12 benthic biota groups. The standard deviations are over 20 replicate simulations.

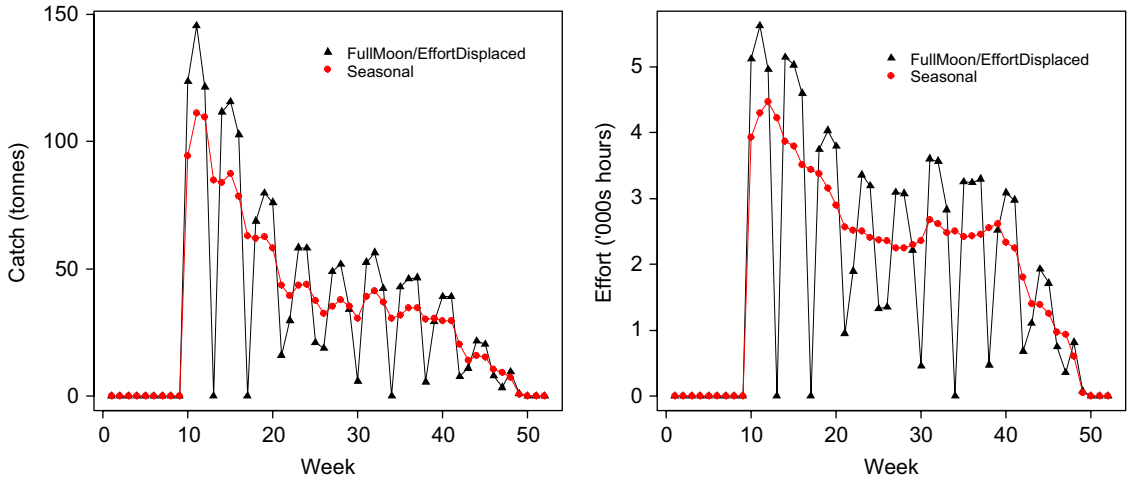

Fig. 5. Weekly catch (*left*) and effort (*right*) in 2005 averaged over 20 replicate simulations for the Seasonal and FullMoon/EffortDisplaced scenarios.

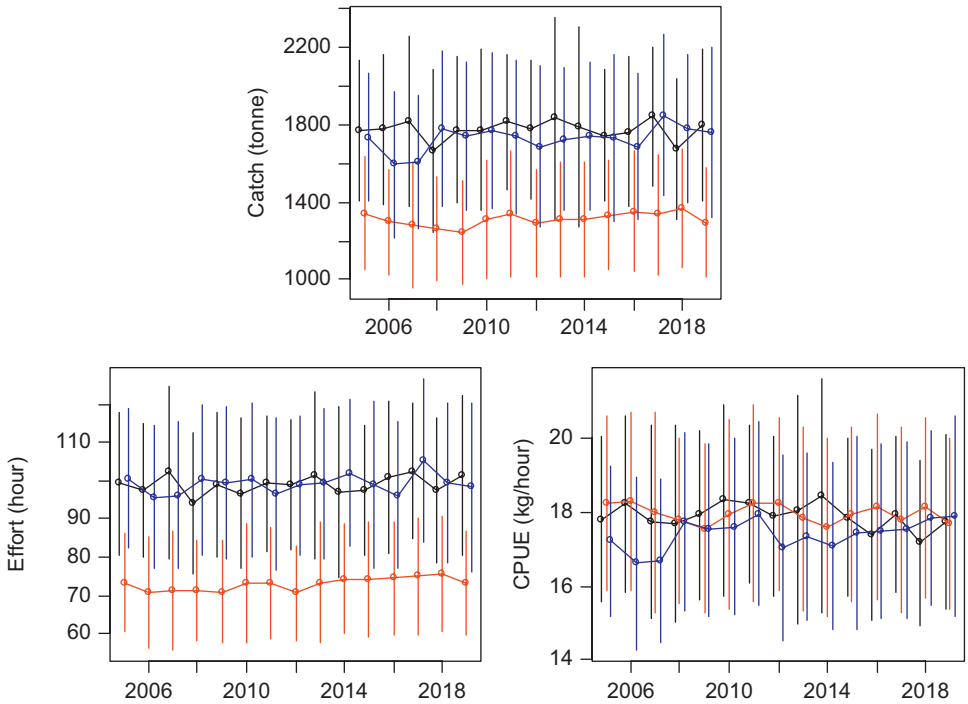

Fig. 6. Annual catch (*top*), effort (*bottom left*) and CPUE (*bottom right*) with error bars representing standard deviation over 20 replicate simulations for the Seasonal (*blue*), FullMoon/EffortDisplaced (*black*) and FullMoon/EffortRemoved (*red*) scenarios.

