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Marine reserves
Designing cost effective options

Kenton Lawson and Peter Gooday
Australian Bureau of Agricultural and Resource Economics

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In analysing any particular marine planning option, policy makers will be concerned with the economic impacts of that option. In particular they will be concerned with the cost effectiveness and the scope for tradeoffs between the environmental benefits and social costs. To make these assessments, information is required on the economic costs of the proposed restrictions and the effective integration of this information into the marine planning process.

In Australia, such assessments were made for forestry as part of the Regional Forest Agreements. The process involved detailed assessments of the biological and economic implications of alternative forest conservation options. As part of this process, ABARE collaborated with biologists to develop a computer based reserve design tool that allocated planning units (small, relatively homogeneous areas that partition the study region) to alternative land uses. This was done by taking into account specified quantitative environmental objectives, the opportunity cost of land use changes and a measure of the shape of the reserve system. The optimisation was performed using a simulated annealing algorithm. The tool can be linked to a GIS based decision support system so that the computer generated designs may be refined and brought into stakeholder negotiations.

The tools that were developed are quite general and may be fairly readily applied to other problems. In this paper, the application of the methodology to the design of marine protected areas is explored.

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ABARE
Innovation in Economic Research
GPO Box 1563 Canberra 2601 Australia
Telephone +61 2 6272 2000 • Facsimile +61 2 6272 2001
www.abareconomics.com

Introduction

While much of the attention associated with marine reserves has been directed at their use to manage fisheries resources, marine reserves are also used as a tool for managing a variety of other risks and externalities associated with marine resource uses. The net benefits of areas to include in a marine reserve system may be relatively obvious for mitigating some risks, such as the protection of important spawning grounds. However, in many cases the net benefits may be less apparent and alternative reserve configurations will need to be considered. For example, there will usually be many configurations of area closures that could be used to meet biological conservation objectives within an area, some more costly to commercial concerns than others.

Policies such as the *National Representative System of Marine Protected Areas* in Australia provide an example of this type of reserve design problem. The system aims to '… establish and manage a comprehensive, adequate and representative system of marine protected areas to contribute to the long term ecological viability of marine and estuarine ecosystems, to maintain ecological processes and systems, and to protect biological diversity at all levels' (Commonwealth of Australia 1998, p. 45).

In designing systems to manage the costs and risks associated with using marine resources the objective is generally to maximise some notion of the change in social welfare from the status quo. Ideally decisions on the management regimes applying to any relevant commercial or recreational fisheries and decisions on the design of a marine reserve system should be considered jointly. The use of marine reserves should be considered as a complement to traditional fisheries management instruments (for example, output and input controls) rather than as a substitute.

The manner in which a measure of the change in social welfare could be maximised is most transparent in cases where all the major values, including environmental values, can be adequately monetised.

The monetary value of benefits and costs of alternative uses of marine resources can be difficult and costly to estimate. However, owing to a lack of market values, it is generally the monetisation of the environmental benefits of risk reduction that presents the most difficulty. It is possible, in some cases, to obtain estimates of key indicators of environmental risks (for example, the probability of survival of a viable population of a specific species of flora or fauna); however, it has generally proved problematic to transform these estimates into monetised values.

When it is not possible to monetise all values associated with risk reduction, an alternative approach is to specify environmental standards and to then design management regimes, which may include a zoning system, that achieves these standards at least cost.

An important practical issue to be considered is the large number of potential zoning systems. In order to better explore this decision space, effective use may be made of decision support tools that incorporate computer algorithms for reserve design. In this paper a methodology for effectively integrating economic and environmental objectives that is able to incorporate relatively sophisticated reserve design objectives is presented. This decision support tool can relatively quickly generate an optimal zoning system as the solution to a constrained optimisation problem.

While it may be possible to choose a reserve system to minimise the cost of meeting specified environmental standards, there is no guarantee that the environmental standards used are socially optimal. In order to explore the scope for tradeoffs it is necessary to consider a range of other environmental standards and to find least cost zoning systems for each standard. These least cost zoning systems, together with measures of their impacts on a range of use and non-use values, are key inputs into negotiations over which system is to be implemented.

