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Evaluating marine spatial closures with conflicting fisheries and conservation objectives

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Summary

1. Spatial management is used extensively in natural resource management to address sustainability and biodiversity issues, for example through declaration of terrestrial National Parks and marine protected areas (MPAs).
2. Spatial management is used also to optimize yields or protect key parts of the life cycle of species that are utilized (hunted, farmed or fished), for example through rotational harvesting.
3. To evaluate the effectiveness of marine spatial closures with conflicting fisheries and conservation objectives, a series of marine fisheries closures are here analysed using an integrative modelling tool known as management strategy evaluation (MSE).
4. This modelling framework combines a food web model of a tropical ecosystem fished by a prawn (shrimp) fishery that emulates the resource being managed, together with the present management system and risk-based tools of fishing the prawn species at maximum economic yield.
5. A series of spatial closures are designed and tested with the aim of investigating trade-offs among biodiversity (MPA), benthic impacts, ecosystem function, key species at risk to fishing, economic and sustainability objectives.
6. *Synthesis and applications*. This paper illustrates that existing tools often available in actively managed fisheries can be linked together into an effective management strategy evaluation framework. Spatial closures tended to succeed with respect to their specific design objective, but this benefit did not necessarily flow to other broad-scale objectives. This demonstrates that there is no single management tool which satisfies all objectives, and that a suite of management tools is needed.

Key-words: benthic impacts, conflicting management objectives, ecosystem, effects of trawling, management strategy evaluation, marine protected areas, risk assessment, threatened and endangered species, trade-offs

Introduction

Fisheries management is inherently complex as it involves a resource that is a national public asset, yet (often) only entrusted to a few owners (Gordon 1954; Beddington, Agnew & Clark 2007). This means that individual business objectives such as maximizing profits need to be traded against the medium-term impacts of fishing on the ecosystem and intergenerational needs for a clean and sustainable environment (Grafton, Kompas & Hilborn 2007; Costello, Gaines & Lynham 2008). Defining objectives and trade-offs for fisheries management is therefore often extremely complex and politically sensitive (Gislason *et al.* 2000; Mardle & Pascoe 2002; Hilborn 2007). Ecosystem-based Fisheries Management (EBFM) or the Ecosystem Approach (EA), has an overall objective to 'sustain healthy marine ecosystems and the fisheries they support' (Pikitch *et al.* 2004). It is therefore important to investigate the impact of fishing both at the scale of the target species' distribution and the ecosystem.

The implementation of EBM approaches involves integration of a range of data sources, tools and often complex models involving bioeconomics, ecology and

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biophysics (Christie & White 2007). One approach is management strategy evaluation (MSE), a simulation-based framework that allows evaluating broader ecosystem-based fishery management strategies via the use of integrated models coupled with management decision rules (Smith 1994; Fulton *et al.* 2011). An MSE simulates the whole management and biological systems together, allowing comparison and evaluation of the relative performance of different management strategies (Sainsbury, Punt & Smith 2000). The MSE consists of an operating model that can be considered as a 'virtual' resource and is seen as a representation of the 'true' underlying dynamics of the system and the fishery. The operating model for an EBM approach would include the biology of the components affected by fisheries, such as the benthos and associated ecosystem, and all the processes that control the dynamics of that system. The operating model also generates all the data needed within the management strategy. This management strategy remains 'ignorant' of the 'truths' included in the operating model other than the data given it. Each combination of the types of data used, the assessment-related analysis method applied, and the decision rules used constitutes a different management strategy. The outcome of the management strategy (e.g. the level of effort, which areas and times are open to fishing) is fed back to the operating model and is used to determine the dynamics of the 'true' situation being managed. The key component that links the management strategy and the operating model is a fleet dynamic model that applies the effort spatially and temporally, or as required.

Biological populations and their communities and habitats have an inherent spatial structure and our knowledge about this spatial patchiness is based on how aquatic processes change over space and time, and how multispecies assemblages interact with each other and with forces at different spatial and temporal scales (Botsford, Castilla & Peterson 1997; Sanchirico 2005). Spatial management in fisheries is therefore one of the many tools that can be used to address fisheries management objectives. They are certainly not a panacea, but can be a powerful tool to address many complex aspects of fisheries management (Pelletier & Mahévas 2005; Garcia & Charles 2007).

Networks of marine protected areas (MPAs) and spatial planning are the centrepieces for management and conservation of Australia's marine biodiversity (Commonwealth of Australia 2006). These spatial management provisions are being implemented within the bioregional planning process that has divided Australia's Exclusive Economic Zone into five major bioregions (Commonwealth of Australia 2006; Heap *et al.* 2006).

In the Australian Commonwealth's fisheries harvest policy (DAFF 2007), controlling total effort or catch in a fishery is generally managed through direct and often nonspatial Total Allowable Catch or Total Allowable Effort controls. However, spatial management is used extensively for other purposes in Australian fisheries management such as protecting key habitats. Pascoe *et al.* (2009) provide four high-level generic objectives for Australian fisheries management: 'enhance economic performance, ensure sustainable resources (commercial species), minimize environmental impacts and minimize externalities (encompassing both commercial and social objectives)'.

In this study, a tropical prawn trawl fishery was used to simulate the potential impacts of different spatial management scenarios that include various forms of closure to fishing while achieving the biodiversity or specific fisheries management objectives. The Northern Prawn Fishery (NPF) is a small fishery of about 50 vessels in the far north of Australia and is managed by a transferable gear unit system (a tradable unitization scheme where the gear units are expressed as a share of the total allowable headrope length), seasonal closures and extensive spatial closures. Most of the present closures are aimed at minimizing the capture of small prawns and avoiding key habitats such as seagrass beds (Kenyon *et al.* 2005).

