

A dynamic mass-balance model for marine protected areas

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Abstract

A modified ECOSIM model was used to investigate the impact of establishing marine protected areas (MPAs) in ecosystems defined by existing ECOPATH models. The impact of MPAs of various sizes was simulated, and changes in biomass and catch over a range of years observed. The response of biomass and catch to MPA size depended on the time period examined. For some ecosystem groups, the initial response was negative, but for all groups there were increases after 10 years. The greater the biomass exchange rate across the MPA boundary, the larger the MPA required to increase biomass levels. Within the range of exchange rates simulated, the maximum increases in catch and overall biomass levels were reached when 20% of the system was protected.

Keywords ecological models, ECOPATH, ECOSIM, fisheries, marine reserves; protected areas.

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Introduction

One approach used widely to manage complex marine systems is the establishment of no-take marine reserves or marine protected areas (MPAs), which over the last 30 years have been established throughout the world. McNeill (1994) examined the selection and design of MPAs in Australia and found that few studies had tested hypotheses concerning the effects of location, size or shape. These are especially important when proposed MPAs are expected to contribute to sustainable fisheries management, and when their establishment will reduce access by traditional resource users.

Many scientists and managers now accept that an ecosystem approach is needed to evaluate the establishment of marine or fish reserves (Done and Reichelt 1998). Management at the ecosystem level requires information on the resources and their interdependencies, including potential trophic interactions. Several tools are available to create ecosystem models; the most widely used for marine

systems is the ECOPATH mass-balance approach introduced by Polovina (1984) and expanded by Christensen and Pauly (1992). ECOPATH uses assumed mass-balance (production/consumption) relationships to describe equilibrium trophic flux patterns. Such models were of limited interest to most fisheries managers until recently when Walters *et al.* (1997) introduced ECOSIM, which provided a means of using ECOPATH assessments in a dynamic way. ECOSIM allows scientists and managers to vary previously static variables such as fishing mortality rates, and to predict transient and equilibrium changes in catch rates as well as biomass. This tool provides managers with the means to investigate the potential effects of various management strategies on groups described in ECOPATH models, including those of commercial significance.

While ECOSIM offers some facilities to managers, it does not provide a means of describing the spatial relationships of biomass and fishing mortalities, which are required to examine the potential impacts of MPAs. This paper describes a simple modification

to ECOSIM that allows the biomass of ECOPATH groups to be partitioned into two portions with exchange processes operating between them. One portion is assumed to be within an MPA, and is subject to lower levels of fishing mortality (at least for some groups) than the other portion. Our modification to ECOSIM allowed us to observe the transient impact of MPA 'size' and of biomass exchange rates on the biomass and catches of ECOPATH groups.

Methods and materials

Simulations were performed using a modified version of ECOSIM (Walters *et al.* 1997), which is based on the equation:

$$dB_i/dt = f(B_i) - Mo_i B_i - F_i B_i - \sum_{j=1}^n Q_{ij} \quad (1)$$

where B_i represents the biomass of ECOPATH group i , Mo_i is the natural mortality rate of group i , F_i is the average fishing mortality rate of group i , and Q_{ij} is the consumption of group i by group j predators; $f(B_i) = g_{ij} \sum_{j=1}^n Q_{ij}$ represents the growth rate as a function of B_i , where g_{ij} is the growth efficiency of group i consuming group j . ECOSIM used the mass-balanced ecosystem models developed using ECOPATH 2.0 (Christensen and Pauly 1992).

Modifications were made to the ECOSIM program described above to allow all ECOPATH biomass pools (B_i) to be split in a quasi-spatial way. The biomass of each ECOPATH group was divided into two subpools that were treated as if they were spatially separated and subject to different fishing patterns. This allowed for one subpool to be treated as if it were in an MPA and not subject to fishing, while the biomass of the other subpool was subject to fishing mortality. The rate of exchange or transfer of biomass between the two subpools could be set independently. At the start of each simulation, the biomass of each group was distributed between the areas in proportion to the relative size of the protected area; for example, when 5% of the modelled area was an MPA, then 5% of the biomass described in the ECOPATH model for each pool, such as 'Large Demersal Fishes,' was initially placed in the MPA subpool, and the remaining biomass was placed in the non-MPA subpool and subject to fishing mortality.

All ECOPATH groups were protected from fishing within the MPA. The biomass reported is the total from the MPA and non-MPA areas. Fishing

mortality rates were held constant, simulating a displacement of the total fishing effort, not a reduction as the MPA size was increased. Catches reported are in biomass units per unit area.