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.

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 (Fig. 6). Such variation may be related to the timing of full moon within the season, but the relationship, if one exists, is not straightforward. On the other hand, it may simply be random variation, since the standard deviations themselves are much larger than the difference in the means.

6.2. Benthic impact performance measures

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

From Table 2, 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. Fig. 7 shows the separate indicators for each benthic group. The overall resilience of a benthic group is directly related to the extinction coverage.

From Table 1, 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 Fig. 7: the bottom five scenarios, with, roughly the same amount of effort (100,000 h per year), are very similar to one another whereas the top three scenarios have increasingly more benign effect of the benthic groups (the top scenario having only 55,000 h effort per year).

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. Fig. 8 shows the overall density in 2020 relative to the overall pristine density, taking into account the estimated spatial distribution. (Overall density is the average of the density over all cells.) This is in contrast to Fig. 7 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).

Fig. 8 also shows the second effect of differentiation among the spatially explicit scenarios. On average, the bottom five 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 benefit from the Masig closure, because they have higher pristine densities *inside* the closure.

This is shown in even more detail in Fig. 9, which shows the standard deviations as well. The variation across replicate simulations is such that the bottom five scenarios are largely indistinguishable. Also it may be that the spatial effect for Porifera is merely random variation, but the effect for Holothuroidea appears real. In fact, Fig. 10 (*left*) shows that the highest density region for Holothuroidea is a band that encloses the Masig region. At the end of the historical period, the area around Masig is depleted (Fig. 10, *middle*). Under *StatusQuo*, this situation would persist, and the map in 2020 (not shown) looks very similar to

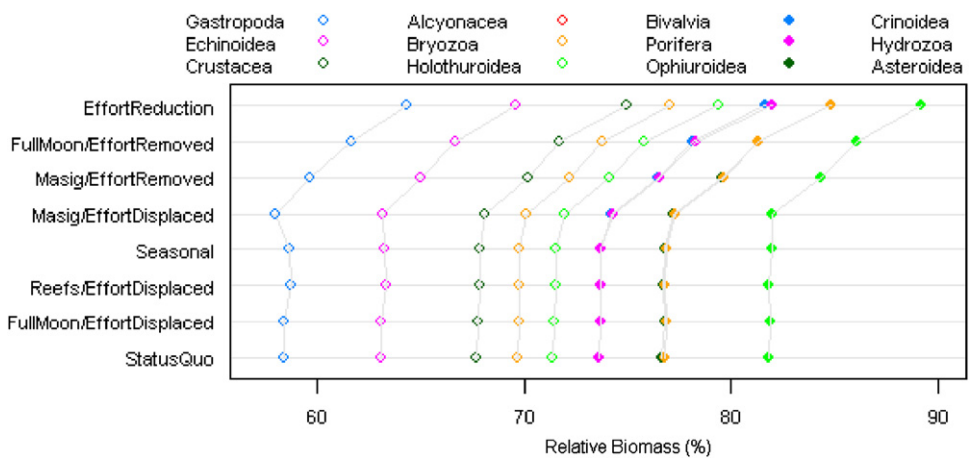

Fig. 7. Overall relative biomass in 2020 of 12 benthic groups for each scenario. Overall relative biomass is the average of the relative biomass over all cells. The Hydrozoa and Crinoidea are exactly coincident, and so are Bryozoa and Alcyonacea. The scenarios are ordered by overall relative biomass averaged over groups. Standard deviations (omitted for clarity) are around 1%.