This study tested spatial management options with objectives other than those for target species management. It investigated the effects of a range of potential spatial closures (beyond those already in use) that include (i) large biodiversity-centric MPAs closed to all fishing, (ii) ecological risk assessment (ERA) closures aimed to reduce the number of nontarget species that are assessed as being at risk in the Commonwealth risk assessment/management approach (Hobday *et al.* 2011) and (iii) closures that have a core objective to reduce the overall impact of trawling on vulnerable key taxa groups (e.g. sea snakes, sawfishes). The potential closures were assessed in terms of their influence on benthic and ecosystem impact while assuming effort was shifted with no economic or target species impact.

Materials and methods

BACKGROUND

The basic MSE framework has been described extensively elsewhere (e.g. Kell *et al.* 2007). Conceptually, however, it includes three key components: (i) the operating models that describe 'reality'; (ii) the management strategies that are to be evaluated; and (iii) the performance measures (PMs) that will be used to evaluate the performance of each management strategy in relation to the objectives. The use and development of MSE principles are well developed within the NPF for target species (Dichmont *et al.* 2006), and include economics (Dichmont *et al.* 2008). However, this framework is expanded so that (i) the operating model is a spatially explicit ecosystem model (Ecospace: Walters, Pauly & Christensen 1999) and (ii) the management strategies include the present bioeconomic assessment and rules, but is extended to include an effects of trawling (EoT) model (Ellis *et al.* 2008) that uses a set of rules to adaptively close (but not re-open) mostly low-impacted spatial grid cells within a habitat. The PMs are aligned with assessment of ecosystem, benthic impact and nontarget risk (see section below). The MSE concentrated on the stocks within the Gulf of Carpentaria (GoC; Fig. 1), where on average

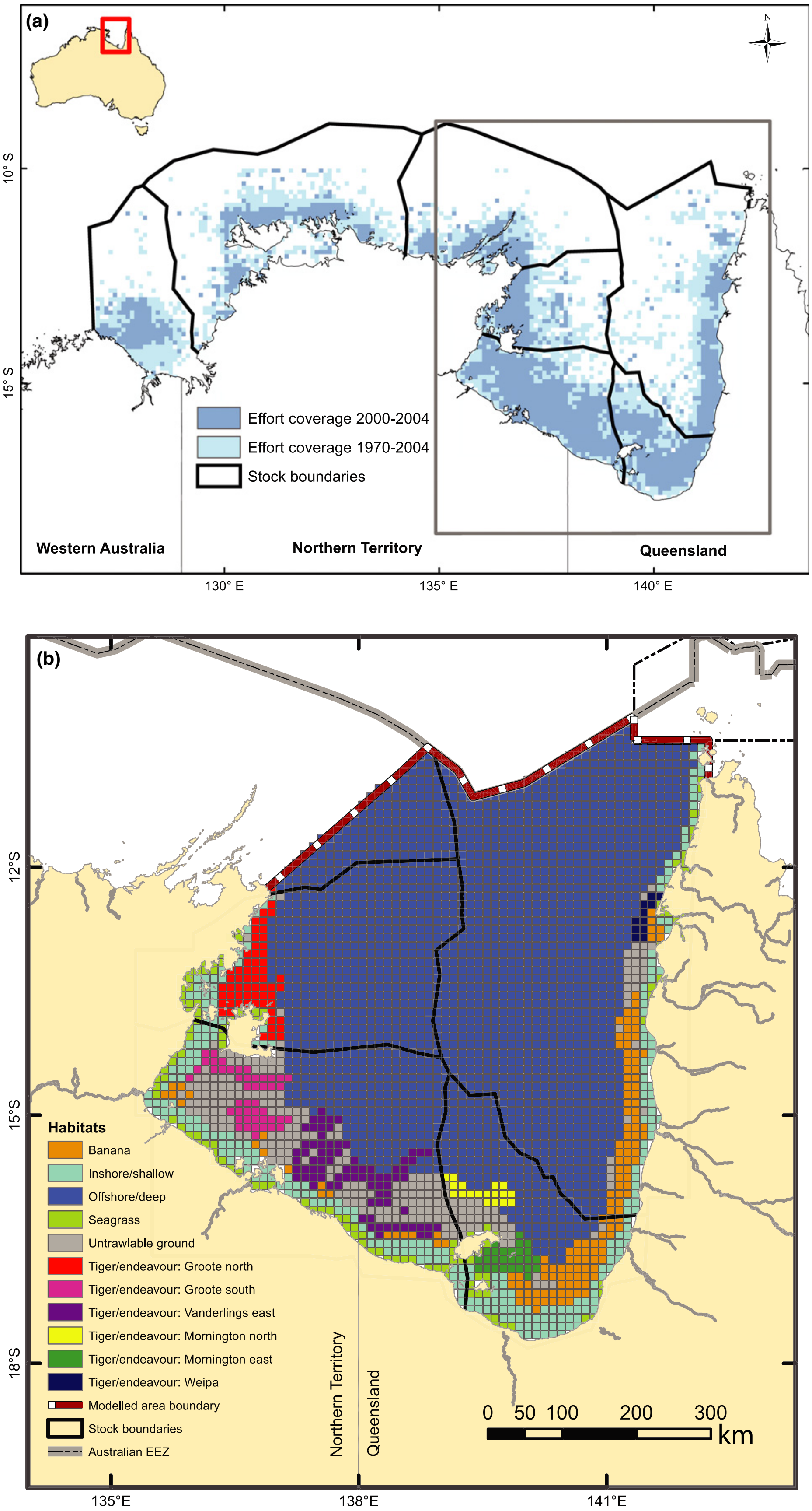

Fig. 1. Map of the Gulf of Carpentaria region of the Northern Prawn Fishery (a), main stock regions (black lines) and main fishing grounds (grey) and (b) the main benthic habitats defined in a 6-min grid and used by the Ecospace operating model.