At equilibrium the change in biomass outside an MPA, dB_{out}^i/dt , of any ECOPATH group i (regardless of the MPA's size) is balanced by the change in biomass within the MPA, dB_{in}^i/dt , therefore

$$dB_{out}^i/dt = R_{out}^i B_{out}^i = dB_{in}^i/dt = R_{in}^i B_{in}^i \quad (2)$$

where R_{out} and R_{in} are the proportion of biomass transferred from outside and inside the MPA, respectively. Because the movement of biomass out of the MPA is proportional to its perimeter (roughly the square root of its area),

$$R_{out}^i = \frac{T_i}{\sqrt{S}} \quad (3)$$

Here, S is the proportion of the modelled area that is within an MPA, and T_i is a user-supplied exchange rate for group i with units time^{-1} . The biomass of group i within the MPA

$B_{in}^i = S B_{total}^i$ and that outside the MPA

$B_{out}^i = (1-S) B_{total}^i$, therefore at equilibrium

$R_{out}^i S B_{total}^i = R_{in}^i (1-S) B_{total}^i$, which means

$$R_{in} = R_{out} \frac{S}{1-S} \quad (4)$$

ECOPATH models representing a variety of marine environments have now been published. We were most interested in heavily exploited systems where the impact of protecting part of the biomass from fishing through the use of an MPA would be most obvious. We chose an ECOPATH model (Thai10) that represents shallow water areas (0–10 m) of the Gulf of Thailand (Model A, Pauly and Christensen 1993). Although all model groups were included in the simulations, for simplicity the results from only four representative fished groups are presented.

Results

MPA size and simulated period

The response of the Thai10 ECOPATH model to differing MPA sizes for a 10-year period was simulated using a moderate biomass exchange rate (Fig. 1). After one year, biomass for three of the four ECOPATH groups followed showed slow increases with

MPA size (the crustacean group responded negatively; Fig. 1a). By year three, the biomass of another group (molluscs/jellyfish) was responding negatively. The response of biomass *ECOPATH* groups to MPA size, however, differed depending on the time period examined, and after 10 years the simulations predicted that there would be a marked increase in biomass of most groups. This was particularly evident for crustacean and mollusc/jellyfish, two groups that responded negatively in simulations of shorter time periods (Fig. 1c). With MPA sizes exceeding 15% there were no further increases in predicted biomass.

Changes in catch rates with the proportion of MPA size largely paralleled those predicted for biomass (Fig. 1d–f). Initially (after one year), the catch of all groups (except intermediate predators) was restricted by MPA size. By year 10, catches of all groups increased with MPA size, reaching a maximum with MPAs of 10–15% in size (Fig. 1f).

MPA size and exchange rates

As expected, the rate of biomass exchange assumed in simulations had an impact on biomass and catch predictions with differing MPA size after a 10-year period. The biomass of most groups showed a very peaked response at MPA sizes of 5–10% when a slow exchange rate was assumed (Fig. 2a). With a more

rapid exchange rate the response of biomass was less peaked, and occurred at larger MPA sizes (10–15%; Fig. 2b). This trend continued, and a rapid biomass exchange rate required even larger MPAs to reach the biomass maxima of slower rates, after which biomass continued to increase slowly (Fig. 2c).

Catches generally followed the biomass pattern of response (Fig. 2d–f). Catches of molluscs/jellyfish continued to peak at smaller MPA sizes than did other groups regardless of the exchange rate used. The peaked responses of slow exchange rates became more gradual and occurred at larger MPA sizes. Unlike biomass, the catches of most groups remained peaked for all exchange rates examined, predicting that even at rapid exchange rates larger MPAs would not allow fishers sufficient access to enhanced biomass levels to maintain catches.

Discussion

There have been several approaches used to determine the optimum MPA size for exploited species; however, most have ignored trophic interactions. Not surprisingly, optimal MPA size in this context is related to intrinsic stock production rates, the prevailing harvest rates for the species involved (Mangel 1998) and the interchange of fishes between the fished and unfished areas for all the life history stages (Russ *et al.* 1992; Man *et al.* 1995;