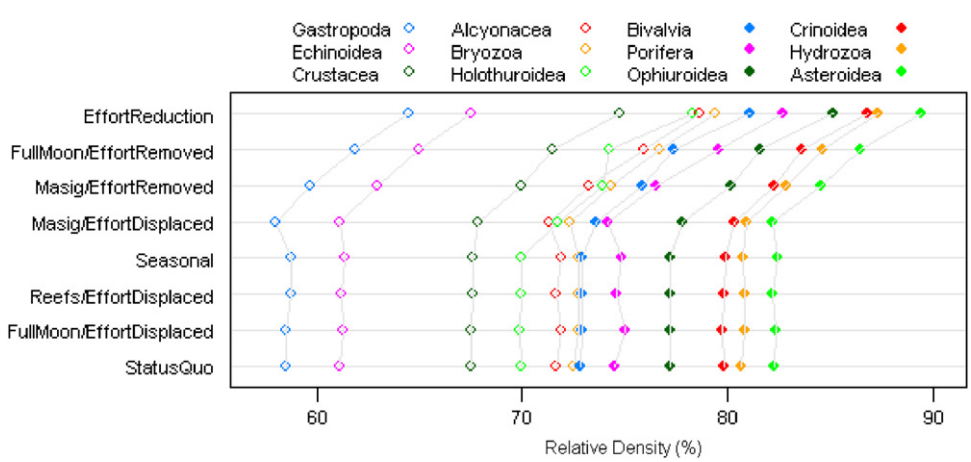

Fig. 8. Ratio of mean density in 2020 to mean pristine density of 11 benthic groups for each scenario. The scenarios are ordered by overall relative biomass as in Fig. 7.

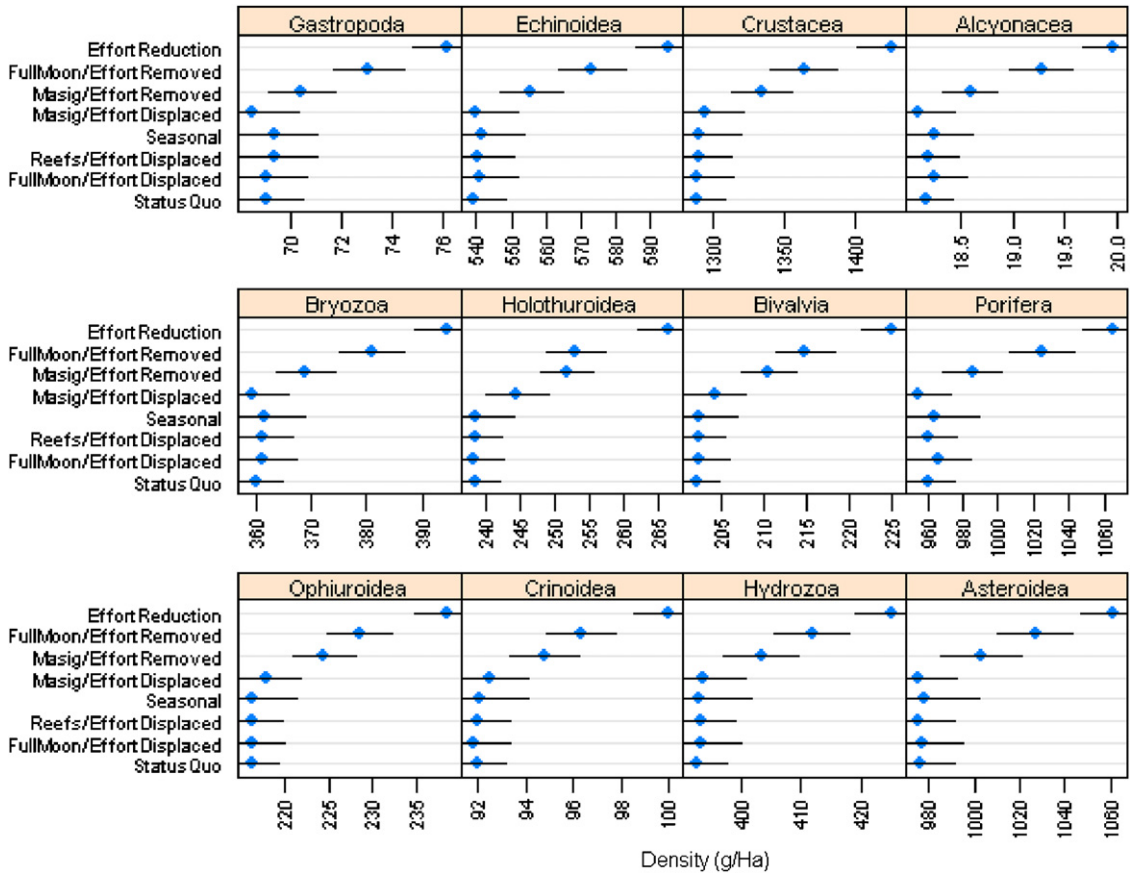

Fig. 9. Overall density in 2020 of 12 functional groups for each scenario. The scenarios are ordered by overall relative biomass as in Fig. 7. Standard deviations over 20 replicates are indicated by thin bars.