87% and 78% of the tiger (*Penaeus esculentus* and *P. semisulcatus*) and endeavour (*Metapenaeus endeavouri* and *M. ensis*) prawn species are caught, respectively.

Figure 2 depicts the overall system, the relationship among the spatial MSE framework components and the data types used by each component of the MSE. There is a clear vertical separation of the large and small spatial scale components, which distinguishes between the overall stock-region prawn models and the fine-scale 6-min. grid ecosystem, risk assessment or benthic impact models (Fig. 2). A hierarchical fleet dynamic model, which starts at stock level and then distributes effort at the fine-scale level within a stock, was used.

OPERATING MODEL

Ecosystem

The temporal and spatial dynamics of the ecosystem was described using Ecopath with Ecosim (EwE 6·3) software (Christensen & Walters 2004). How the model was set up is described in Section S1, Supporting Information. The model comprised 53 functional groups (FGs; Table S1a, Supporting Information), some of them aggregated by their ecological functional roles (e.g. planktivores) or biology (e.g. similar production/biomass ratio). A number of single-species FGs were defined for species of significance to commercial fisheries (e.g. banana and tiger prawns, rock and slipper lobsters). The overall model comprised 16 fish groups, 27 invertebrate groups, four primary producer groups, two groups of marine mammals, two reptile groups and two nonliving groups (discards and detritus). The banana and tiger prawn groups were subdivided into two FGs (juvenile and adult) because of the significance of these species in the MSE.

The Ecospace model is structured on FGs, which are biomass pools, linked trophically by predator–prey relationships (i.e. Lotka–Volterra submodels) and spatially by migration among neighbouring grids. Movements of FGs are driven by parameters such as foraging behaviour, avoidance of predation and dispersal rates that are linked to a range of defined habitats preferred by each FG.

The GoC model was represented by an array of 80 × 80 grids of size 3·76 × 3·76 min. The 11 Ecospace ‘habitats’ (loosely defined as a combination of habitat and fishing region; Fig. 1) were built around the major fishing grounds identified in the gulf (Table S1b, Supporting Information).

Prawns

We used the approach of Dichmont *et al.* (2008) to model the population dynamics of prawns in the operating model. Endeavour and tiger prawn species were described using a multispecies, multistock model (boundaries are defined in Fig. 1). The population dynamics of each species in each region where they occur were governed by a delay-difference model operating on a weekly time-step. The conditioning of the operating model involved a series of steps: (a) estimating annual recruitments from data on catches and standardized catch-rates, (b) using these to estimate the parameters of a Ricker stock–recruitment relationship (Dichmont *et al.* 2003), and then (c) estimating Maximum Economic Yield (MEY) from the output parameters of (a) and (b), combined with the economic parameters as described in Dichmont *et al.* (2008).

Risk assessment

The ERA method described in Zhou & Griffiths (2008) was used to assess the risk of fishing and categorized species within the ecosystem at risk. This quantitative Sustainability Assessment for Fishing Effects method consists of two components: estimating fishing mortality and assessing this with respect to biological limit reference points. The mean harvest rate (u) depends on the fishing activity overlapping upon each species' core distribution area within the GoC, which is then adjusted by the probability of