The development of consistent procedures for assessing the cost of environmental standards is then required. These costs need to incorporate implementation and management costs. The accurate measurement of costs can be a difficult and costly undertaking. As a number of proxies for the cost associated with restricting access to an area exist, the questions of how appropriate those proxies are for choosing the most cost effective reserve design and for reporting on the cost of a specified set of environmental targets are important.

Using economic information to assist in reserve design

ABARE has recently been involved in examining the costs to the logging and sawmilling industries associated with the introduction of new conservation reserves in Australia's native forests. This work provides useful insights into the techniques that are available to assist marine reserve planning processes.

The Regional Forest Agreement (RFA) process in Australia involved assessing the natural, cultural, economic and social values of Australia's forests. RFAs are intended to provide stability by establishing a sustainable resource base for industry, while at the same time ensuring the protection of Australia's biodiversity, old growth and wilderness through a comprehensive, adequate and representative reserve system and complementary off-reserve management (Commonwealth of Australia 1992).

In providing economic advice to assist the RFA process ABARE developed the Forest Resource Use Model (FORUM), a regional linear programming model that can be used to simulate the interactions between regional forest resources, wood based forest industries and final wood product markets. The model is described in Dann et al. (1997).

FORUM was used to calculate the logging and sawmilling related costs of proposed forest reserves by comparing the estimated value of resource rents accruing to land and tree resources under a baseline scenario (projected timber availability through time under current land access arrangements) with those estimated using alternative assumptions regarding land access and timber availability. Simulations were generally run using a 20 year timeframe. In order to account for industry developments over time a number of industry investment opportunities were identified and incorporated in the model.

The use of FORUM was initially restricted to examining the impacts of alternative reserve options after those options had been identified by various stakeholder groups. The performance of each option against environmental targets was examined using separate models. Using this information (and information gathered on cultural and social implications) the options were refined and, if necessary, the economic and environmental implications were assessed again. Negotiations between stakeholder groups determined the final reserve configurations.

This type of process has the potential to be inefficient as it is difficult for stakeholder groups to be able to design reserve systems that provide the desired level of protection for forest resource values of interest to them while maximising values associated with other uses. That is, there is no guarantee that the forest reserve options designed to protect specific environmental values do so at least cost or that reserve options developed by forest industry groups provide the maximum environmental benefits possible.

In an attempt to mitigate this problem ABARE collaborated with biologists to develop a reserve design tool that incorporated economic and biological information about the region under consideration. This work built on the previous reserve design literature (Possingham, Ball and Andelman 2000), with the key advance being the inclusion of the costs of reservation. The resulting optimisation problem was solved using a ‘simulated annealing’ method (Kirkpatrick, Gelatt and Vecchi 1983).

In this approach the study area is partitioned into a large number of planning cells. Each cell can be allocated to one of a number of management regimes — for example, either closed to extractive uses or not. For each cell there is a number of environmental attributes that allow the effectiveness of the reserve system against a range of environmental conservation objectives to be assessed. In addition, for each cell there is a measure of the cost of reservation.

In the RFA process a key goal agreed by governments was to reserve 15 per cent of the area of specified regional forest types that existed before European settlement. In its simplest formulation this optimisation problem — to minimimise the cost of achieving the targets through allocating cells to the reserved category — increases in computational complexity exponentially with the size of the problem. This means that exact mathematical programming methods are not feasible for large reserve design problems and explains the use of heuristic methods, such as simulated annealing.

A major problem with the reserve design approach outlined above is that it fails to account for the location of selected reserve areas in relation to each other. This has the consequence that optimising the above problem generally yields highly fragmented reserve systems (as illustrated in figure 3 below). Such a reserve system would be very expensive or infeasible to manage and of dubious conservation value. An alternative model would be based on the assumption that management costs depend on the boundary length of the reserve system. The objective function can then include these management costs. While this addition does make the problem nonlinear it does not significantly raise the computational burden when using simulated annealing. In order to estimate the minimum possible cost of achieving a given set of environmental targets this parameter can be set to zero.