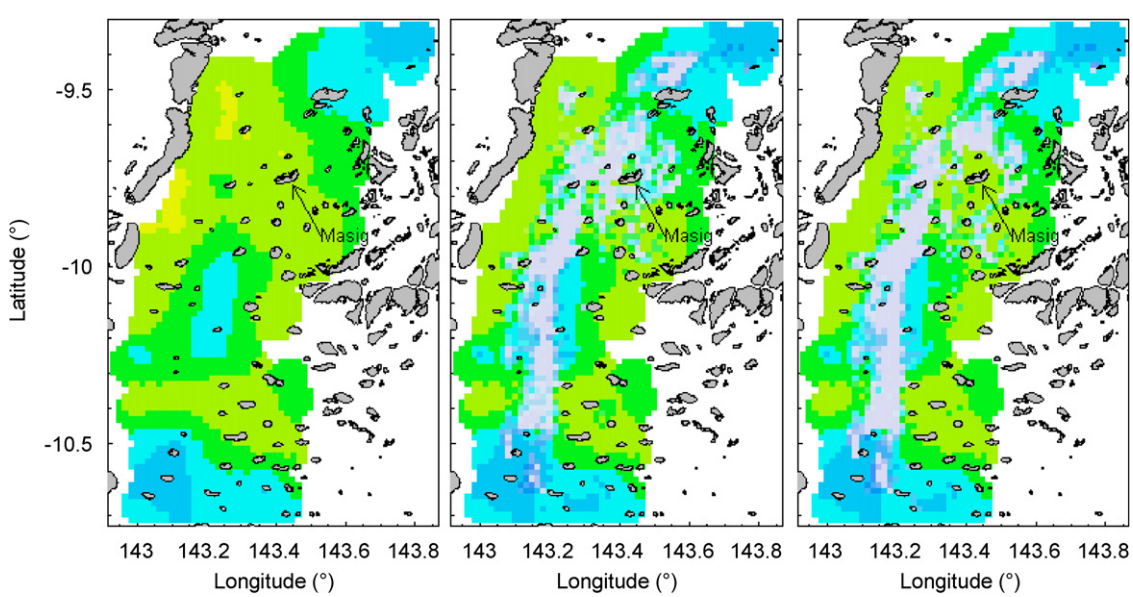

Fig. 10. Density (g ha^{-1}) of sea cucumbers (Holothuroidea) (*left*) in the pristine state, (*middle*) in 2004 at the end of the historic period and (*right*) in 2020 under the *Masig/EffortDisplaced* scenario. The colour scale is given in Fig. 11.

that in 2004. The biomass recovers somewhat inside the Masig closure, although some further degradation is visible in the main fishing band to the west, albeit in the lower density regions (Fig. 10, *right*) (Fig. 11).

The spatial effect can perhaps be seen more strongly for individual species. For instance, according to the detailed decision table in Fig. 12, *Dichotella* sp.1, *Alertigorgia orientalis* and *Subergorgia suberosa* are all impacted more by the Masig closure

scenarios. This is because these species have substantial densities away from the Masig closure. For instance, *A. orientalis* has relatively higher pristine densities in an area to the west of Masig (Fig. 13, *left*), which is also a trawling hot spot. Under the Masig closure, more effort is directed to this area leading to an overall loss in biomass (Fig. 13, *left*) relative to the *StatusQuo*.

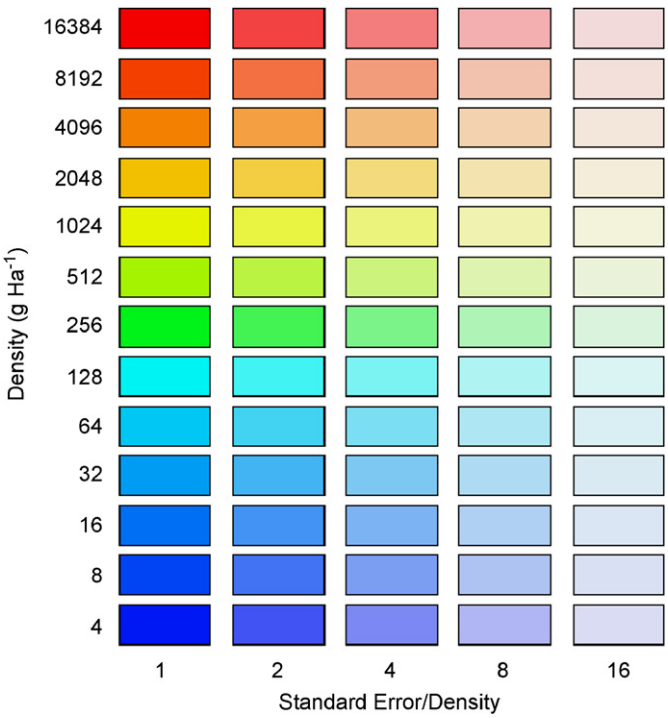

Fig. 11. Colour scale for maps of benthic density. Density is indicated by hue and precision by saturation. Washed out colours tending to grey imply higher uncertainty in the estimates.