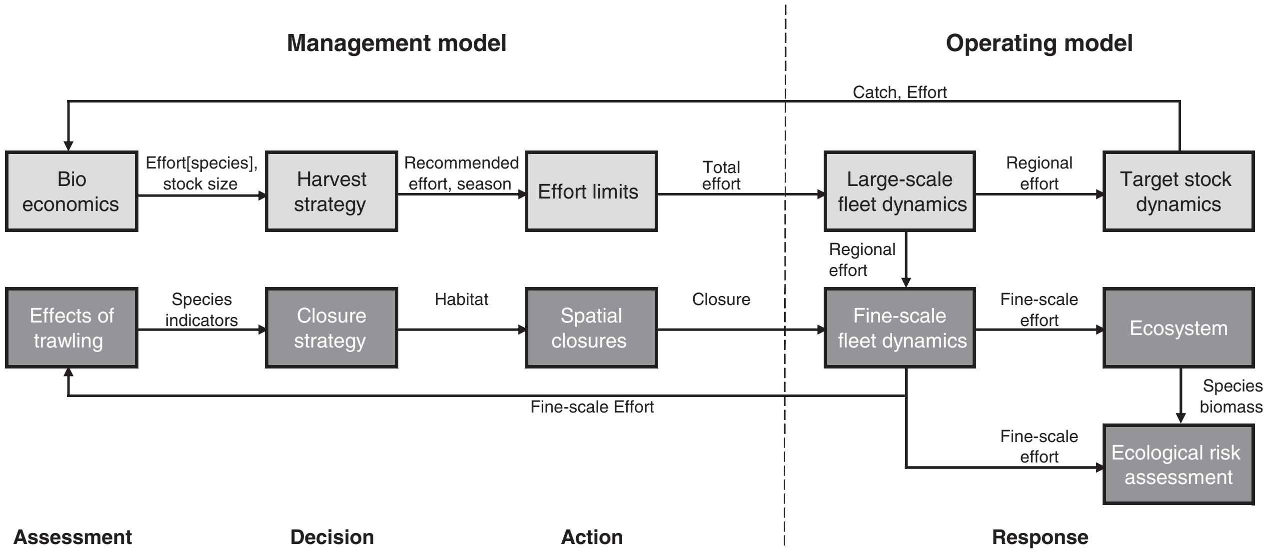

Fig. 2. Schematic diagram of the management strategy evaluation system. Each box represents a separate submodel. Communication between submodels and the kind of data supplied are shown by the labelled arrows. The dashed line separates the management model from the operating model. The models also split into large-scale components (light grey) on the scale of stock regions and fine-scale components (dark grey) on the scale of 6-min. management grids. The only link between the large-scale and fine-scale components is the regional effort from the large-scale fleet dynamics submodel.

being caught (i.e. selectivity) by the prawn trawl. This means that fishing mortality here is not the fishing mortality rate as estimated in stock assessments, but rather the fraction of the population that can potentially be caught by the fishery. This calculation requires an estimate of the population biomass, which is provided by the Ecopath model. Two reference points were used: u_{msm}, the fishing mortality corresponding to the maximum sustainable fishing mortality (MSM) at the biomass that supports MSM [B_{msm}; equivalent to maximum sustainable yield (MSY) for target species]; and u_{crash}, the minimum unsustainable fishing mortality that, in theory, may lead to population extinction in the long term (Zhou & Griffiths 2008). If the mean harvest rate is greater than the calculated u_{crash} for a particular species, then that species is considered to be at extreme risk of its population becoming unsustainable in the long term under the current level of fishing effort.

Table S2 (Supporting Information) lists the main taxa included in the ERA and their related parameters, which includes a number of fish-based FGs that are known to have high susceptibility to trawling and exhibit a range of recovery rates and fishing mortalities (Stobutzki *et al.* 2001b). These FGs were determined by the availability of data on biological limit reference points for the species in those groups. A risk assessment of these FGs, based on the effort distribution in the GoC from 2001 to 2005, identified which FGs were 'at risk'.

Since only FGs and not species occur within Ecopath (apart from commercially important prawn species), their respective FGs were used instead. Undertaken with care, Caughley (1982) shows that amalgamating species may not have a significant effect on overall dynamics if the population parameters of the constituent species are similar, which is the case with sea snakes caught by the NPF.

Fleet models

The fleet dynamics models that describe the movement of the fleet both at a stock level and then subsequently at a 6-min. grid level within a stock were described by Venables *et al.* (2009). The models consist of two phases to provide the operating model with effort at the necessary spatial and species (dis)aggregated scale.

The first phase is a vessel-movement model that assigns effort at the prawn stock and species level at a weekly time-step. It is based on a discrete-state, time-inhomogeneous Markov chain model, with transition probabilities following a multiple logistic model. For each stock, the prawn operating model provides the vessel-movement model with: the total effort directed towards tiger/endeavour and banana prawns for that week; banana catches for the previous week; and the tiger/endeavour and banana prawn efforts for the previous week. The vessel-movement module schedules effort to each of the stock regions for the current week considering the above information, as well as the average cost of travel between stock regions (based on distance travelled and fuel price) and spatio-temporal variables, such as the stock region, time of year and the time since fishing commenced in the season.

The second phase takes the effort allocated to a stock region in that year and assigns this to 6-min. grids, as needed by the benthic impact model. This phase assumes that fleet behaviour is a reflection of past behaviour but modified based on what was caught in the previous week and at that time in previous years. Closures that removed very large areas of the main fishing ground were not included in the analysis as the assumptions underlying the second phase model may not be valid under such circumstances.

MANAGEMENT MODEL

Management models normally consist of monitoring, assessment and decision components, framed within an adaptive cycle.

Target prawn species

The target species monitoring and assessment model is described in detail by Dichmont *et al.* (2008). The monitoring component provides catch and effort data by week and species for each 6-min. grid from the operating model assuming no error (vessels are monitored using VMS). Two survey indices for recruits and spawning numbers are generated, as well as economic cost and price data for all closure strategies. The decision on total effort and season length in each projected year is as per Dichmont *et al.* (2008), which uses a delay-difference bioeconomic model to set annual total effort and season length such that the spawning stock at MEY is reached within 7 years of that year's assessment (and which is updated each year).

Benthic impacts

The EoT model estimates the primary benthic impacts of repeated trawling on the biomass of benthic organisms, ignoring any long-term consequences of that removal on the ecosystem, including any effect on prawn productivity (Ellis *et al.* 2008). Each 6-min. grid contains a separate and self-contained Schaefer-type biomass-dynamic model for a range of benthic taxonomic groups. Each taxon undergoes depletion according to the degree of effort in the grid combined with the taxon's vulnerability to such effort, and recovers from that depletion along a logistic trajectory according to the taxon's intrinsic recovery rate. The biomass is assumed to be at carrying capacity prior to the establishment of the fishery. The model uses the entire historical spatial pattern of fishing (or estimates thereof) up to the current time to construct an estimate of the current biomass distribution relative to local carrying capacity. For the management model, a data-poor environment is assumed, in which the spatial distribution of FGs is unknown and is consequently constrained to be uniform in space. The model is described in detail in Dichmont *et al.* (2008).