Application to a marine context

There are a number of difficulties associated with applying the methodology developed for the case of forests to a marine context. The main difficulty is in estimating the cost of excluding fishing from an area. (While the focus in this paper is on the costs to the community of excluding commercial fishing it is important to recognise that costs associated with restricting access to marine resources may also be significant, and difficult to quantify.)

The estimation of the cost of excluding an area from production can be considerably more complex in the fisheries example than in the case of forest reserves. For example, the opportunity cost of including an area in the reserve system may need to account for the fact that in response to a change in resource access arrangements fishing activity may be redirected to other areas. It would also account for any changes in the distribution (and migration patterns) of fish throughout the region.

This is further complicated by the fact that marine reserve planning is part of the larger problem of the management of marine use in an area. For example, the management of fisheries will generally involve combinations of input and output controls on fish harvesting across time and space. A marine reserve system will, in general, also affect fish harvesting (and the revenues and costs associated with harvesting) across time and space. The consequence of this interdependence is that solving the spatial aspect of the marine reserve planning problem independent of other dimensions of the problem is likely to lead to inefficient outcomes.

To estimate opportunity costs to this level of detail would require extensive biological and economic information (including information on the management regime in place) and could be expected to be a difficult and expensive undertaking. In most cases the data required to allow a calculation of the full cost of changing access regulations will not be available. However, it is likely that the use of alternative cost measures may provide significant benefits to the reserve design process compared with not incorporating any measure of cost. Using estimates of average annual rents in existing marine uses without regard to the 'dynamic' implications of changing access regulations makes the task somewhat simpler. In the rest of this paper the problem of *minimising* annual fishing rents forgone *plus* annual management costs associated with reserve boundary length *subject to* achieving minimum biodiversity targets will be considered. This problem can be represented as:

Minimise: $$\sum_i x_i r_i + \lambda l$$

Subject to: $$\sum_i a_{ij} x_i \geq b_j$$

where

- x_i indicates whether cell i is closed (1) or open (0) to fishing,
- r_i is the average annual rent earned from fishing in cell i,
- a_{ij} is the amount of biodiversity component j in cell i,
- l is the boundary length of the reserve system,
- λ is a parameter indicating the management cost per unit of boundary length, and
- b_j is the required amount of biodiversity component j for the marine area as a whole.

This is a minimal specification for integrating economic and environmental information in reserve design. It is implicitly assumed in this specification that fishing revenues and costs in any one cell are independent of the status (reserved or not) in any other cells — that is, 'static' average annual rents are assumed. In practical applications it is necessary to determine an appropriate specification in light of the nature and scale of the problem at hand as well as the cost of information and time constraints.

To estimate a 'static' value, fishing revenues and costs for specific areas within the region are required. The availability of data to estimate actual revenues and costs by area varies significantly between fisheries, particularly the data required for estimating costs. As a result, the proxies that are available for resource rent will vary from fishery to fishery. In the example provided below, results obtained with several proxies are compared.

It may be necessary to specify a number of marine reserve categories to allow for the fact that marine areas can provide a wide variety of uses simultaneously. For example, several types of commercial fishing (trawl, line, pot), recreational pursuits, oil and gas production, shipping and conservation activities can all occur within a marine area simultaneously.