It appears from Figs. 7 and 8 that the relative effect of different scenarios is fairly similar for each benthic group. This is more clearly seen in Fig. 14 where the effects relative to *StatusQuo* are plotted for both the relative biomass and density indicators. The strongest effect (up to about 10%) is due to the decrease in effort for the top 3 scenarios. The bottom 4 scenarios are effectively indistinguishable. Within each of the top 4 scenarios, there is some variation across benthic groups.

For relative biomass (R), there is a subtle non-linear dependence on extinction coverage. Most groups (Crustacea–Hydrozoa, referring to the ordering in the key) lie within a tight cluster. Asteroidea, however, always lies to the left. This is because Asteroidea have a relatively high extinction coverage and so they are less affected by changes in effort. In the extreme case of a completely invulnerable species ($d/r = 0$), all scenarios would be identical and would line up on the dashed vertical line. The other exceptions are Gastropoda and Echinoidea for the Masig scenarios. Their relatively low values arise because, with their low extinction coverage, these groups are already highly depleted in the Masig area before the scenario is put in place. They are not able to fully recover in the closed area by 2020 and, what is more, they are further depleted outside the closure relative to *StatusQuo*. This is an example of a transient deleterious effect: had the scenario been continued indefinitely, there would have been a slight improvement relative to *StatusQuo*. The effect of extinction coverage, excluding the exceptions already noted, is about 1%.

The density indicator (D) includes the effect of spatial distribution of the biomass. This indicator tends to have a slightly higher variation than the relative biomass. The large effects on Porifera and Holothuroidea in the Masig scenarios are evident here. Furthermore, we can see that Hydrozoa and Crinoidea do not benefit as much from the top 2 effort reduction scenarios.

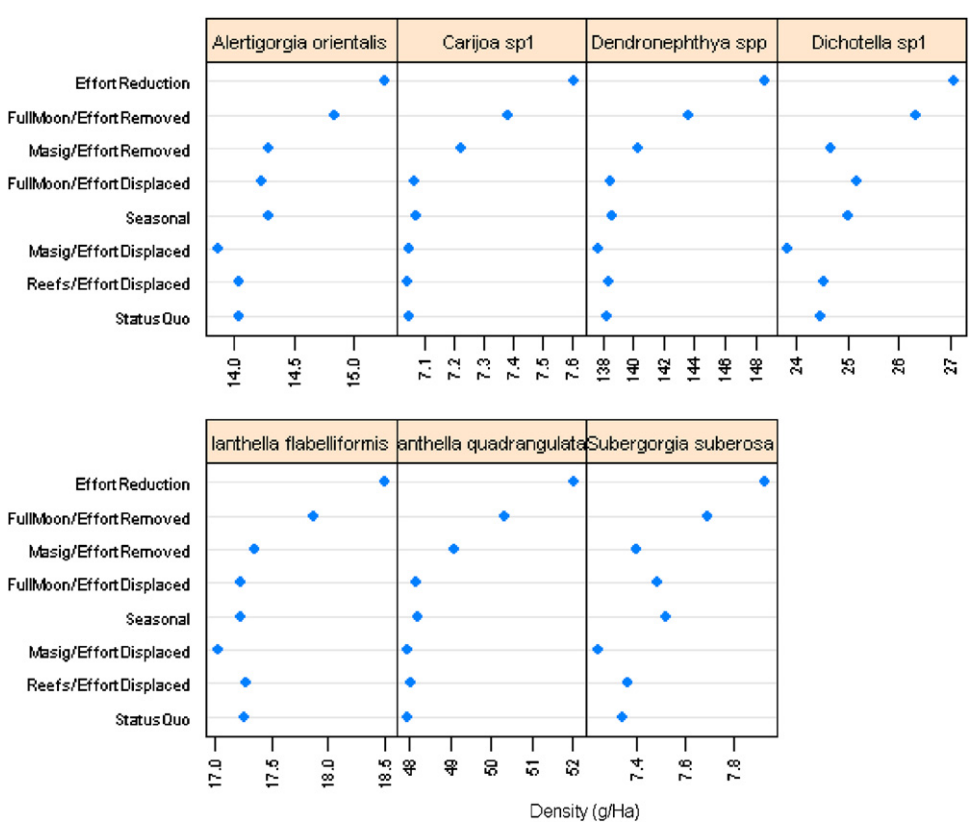

Fig. 12. Overall density in 2020 of seven individual species for each scenario.