The EoT model was used in the adaptive closures scenario to provide indicators of the long-term biomass of the most vulnerable species within each habitat (Section S3, Supporting information). The most vulnerable species, as measured by the ratio of per-tow depletion rate to intrinsic recovery rate, were large gastropod carnivores, echinoids, holothurians and sessile epibenthos. Habitats were spatially defined to be similar but, not identical, to the habitats in the operating model, so as to simulate some lack of knowledge of habitat boundaries.

SPATIAL SCENARIOS

Several scenarios were tested with differing objectives to address spatial closures that could be used (beyond those currently existing for target species and seagrass beds) to protect species within the ecosystem (Fig. 3). These objectives were to reduce: (i) direct and indirect ecosystem impacts through a full set of MPAs ('MPA'); (ii) number of species at risk through an externally (to the MSE) derived risk assessment model (Zhou & Griffiths 2008) aimed at moving the classification of 'at risk' FGs to 'not at risk' (scenario medium effort – 'ME'); and (iii) impacts on the benthos through adaptive spatial closures ('EoT' – Section S3, Supporting information). This is compared with a Base Case ('BC') scenario that describes the presently used fisheries management system with no additional closures. The ME scenario closed grids with <90th percentile of cumulative total effort over 5 years (i.e. grids with 200 or fewer effort days). For the EoT scenario, various low 'EoT' thresholds were tested (10%, 20% and 30%), but closures were never initiated since the thresholds were never triggered. This paper therefore only presents results for the high threshold (70%) to test the utility of such an adaptive method.

In all scenarios, total fishing effort was kept consistent among simulations and was set based on the bioeconomic model. These closures are shown (overlapping the fishing effort distribution) in Fig. 3.

PERFORMANCE MEASURES

Performance measures (described in Section S4, Supporting information) came from various components of the model: (1) ecosystem, the EwE operating model; (2) at-risk species, the ERA model; (3) biological, the prawn operating model; and (4) economic and spatial fishery impacts, the prawn and fleet operating models. The effects-of-fishing component (5) came from the EoT assessment submodel in the management model due to the adaptive nature of one of the closure scenarios. The ecosystem PMs were all derived by aggregation from a basic element – the biomass for a FG within a 6-min. grid. The basic elements were

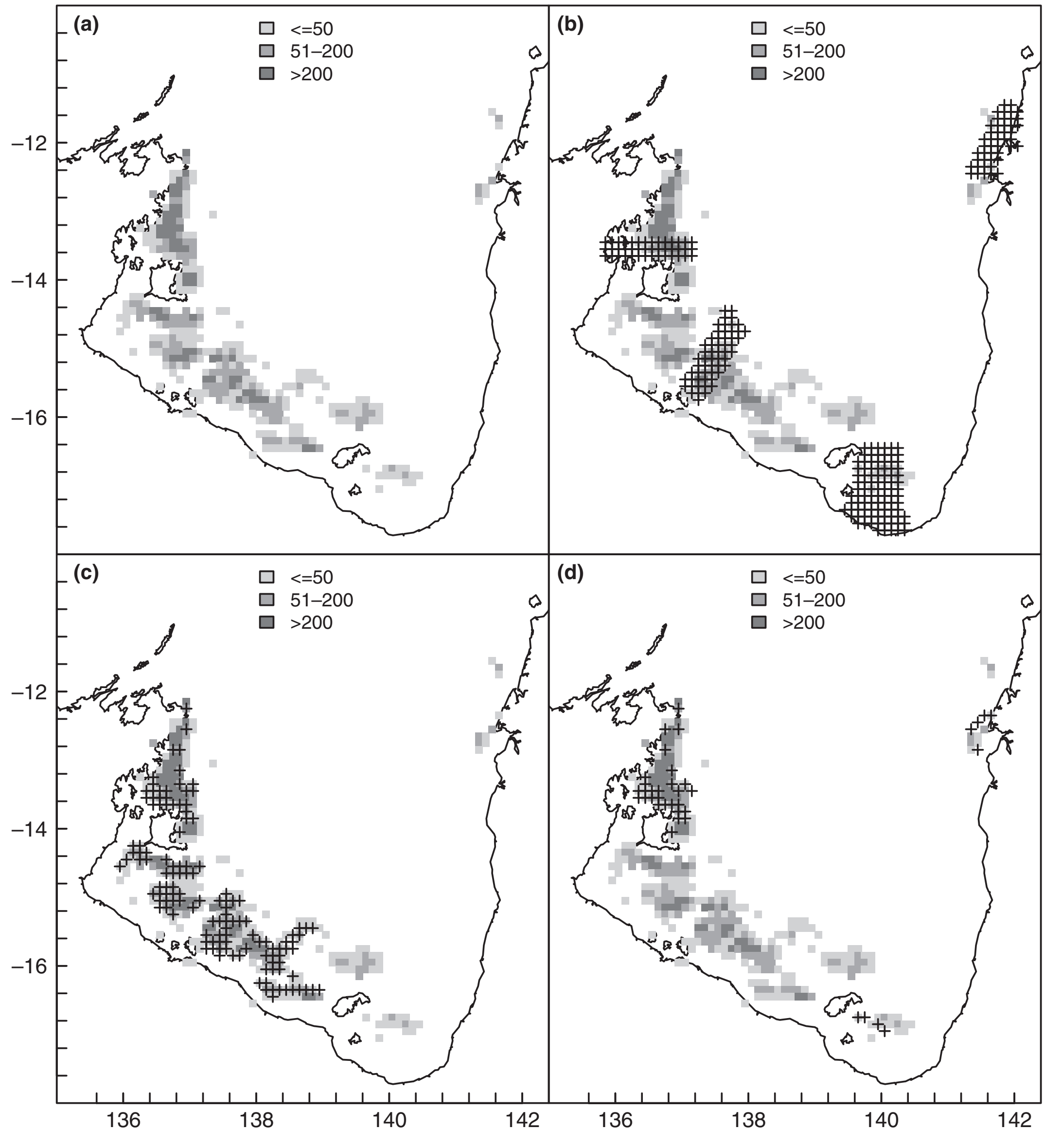

Fig. 3. The Base Case (top, left panel) showing the 5-year average fishing effort which has no closures with: the marine protected areas (MPA) closure (top, right), the medium effort (ME) closure (bottom, left) and one simulation of the adaptive Effects of Trawling (EoT) closures (bottom, right).

used in maps of distributions, and they were aggregated either spatially (to habitats or the entire GoC) and/or taxonomically (to all FGs or over trophic levels).