An illustrative example: the northern prawn fishery

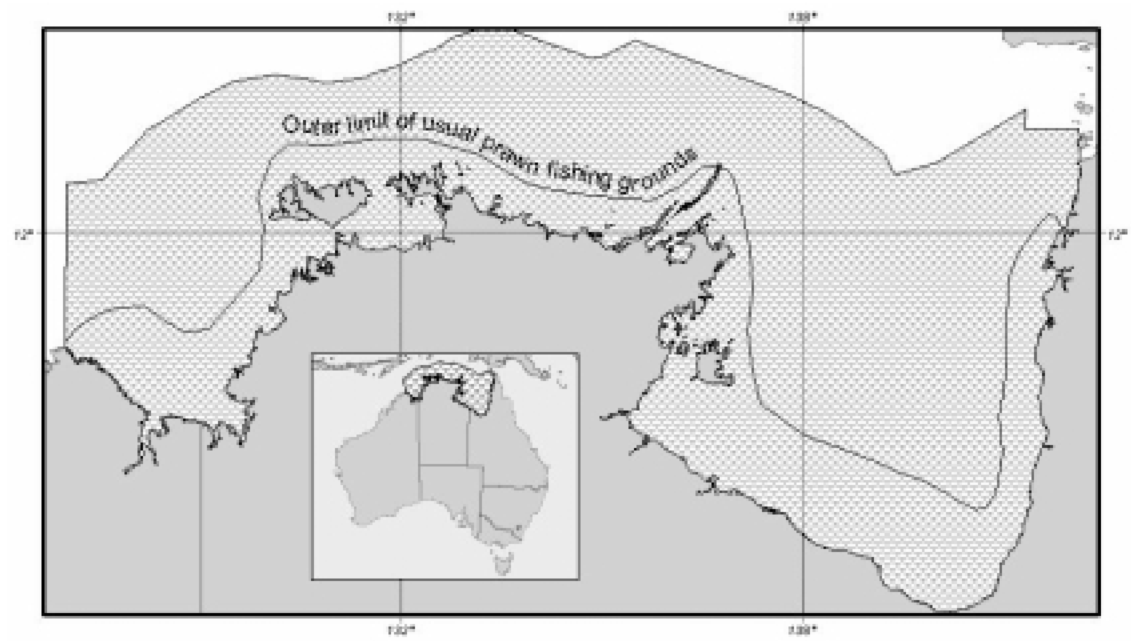
The northern prawn fishery is located in Commonwealth waters in the Australian Fishing Zone and is bordered by Cape York (Queensland) in the east and Cape Londonderry (Western Australia) in the west (figure 1). It is the largest fishery by area in Australia, at over one million square kilometres. The fishery targets nine commercial species of prawn including white banana (*Fenneropenaeus merguiensis*), red legged banana (*F. indicus*), brown tiger (*Penaeus esculentus*), grooved tiger (*P. semisulcatus*), blue endeavour (*Metapenaeus endeavouri*) and red endeavour (*M. ensis*). Bycatch includes squid, scallops and bugs (AFMA 1999).

Commercial prawn species have a life span of up to two years. Juvenile prawns live in coastal and estuarine areas in beds of seagrass or mangrove lined creeks. After one to two months on the nursery grounds, the prawns move offshore into the fishing grounds. While banana prawns reach commercial size around six months of age, tiger prawns are usually required to be larger for the market, reaching commercial size at around 9 to 12 months of age.

Management of the fishery has involved the use of input controls. The key feature of the current management system is limited entry, permanent and seasonal closures, gear and boat limitations, controlled season start and, in recent years, a reduction in the size of the fleet (assisted by an effort unit buyback scheme).

The fishery is closed during the winter months to reduce fishing effort on the pre-spawning stock of tiger prawns. Until recently, this closure was usually from 15 June until 1 August. The end of year closure, usually from 1 December until 30 March is aimed at

Figure 1: **Northern Prawn Fishery study area**



preventing the capture of small tiger prawns that begin to recruit to offshore grounds at about this time. It also protects small banana prawns which appear in the new year.

Northern prawn fishery environment

The entire northern prawn fishery area is relatively pristine because of its isolation and relatively limited coastal development. The area is one of the few remaining relatively undisturbed shallow water ecosystems in the tropical world (Hill 1994a). The coastal area of the Gulf of Carpentaria is shallow and has high turbidity and runoff from the land. Production of many species is largely limited to this coastal area. In contrast, the offshore area of the Gulf is deeper with lower productivity and overlies a bottom that generally supports fewer animals (Hill 1994b). The diversity of habitats in the managed area of the northern prawn fishery is demonstrated by the identification of fifteen bioregions in this region (IMCRA 1998).