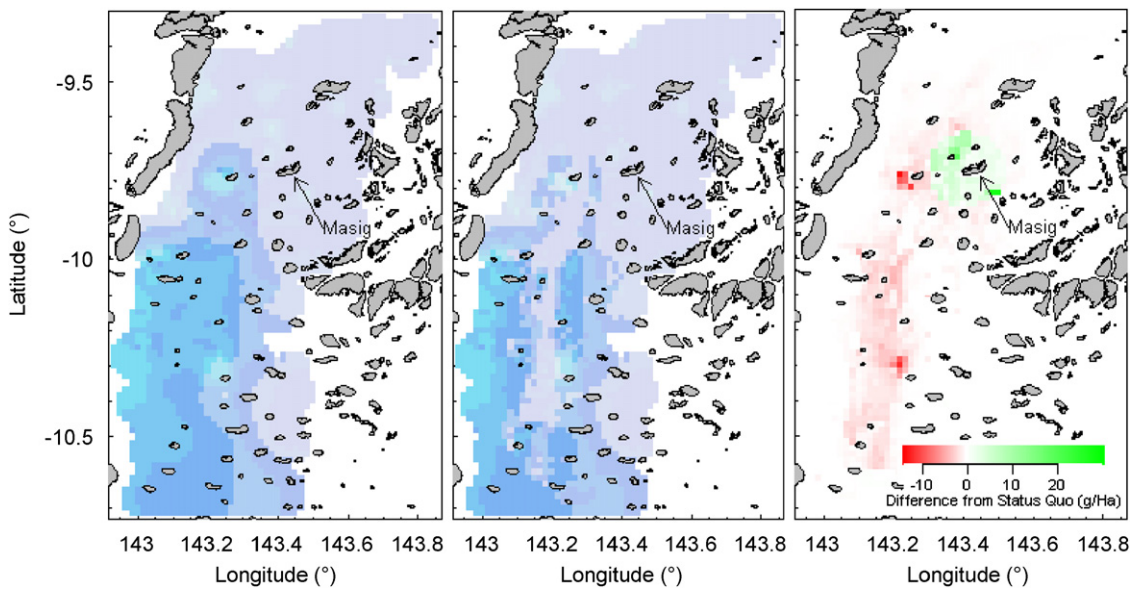

Fig. 13. Density (g ha^{-1}) of gorgonian (*Alertigorgia orientalis*) (*left*) in the pristine state and (*middle*) in 2020 under the Masig/EffortDisplaced scenario; (*right*) the difference in density between this scenario and *StatusQuo*. The colour scale for the left and middle plots is given in Fig. 11.