In order to compare the performance of the spatial closures, the results of the spatial MSE framework can be integrated into several forms of graphical reporting – by time, habitats, regions, scenarios and various combinations among them. Only a subset is shown in this paper or in the Supporting Information. Due to the computational demands of the calculations, only 30 simulations were undertaken for each scenario.

Results

TARGET SPECIES AND FISHING EFFORT

The 1970–2016 time series of the median annual trawling effort for the two tiger prawn species are shown in Fig. S5a (Supporting information). This series was derived from the bioeconomic stand-alone model which is part of the management model and aims to satisfy the fisheries management target species objective of maintaining the stock also at MEY (S_{MEY}) (Fig. S5b, Supporting information). At the end of the projection, the resources are at S_{MEY} for tiger prawns, whereas endeavour prawns are still to reach the target. All species are well above the limit reference point – the 5-year moving average of annual stock S_Y should be >50% of S_{MSY} (the stock at MSY).

SPATIAL MANAGEMENT SCENARIOS

The four scenarios (Fig. 3) implemented a range of fisheries closures of various sizes. The resulting amount of fishing area excluded from trawling according to the different spatial management scenarios is shown in Table S6 (Supporting information). In the modelled area of the GoC (396 483 km^2), trawling for tiger and endeavour prawn species is concentrated in only 20·8% of largely the southern and western shallows (<40–45 m) parts of the GoC (Fig. 3, Table S6, Supporting information). It is clear that the closures imposed by the scenarios are small when the whole GoC is considered, since the maximum area closed is 6·6% in the case of the MPAs. The MPA scenario had the largest closure with *c.* 26% of the tiger–endeavour trawling grounds being closed, while the EoT scenario had the smallest closure with only on average *c.* 6% of fishing ground closed.

COMPARATIVE ANALYSIS

Given the large amount of output, results concentrated on the FGs that (i) consist of threatened, endangered or protected (TEP) species, (ii) are habitat-forming or (iii) consist of target or by-product species. The potential trade-offs across different conservation, benthic impact, ecosystems and economic objectives are shown in Fig. 4. All these scenarios are in the context of the same total effort projections and only the spatial footprint of the fishery is potentially different. The EoT scenario is different to all the others, in that closures only occur adaptively when a trigger is reached.

Overall, no single scenario provides the best option across all objectives but, in most cases, a scenario does produce the best outcome for its specifically designed purpose. For example, the ME scenario out-performs the others with respect to the 'at risk' PM (graphs labelled as 'ERA' in Fig. 4), whereas the EoT scenario out-performs in terms of the EoT indicator ('Management') PM in Weipa and on tiger prawn habitats. However, the MPA scenario performs no better than the BC with respect to ecosystem indirect impacts (right-hand column of graphs).

In terms of potential economic impact – as demonstrated through the PMs, area closed and displaced effort ('Fleet') – the MPA scenario closes the most area, but does not necessarily lead to the most median displaced effort, since some of the MPAs included untrawled ground. The highest effort displaced is for the ME scenario. The EoT scenario is often similar to the BC in some regions as it only closed grids when the EoT trigger is reached and this may occur infrequently during the projection (or not at all).

For the EoT scenario, the adaptive closures were designed to keep the EoT indicator in any region >70%. The Weipa region tends to have the lowest EoT PM, the median EoT PM is below 70% for the BC and ME and just above for the MPA scenarios. Here, the EoT performs marginally better than the other scenarios but some simulations still remain below 0·7 due to slow recovery rates. The confidence intervals for the EoT scenario where the indicator is triggered (Weipa, tiger prawn habitats) are generally smaller, highlighting the adaptive nature of this scenario.

There is almost no difference with respect to the percentage 'TEP species' biomass – median and confidence intervals – relative to the BC. For the 'Target and by-product species', small changes occur in the ME (more tiger prawns and fewer bugs – *Thennus* spp.) and EoT (fewer tiger prawns and more bugs) scenarios. The 'habitat-forming taxa' are unaffected by scenario – only tiny changes are shown for the epibenthos. Trawling does not occur on seagrass beds (either due to existing closures or to the beds being too shallow), and so there are no direct effects; nor do the simulations show any marked changes that would reflect indirect trawling impacts cascading through the ecosystem. In terms of whole 'ecosystem' PMs, these again show little change. Since the largest biomass pools are detritus and primary producers these are removed from the PMs but, nevertheless, the same insensitive result was produced.

In general, the highly aggregated PMs were quite insensitive to scenario (even though the impact is also small), whereas small-scale changes both in time and space do occur (see Sections S7 and S8, Supporting information).