Prawn trawling clearly affects prawn stocks, some bycatch populations and some of the organisms attached to the bottom (Taylor and Die 1999; Stobutzki et al. 2000; Poiner and Harris 1996; Poiner et al. 1999). Direct effects of trawling include scraping and ploughing of the substrate, sediment resuspension, increased turbidity, destruction of benthos and epibenthos and the dumping of processing waste. Changes in habitat may lead to reductions in populations of animals that are dependent on it for shelter, food or as a spawning area. Indirect effects include post-fishing mortality and long term trawl induced changes to food chains and to other marine flora and fauna.

Research on the effects of trawling on the Great Barrier Reef has shown that the impacts of trawling on benthic habitats, target species, bycatch and biodiversity depend on two factors: the intensity of trawling; and, the vulnerability of the biota in the area being trawled (Poiner et al. 1999). The vulnerability of a species to trawling depends not only on the quantity removed by a trawl, but also on the recovery rate between trawls and the location of trawling in relation to where organisms live.

Data

For the purpose of illustrating the application of the reserve design approach described above to the northern prawn fishery, a number of proxies for resource rent were estimated. In addition, proxies for environmental attributes of the area were calculated.

In estimating the proxies for resource rent, cost estimates were based on data obtained from ABARE economic surveys of the northern prawn fishery and on northern prawn fishery logbook data (catch and effort by location). Revenue estimates were based on logbook data and on price data contained in ABARE (2000). Data were available for the period 1991-92 to 1996-97 and the resource rent proxies were constructed on a six minute grid. Of the 6533 six minute grid cells within the northern prawn fishery 1894 cells were fished during the period 1991-92 to 1996-97. The outer limit of the prawn fishing grounds are indicated on figure 1. It would appear from figure 1 that there may be considerable scope for sea floor closures to achieve environmental targets at a small cost. In practice, this would be modified with information on other affected industries.

In this example the fact that the logbook data were measured on a six minute grid explains the choice of planning cells. However, the approach can use any cell shape. In terrestrial applications the approach has generally used forest compartments or subcatchments as planning cells.

In producing a proxy for environmental attributes, available bathymetry, slope and sediment data of the area were used. Cluster analysis was used to cluster depth, slope and two dimensions of sediment type into 100 clusters (that is, 100 environmental components or domains were constructed). The area of each component in each six minute grid cell was then calculated. For the sake of this illustrative application, each component is assumed to be equally vulnerable to trawling and equally valuable in ensuring the persistence of the regions' benthic biodiversity. In practice such environmental components would be calculated through sampling and spatial modeling with separate representation targets for each biogeographic subregion of the study area. These changes would increase the number of constraints to the optimisation problem and thus increase the opportunity cost of achieving any particular level of environmental targets.

Simulations

Simulations were run using several proxies for the cost of excluding fishing from a cell.[1] The required amounts of each biodiversity component were specified as a proportion of the total area of the biodiversity component in the study area. These are initially set at 20 per cent.

The proxies used for each cell were:

a) Average annual net returns. Calculated as revenue *less* operating costs (excluding interest and lease payments) *less* the opportunity cost of capital.

b) Average annual total revenue.

c) Catch per unit effort

d) Average annual effort (number of days)