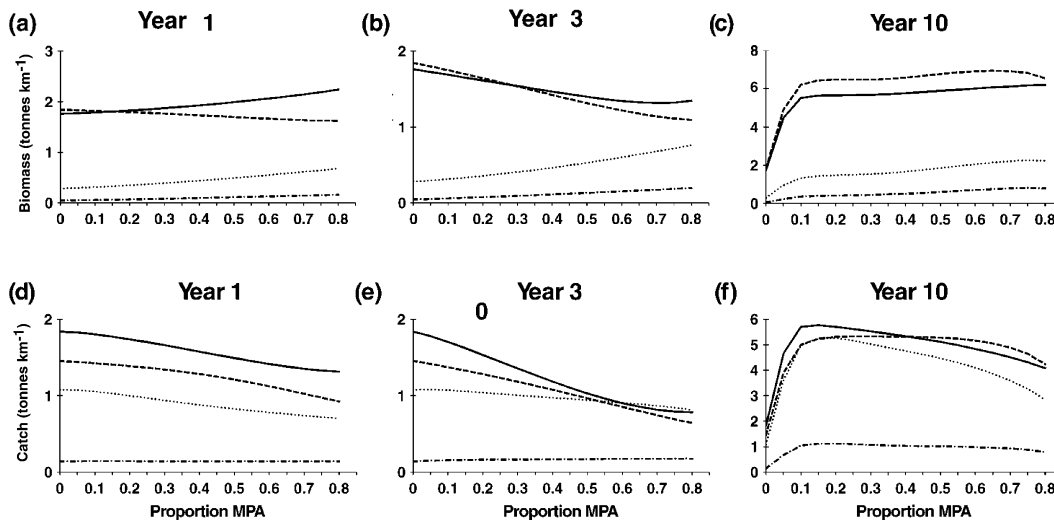


Figure 1 Effect of proportion of biomass from *ECOPATH* model Thai10 included in a marine protected area (MPA) on total biomass (tonnes km⁻²) values after (a) 1 year, (b) 3 years and (c) 10 years and catch (tonnes km⁻²) after (d) 1 year, (e) 3 years and (f) 10 years using a biomass mobility $T = 1$. Selected *ECOPATH* model groups shown include: molluscs/jellyfish (solid line), crustaceans (dashed), small demersal fishes (dotted), and intermediate predators (dash-dot).

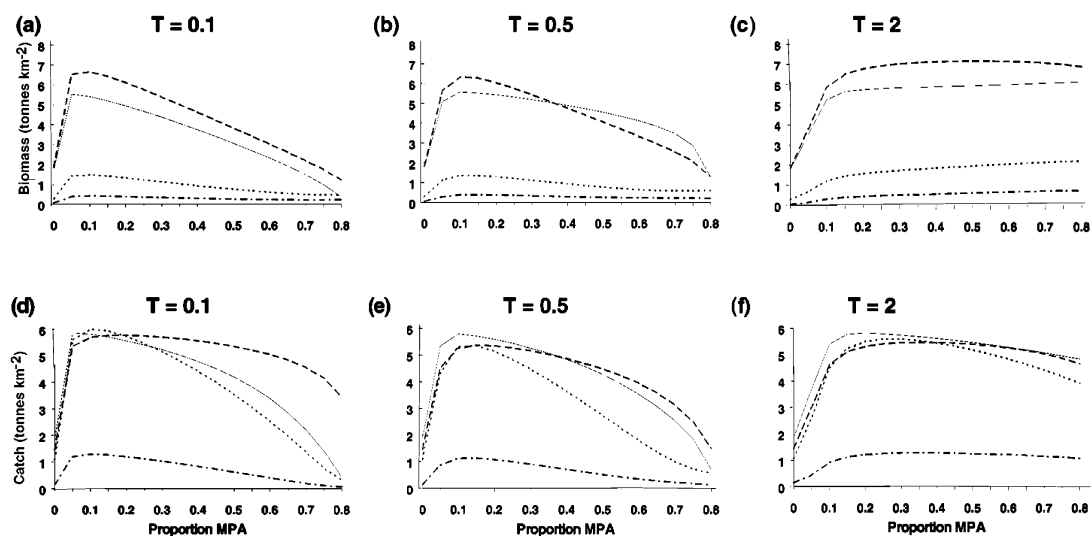


Figure 2 Effect of proportion of biomass from *ECOPATH* model Thai10 included in a marine protected area (MPA) on biomass (tonnes km⁻²) values after 10 years using a biomass mobility, T of (a) 0.1, (b) 0.5 and (c) 2.0, and on catch (tonnes km⁻²) using a T of (d) 0.1 (e), 0.5 and (f) 2.0. Selected *ECOPATH* model groups shown include: molluscs/jellyfish (solid line), crustaceans (dashed), small demersal fishes (dotted), and intermediate predators (dash-dot).

Gu nette and Pitcher 1999). The impact of MPAs on whole community structures, not just single species, has now been demonstrated in natural systems (Russ and Alcala 1989) and should not be ignored.

Models using an ecosystem approach to management need both temporal and spatial components. Most models have a temporal component but few have the facilities to partition and manipulate space. Indeed, Done and Reichelt (1998) pointed out that defining appropriate spatial scales was a major problem in adapting ecosystem approaches to fisheries' management problems. The approach used here applies in a range of reserves as long as the management system includes a component that excludes fishing.