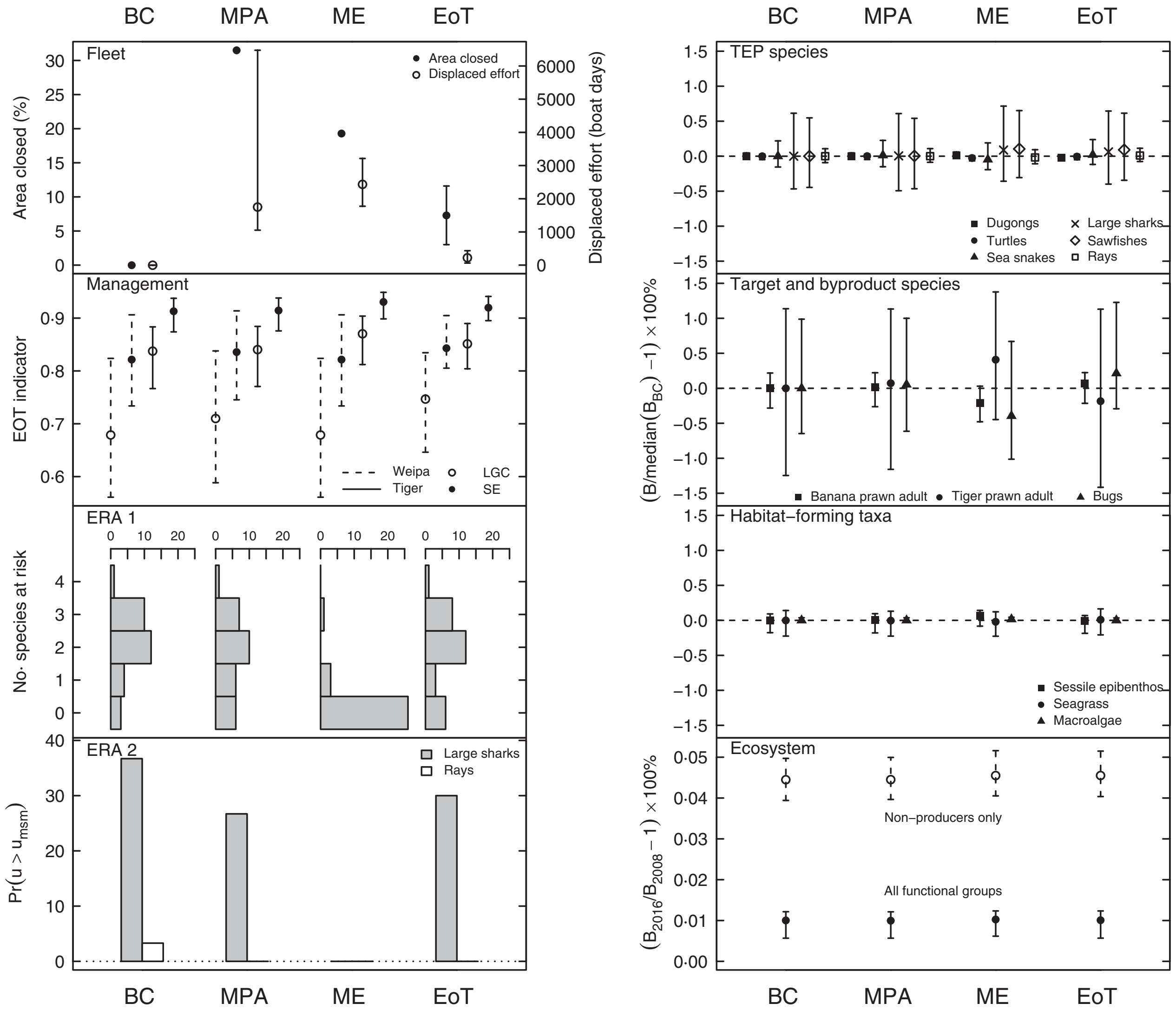

Fig. 4. Performance measures at end of simulation (January 2016) from all components of the management strategy evaluation models: (left panels) fleet, management and ecological risk assessment (ERA) measures; (right panels) EwE measures for specific functional groups (FGs) and for the whole ecosystem. Within each panel, the four strategies (BC, MPA, ME and EoT) are presented from left to right. In all (except ERA) panels, symbols denote median value and whiskers the 5th and 95th percentiles over 30 simulations. Fleet: percentage of area closed (filled circles) and effort displaced from closed areas (open circles). Management: EoT indicator for Weipa subhabitat (dashed lines) and Tiger habitat (solid lines) for large gastropod carnivore (LGC – open circles) and sessile epibenthos (SE – filled circles). ERA 1: frequency of number of species at risk over the 30 simulations. ERA 2: probability (% of simulations) of exceeding the U_{msm} reference value for large sharks (grey bars) and rays (white bars). Threatened, endangered or protected (TEP) species, target and by-product species, habitat-forming taxa: percentage change relative to median BC of total biomass for various FGs. Ecosystem: percentage change relative to end of historical period (beginning 2008) of total biomass for all FGs (filled circles) and all nonproducers (open circles). BC, Base Case; EoT, Effects of Trawling; MPA, marine protected areas; ME, medium effort.

Discussion

Evaluating the ecosystem effects of demersal trawling has attracted substantial research efforts in past decades (Collie *et al*. 2000; Kaiser *et al*. 2003; Thurstan & Roberts 2010). These efforts have been largely biased towards studies in the northern hemisphere, and temperate and deep-water ecosystems where the impacts often have been demonstrated to be substantial (Thrush & Dayton 2002; Heath & Speirs 2012). The results of such studies have strongly influenced the perceptions of trawl fishery impacts.