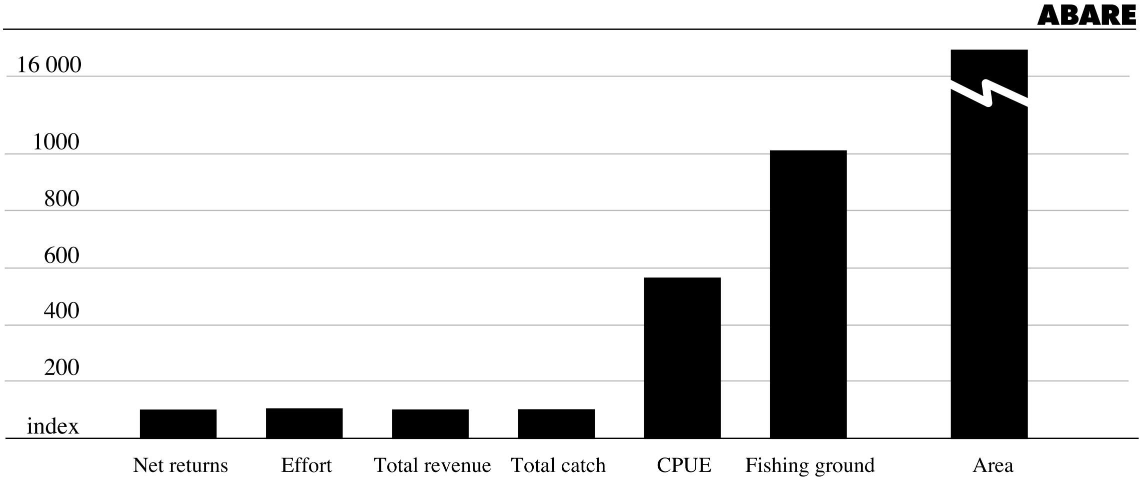
e) Average annual total catch

f) A binary variable representing whether or not the cell was fished

g) The area of the planning cell.

1 The simulated annealing optimisation was coded in C++ and run on a 233MHz PII PC. The results presented in the graphs are each the average of seven runs of the algorithm. Each run took approximately 2.5 minutes.

Figure 2: **Opportunity cost using alternative cost measures in designing the reserve system** a



a Alternative cost measures were used to design 'least cost' reserves. Opportunity costs were then calculated using the net returns to the fishery measure.

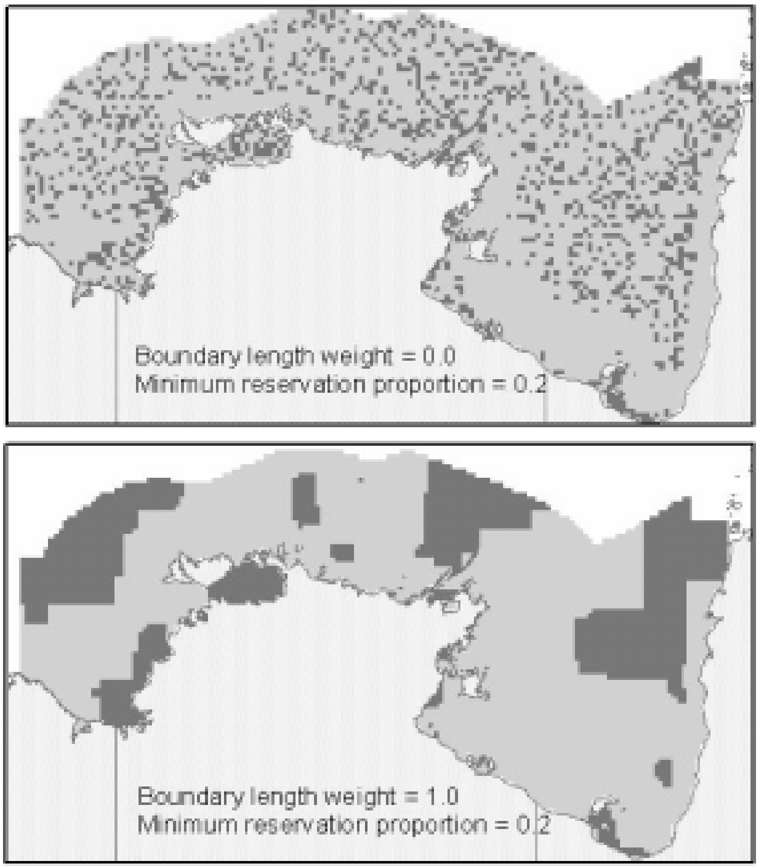
Figure 2 illustrates the opportunity cost (calculated using measure (a)) of the optimal reserve system found when each of the measures is used in the objective function. In this example the high correlation of measures (b), (d) and (e) with measure (a) explains their good performance in the objective function. Measures (c), (f) and (g) predictably generate increasingly inferior solutions. Measure (g) corresponds to the case of ignoring cost in the reserve design. Ignoring costs is fairly standard in conservation planning and makes the revision and negotiation process much more drawn out.