The case study presented here suggests that an MPA that protected 10–15% of the biomass of the Thai10 system from fishing would increase both biomass and catches of the groups followed over a 10-year period. If the exchange rate was slowed, the reserve area could be reduced, but if the exchange rate was faster then the reserve size should be increased to protect 20% or more of the biomass. In practice, there is little information on rates of movement or biomass exchange rates for marine systems; the required information about immigration and emigration rates is often lacking.

It is worth noting that the movement of larvae or prerecruitment individuals can occur at different rates than that of older individuals. In such circumstances it would seem warranted to use the split-pool option of *ECOSIM* that allows different developmental stages to be split but linked through a delay-difference model structure (Walters *et al.* 1997). Although this was originally conceived to allow for developmental changes in diet it could also allow for changes in movement rates.

Managers are often concerned about detecting changes caused by the introduction of an MPA, and whether the initial indications will be favourable. This is especially so if they have to justify management changes that have denied local fishers access to traditional stocks. We have shown that although the initial changes in biomass and catches of some groups (such as the mollusc/jellyfish of model Thai10) can be negative, this trend can continue for three years or more. By 10 years, there was an increase for all groups after which there was no further change in biomass or catch.

Size is only one consideration in the selection of an appropriate MPA. Edge effects can become significant with small MPA sizes. The rapid exchange of biomass between the MPA and the fishery can reduce biomass levels within the MPA by creating a gradient, with higher biomass levels

found toward the centre of the MPA (Rakitin and Kramer 1996). The strength of the biomass gradient is related to the exchange rate. At low exchange rates it is possible to create a strong gradient between biomass levels in the fishery and those in the centre of the MPA. If, however, the MPA is small enough and/or the biomass exchange rate (= mobility) of species is great enough, then there can be no protection from fishing. MPA shape can also have a significant impact, if the shape of the MPA is narrow and the net movement of animals is perpendicular to its long axis then little protection will be afforded. In contrast, in situations where the animals tend to move along the primary axis of the MPA, staying within its boundaries for more of the time, then even narrow reserves would yield results. This may describe the majority of coastal reef reserves.

Investigation of further aspects of establishing MPAs, such as their location and shape, will require a true spatial model. Such a model could include data on habitat richness, ocean currents and other factors affecting species distribution and interactions. In the mean while, our approach can be used to represent MPAs within the context of an ecological model, and to investigate the effects of MPA size on the biomass and catch of the groups represented.

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References

Christensen, V. and Pauly, D. (1992) ECOPATH II—a software for balancing steady-state models and calculating network characteristics. *Ecological Modelling* **61**, 169–185.

- Done, T.J. and Reichelt, R.E. (1998) Integrated coastal zone and fisheries ecosystem management: generic goals and performance indices. *Ecological Applications* **8**(Suppl.), S110–S118.
- Guénette, S. and Pitcher, T.J. (1999) An age-structured model showing the benefits of marine reserves in controlling overexploitation. *Fisheries Research* **39**, 295–303.
- Man, A., Law, R. and Polunin, N.V.C. (1995) Role of marine reserves in recruitment to reef fisheries: a metapopulation model. *Biological Conservation* **71**, 197–204.
- Mangel, M. (1998) No-take areas for sustainability of harvested species and a conservation invariant for marine reserves. *Ecology Letters* **1**, 87–90.
- McNeill, S.E. (1994) The selection and design of marine protected areas: Australia as a case study. *Biodiversity and Conservation* **3**, 586–605.
- Pauly, D. and Christensen, V. (1993) Stratified models of large marine ecosystems: a general approach, and an application to the South China Sea. *Large marine ecosystems: stress, mitigation and sustainability* (eds K. Sherman, L.M. Alexander & B.D. Gold), pp. 148–174. Am. Assoc. Advan. Sci., Washington DC, USA.
- Polovina, J.J. (1984) Model of a coral reef ecosystem. I. The ECOPATH model and its application to French Frigate Shoals. *Coral Reefs* **3**, 1–11.
- Rakitin, A. and Kramer, D.L. (1996) Effect of a marine reserve on the distribution of coral reef fishes in Barbados. *Marine Ecology Progress Series* **131**, 97–113.
- Russ, G.R. and Alcala, A.C. (1989) Effects of intense fishing pressure on an assemblage of coral reef fishes. *Marine Ecology Progress Series* **56**, 13–27.
- Russ, G.R., Alcala, A.C. and Cabanban, A.S. (1992) Marine reserves and fisheries management on coral reefs with preliminary modeling of the effects on yield per recruit. *Proceedings of the Seventh International Coral Reef Symposium, Guam, 1992* (2). R.H. Richmond, ed. University of Guam, Marine Laboratory, Mangilao, Guam. 978–985.
- Walters, C., Christensen, V. and Pauly, D. (1997) Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. *Reviews in Fish Biology and Fisheries* **7**, 139–172.