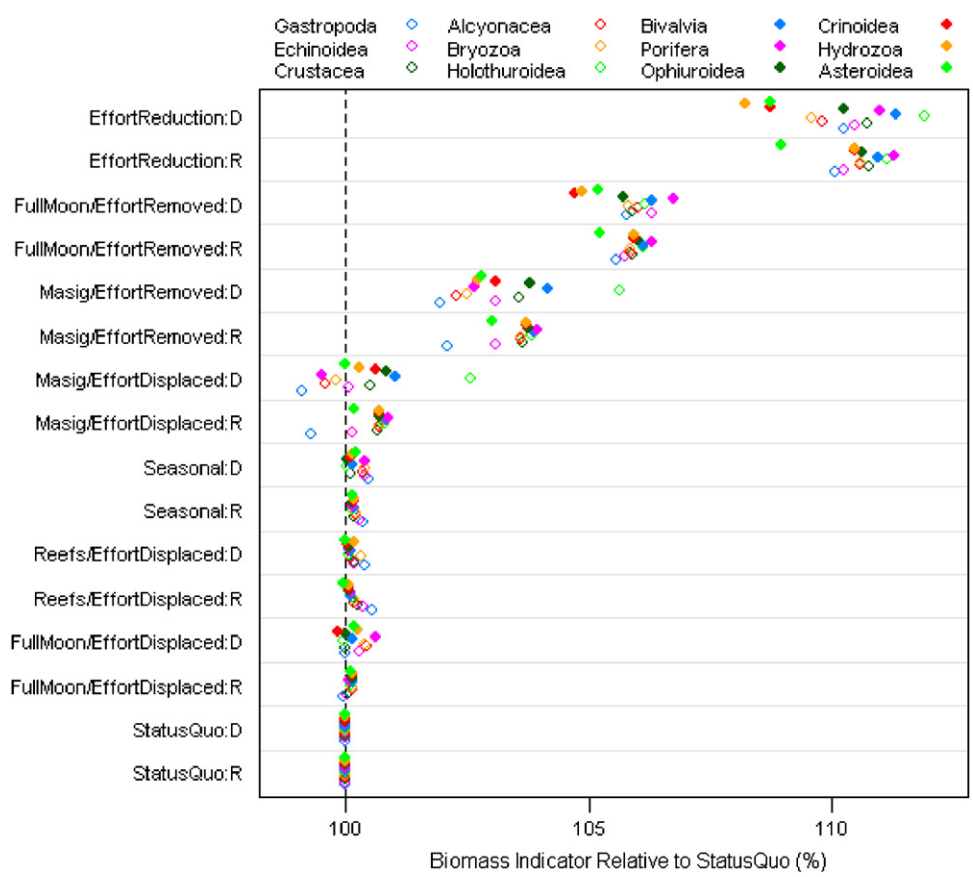

Fig. 14. Ratio of biomass indicator to Status Quo biomass indicator for all scenarios in 2020. The biomass indicators are relative biomass (R) and density (D). Within each horizontal band, the groups are ordered vertically on extinction coverage.

The spatial effect tends to be comparable to the effect of extinction coverage, although in the case of Holothuroidea it can be large enough (about 2%) to make one scenario comparable with another.

As stated earlier, the results in Fig. 14 can be used as a basis for sensitivity analysis. That is, the effect on the indicator ratios of uncertainty in r and d can be estimated by inspection of the systematic dependence through r/d.

7. Conclusions

Generally, the effect of observation error in the simulations is much larger than that of process variation. For instance, in Fig. 9, the error bars represent process variation of a few percent. However, the observation error due to uncertainty in B_0, r and d is much larger, of the order of 100%. On the other hand, the *ratios* of indicators relative to the *StatusQuo*, say, are much more robust. To the manager, who is interested in comparing alternative scenarios, it is these ratios that are of most concern.

The main message of these simulations, as summarised in Fig. 14, 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.

The spatial distribution of effort plays a lesser role. Its effect is usually an adjustment to the main effect of the total effort. Occasionally, however, as with Holothuroidea, the spatial effect can be as important as the effort effect. And sometimes, as with Gastropoda, Porifera, Alcyonacea and Bryozoa (Fig. 14), a spatial closure can be counter-productive, a phenomenon also reported by Fujioka (2006) and Hiddink et al. (2006b).

A recovery of up to 10% for an effort reduction of about 50% in the *EffortReduction* scenario may seem a very slow response. The reason for this is that the effort is fairly aggregated so that some cells have relatively high effort, some have intermediate levels, but most have relatively low effort. The rate of recovery depends on where on the logistic curve the biomass lies. The recovery can be fast in cells with intermediate levels of effort but slow for cells with either high or low effort. Typically, blanket effort reductions are indiscriminate as to which cells are affected, and so the benefits to the benthos tend to be less marked than the reductions in effort and consequently catch. The diluting of the benefits of effort reduction can also be seen in the results of Hiddink et al. (2006b) where changes in biomass tend to be roughly 10 times smaller than changes in effort. A more targetted closure regime taking into account the spatial distribution of the benthos or the sensitivity of the habitat as tried by Hiddink et al. (2007) could be of greater benefit, even in scenarios where the effort is not reduced.

Under the scenarios considered here, however, we have seen that spatial effects are generally not very strong compared to overall effort effects. This is partly 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 as we saw with *A. orientalis* (Figs. 11 and 12).

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 temporally (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.

MSE studies have been carried out by various authors using a range of models (Gribble, 2003; Fujioka, 2006; Hiddink et al., 2006b). The issue of which models are appropriate to use is a complex one. Simple models, though easy to use and understand, can miss important biological processes, such as migration. Highly complex ecosystem models on the other hand, though rich in processes, tend to have large numbers of parameters, with varying degrees of uncertainty, upon which the model outputs depend with uncertain sensitivity. Managers should seek input from a range of models of differing complexity, gain confidence in areas where the models agree and question areas where the models differ. The results of this study, regarding influence of effort and spatial closure, are in broad agreement with those of Fujioka (2006) and Hiddink et al. (2006b).