The comparatively few studies conducted in tropical and subtropical ecosystems have been largely focused on bycatch assessments (e.g. Harris & Poiner 1991; Stobutzki, Miller & Brewer 2001a; Stobutzki *et al*. 2002). Foster & Vincent (2010) made the interesting point that most of these impact studies are concentrated in developed countries, creating a strong knowledge bias relative to ecosystems in developing nations, where most trawling is conducted in tropical soft-sediment habitats.

Australia is a leader in the assessment of ecosystem impacts of fishing in the southern hemisphere – in particular the impacts of trawling – with a range of studies in tropical and subtropical regions (e.g. Pitcher *et al.* 2009), temperate (e.g. Svane, Hammett & Lauer 2009) and deep-ocean ecosystems (e.g. Williams 2007). The findings of such studies have been varied, ranging from irreversible (Williams *et al.* 2010), to strong (Svane, Hammett & Lauer 2009), to less obvious and, in some cases, with undetectable impacts (Burridge *et al.* 2006). Despite these wide variations, trawling does have local and specific impacts, particularly when the fishing grounds overlap with vulnerable biota and the impact assessments are undertaken at the appropriate temporal and spatial scales (Pitcher *et al.* 2009).

In this paper, most of the spatial closures tested were aimed at fisheries management rather than management of biodiversity (the role of the MPA scenario). These are set in the context of a case study with a good governance system, where the NPF is managed in a precautionary manner (Dichmont *et al.* 2008) as demonstrated through its Marine Stewardship Council accreditation. The primary means of managing (in terms of EBFM) the NPF is through input controls (mostly a tradeable gear unit system and season length), habitat and small prawn spatial closures and Turtle Excluder and Bycatch Reduction Devices. Despite this precautionary management, additional spatial closures are used here to manage either key species that remain at risk or ecosystem impacts that have negative effects on particular species, such as TEP species. The aim therefore is to keep the fisheries closures relatively small, given the above, and also to not dramatically affect the economics of the fishery. The results from the operating model show that (i) only minor ecosystem impacts occurred even in the BC, (ii) small dedicated and purpose-designed closures allow for 'at-risk' species to be protected, but these need to generally remove some high-effort regions to be effective (therefore requiring industry to lose some prized fishing ground), (iii) the MPA closure displaced the second most effort (as some of the closed grids were in unfished areas) with very little difference from the 'at-risk' targeted (ME) closures in terms of nonbiodiversity measures and (iv) the adaptive EoT closure closed the least grounds but was not effective at reducing at-risk species or keeping benthic impacts above a threshold because it was rather insensitive.

Management strategy evaluation is a useful tool for investigating conflicting objectives (Sainsbury, Punt & Smith 2000) such as simulated in this study: obtaining MEY, a sustainable biological target species, no species at risk of overexploitation or extinction, minimum impacts on the seabed and no major ecosystem cascades. However, to fully investigate all these conflicting objectives requires an extremely complex (especially operating) MSE model. Several options are available, most notably Atlantis and *In Vitro* (http://atlantis.cmar.csiro.au/) which are ecosystem models purpose-built for MSE. However, these tend to compromise on spatial scale. This means that, for small-scale benthic impacts, these models are not particularly useful, but are extremely useful for multiple use/fishery cases at the ecosystem or subsystem level. In this study, only a range of tools could provide for all these objectives that operate at different spatial and temporal scales. For example, Ecospace works at annual time-steps and 6-min grids – the operating model used in this MSE – and is typically regarded as being useful for relative scenario analysis (as is the case here), but not for tactical management, hence its use as an operating model but not a management model. The compromise here was that not all tools within the operating model were connected in a way that best allowed bias and variance to be propagated through the process as they could in fully connected submodels. Nor could appropriate sensitivity tests be undertaken, since it would be far too time consuming and would not fully investigate all combinations in any case. This means that our estimated uncertainty is likely to be understated, and despite being avoided as much as possible, the potential for different assumptions of the same FGs in the different submodels providing different and counter-intuitive results has increased. In spite of these issues, it clearly demonstrated that this tool can explicitly highlight the wide range of trade-offs across all the above-mentioned objectives – beyond the norm for most MSEs, but well within what is expected from EBFM. The full model moves part way to the view of Armstrong & Falk-Petersen (2008) that habitat effects should more directly be included in, for example stock assessments. In the ideal world all these tools would be fully integrated, providing a goal for researchers in this area to aspire to.

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Supporting Information

Additional Supporting Information may be found in the online version of this article.

Section S1. Ecopath, Ecosim and Ecospace models.

Table S1a. Parameters of the 53 functional groups in the Ecopath model.

Table S1b. Habitats in the Ecospace model.

Section S2. Ecological risk assessment model.

Table S2. Parameters of the functional groups in the ecological risk assessment model.

Section S3. Habitat indicator in effects of trawling management model.

Section S4. Performance measures.

Table S4. Performance measures calculated from the MSE or used for other derived measures.

Section S5. Stock- and effort-related performance measures.

Fig. S5a. Time series of historical and projected fishing effort.

Fig. S5b. Time series of stock-related performance measures from the bioeconomic model.

Section S6. Spatial closures.

Table S6. The extent of closures by area and percentage for the various spatial management scenarios.

Section S7. Detailed operating model results at fine spatial resolution.

Table S7. Functional groups used for reporting detailed spatial and temporal patterns.

Fig. S7. Predicted changes at 2016 of the relative biomass of the functional groups.

Section S8. Detailed operating model results at fine temporal resolution.

Fig. S8. Time series of changes in predicted biomass relative to the Base Case at habitat resolution.

Section S9. List of references cited in the Supporting Information.