As there is no allowance for annual resource rents in a cell to change following a change in access regulations elsewhere in the fishery the costs calculated under simulation (a) will not generally be the true costs. The true costs would account for the induced changes in the spatial abundance of prawns (and the resulting changes in the spatial distribution of fishing effort) throughout the fishery and any other management changes introduced to complement the marine reserve system. Unfortunately it was not possible to use such a measure in this paper.

Figure 3 illustrates the effect of the boundary length parameter. The cells closed to fishing are shown in the darker shade. As expected, ignoring boundary length (setting the boundary length parameter to zero) generates a very dispersed and probably unmanageable reserve system. A parameter value of one generates the very compact reserve system illustrated.

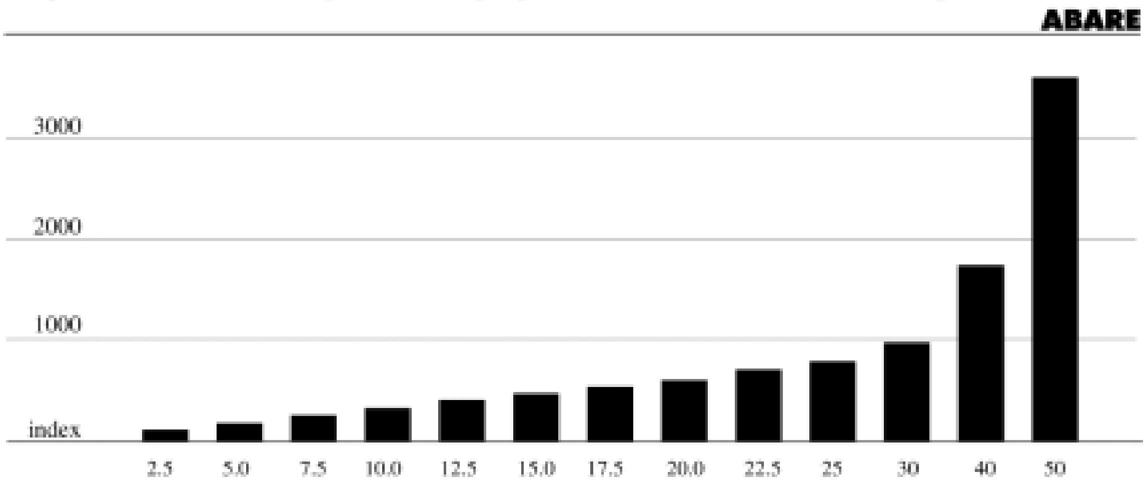
In determining an appropriate balance between environmental and more narrowly specified economic objectives it is important to be able to estimate the tradeoff involved. Figure 4 illustrates the effect of increasing the reservation target proportion. These were calculated using a boundary cost of one. For this set of values and distribution of environmental components the marginal cost of reservation remains reasonably low and constant until a fairly high reservation target of around 30 per cent is reached. From then on marginal cost

Figure 3: **Effect of the boundary length term in the objective function**



continues to increase at an increasing rate until the fishery is closed down completely. In providing information to stakeholders or decision makers, more detailed impact analyses — than the simple cost measure and reservation proportion — may be carried out on the reserve systems generated.

Figure 4: **Cost of reserving a common proportion of each environmental component**



Directions for further research

The illustrative example outlined in this paper demonstrates a methodology for, and the potential benefits of, integrating economic information into the reserve design process. This application did not involve modeling the biology of the prawn stocks. In order to calculate a true measure of the cost of a marine reserve system it would be necessary to account for the induced changes in the spatial abundance of prawns (and the resulting changes in the spatial distribution of fishing effort) throughout the fishery.

In most instances biological models that have the capacity to predict changes in the distribution of target species throughout a specified area will not be available. This information will need to be generated or the marine reserve planning process will have to proceed without it. The manner in which improved costs of reservation can be obtained by area and the implications of proceeding without knowledge of how the spatial distribution of fish stocks and fishing effort may change is the subject of current ABARE research.

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