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Appendix. MSE Software

The current study was used as a test-bed to develop a robust and scalable software platform for performing MSE. There are certain operations that are always required by MSE: *MSE specification*, *running of MSE scenarios* and *reporting*.

Specification involves the setting of input parameter values and the permanent storage of those values. Input parameters comprise all the data the MSE needs to run, including high-level parameters (e.g., the name of the MSE, how many replicates, start and end times), low-level parameters (e.g., OTU names, depletion rates, TAEs, closures) and also which quantities to output (e.g., names of performance measures and state variables). A traditional approach to parameter specification is to set the values in flat data files and to store those files under a directory whose name signifies the intent of the MSE. Our approach is to formalise this by storing all parameter values in a relational database, with an MSE table at the top of the hierarchy, and with all parameters joined by cascading links to a single record in the MSE table. The user communicates with the database through a graphical user interface. Any large input datasets, such as arrays, are also stored in databases, and the query to extract them is stored in the specification.

Running the MSE scenarios comprises a *parsing* stage, in which the specification is retrieved from the database and the appropriate software modules are instantiated, and a *simulation* stage, in which a master program initialises the modules with data and runs them sequentially within a time-stepping loop. Each module is a component of the MSE. For instance, our modules comprised a manager, human activity, ecosystem, observer and assessor

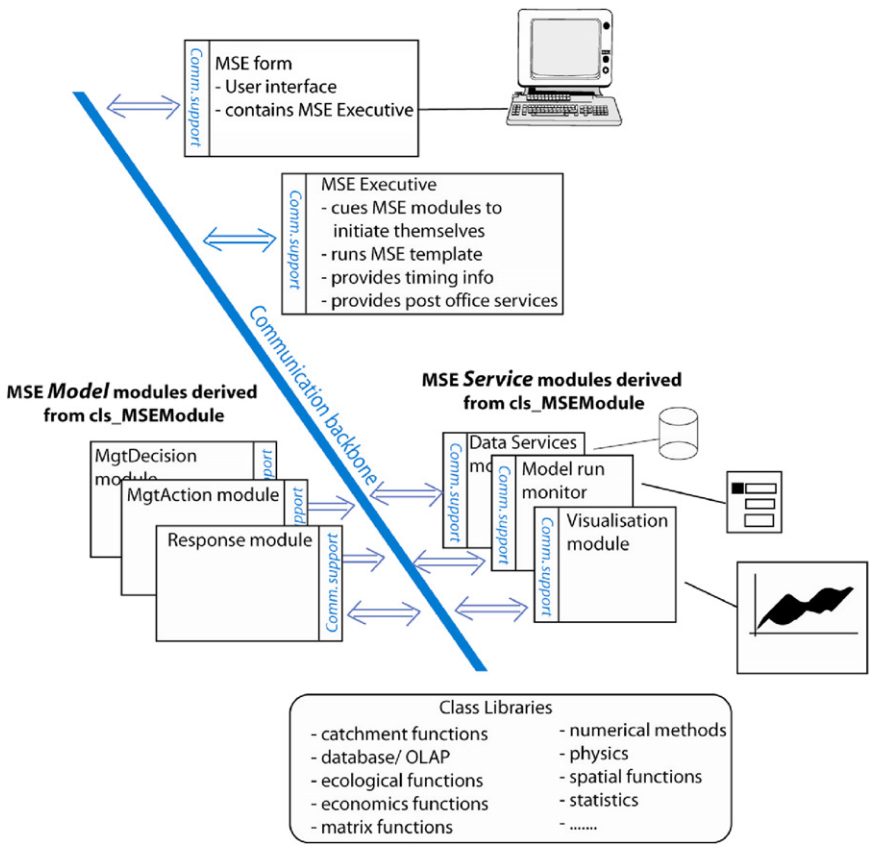

Fig. A.1. The software that implements the Torres Strait MSE is built up of autonomous modules that use a generalised method to communication.

(see Fig. A.1). Modules communicate with one another via a generalised communication interface, which is provided by the software framework. The advantage of this is that two different modellers (say a fishery economics expert and a stock assessment expert) can plug their models into the same MSE. To allow the models to work, the experts only need to ensure their modules conform to the communication interface.

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

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

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