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Abstract

This paper gives an overview of bioeconomic modelling of marine reserves, and illustrates how economists have responded to the modelling results found in the ecological literature. The economic analysis is shown to be far more pessimistic with regard to the potential of marine reserves as a fisheries management tool, than what one finds in the purely ecological analysis, the reason being the latter's neglect of issues such as discounting and economic incentive behaviour. However, the economic analysis, despite some of it being relatively advanced with regard to spatiality, is still simplistic with regard to for instance ecosystem and habitat content. A simple expansion of the existing bioeconomic models with regard to positive habitat effects of area closures is presented and analysed, showing room for improved results from marine reserve implementation as compared to existing analysis.

Key words: marine reserves, bioeconomic modelling, habitat

Introduction

Marine reserves¹ have become one of the hot topics in marine management with proponents in most camps. Biological research on the topic has exploded since it re-emerged in the late 1980s (Conover *et al.* 2000), after Beverton and Holt (1957) laid closed areas on the back burner, and instead directed the focus upon gear restrictions and fishing intensity. Economic research on marine reserves is, however, comparatively new and of a much more limited quantity, though increasing rapidly. In this note the economic critique of the biological approach to marine reserves is presented, while at the same time also adding to the standard bioeconomic model in order to open for broader ecosystem considerations as demanded by ecologists.

Fisheries economists have traditionally concentrated on issues concerning management of commercially important species, and have tended to focus on one or perhaps two-species models (for some exceptions to this see Flaaten 1988, Eide and Flaaten 1998), leaving the broader issue of biodiversity to more general ecological-economic research. In recent years, however, there has been a growing worry that marine resources are being rapidly decimated (Botsford *et al.* 1997, Myers and Worm 2003, Jackson, *et al.* 2003). Resulting concern for the biodiversity in the oceans has awakened policy-makers as well as researchers in many fields, and the fact that *fisheries management* now must move from a single-species focus to a biodiversity focus has been underlined (Hanna 1999). However, *fisheries economics research* does not seem to have taken this step yet. One topic that has emerged where biodiversity and commercial fisheries *could* meet is in management options such as marine reserves. Fisheries economists started publishing work on marine reserves towards the end of the 90s, showing an increasing interest for the topic, and often a slightly different approach and attitude to that of the ecologists. Economists have usually been more critical to marine reserves as a fisheries management option than the ecologists (see Hannesson 1998, Smith and Wilen, 2003). However, the economic analysis is still to a large degree done by applying single-species systems (see however Bonceur *et al.* 2002 and Reithe, 2006), with issues of biodiversity or habitat seldom being included (see however Rodwell *et al.* 2003, Schnier 2005, and Upton

¹ In the literature a large number of different expressions are used to describe closures of areas in the oceans. In this work marine reserves and marine protected areas (MPAs) are used interchangeably to describe permanently closed areas to some or all human activity.

and Sutinen, 2003). This work attempts to add to the existing bioeconomic analysis by taking into account habitat effects of marine reserve implementation. A simple habitat improvement feature is presented in an accepted bioeconomic marine reserve model, showing improved results from marine reserve implementation as compared to existing models. This accentuates the need for further integration of ecology and economics in the study of ocean management.

The advantage of reserves from a biodiversity point of view is clear – an area undisturbed will presumably over time regain some form of natural equilibrium and secure biodiversity at least within the reserve. Positive biodiversity effects for the area outside the reserve could be through migration or density dependent dispersal, or the fact that the reserve may function as a buffer to shocks to the system (see Conrad 1999 and Sumaila 1998, and also Hannesson's 2002 more critical comment). The advantages of reserves for commercial fisheries are usually presented as the increased fish production migrating out of reserves. The norm has been to focus on some form of density dependent dispersal in the economic literature (Hannesson, 1998, Sanchirico and Wilen, 1999, 2001) but also to some degree in the biological literature (Hastings and Botsford, 1999). But, as Gell and Roberts (2003) point out, very little is known regarding density dependent dispersal. The limited amount of empirical work done does however not preclude the possibility that density dependent dispersal exists, and density dependent influences are still clearly important (MacCall, 1990), but presumably more relevant as regards spawning, recruitment and cannibalism. Though some work has been done on differing habitats and marine reserves (Schnier 2005)², habitat effects of reserves have received scant attention (see however, Rodwell *et al.* 2003 and Upton and Sutinen 2003).³ It is clear that commercial fisheries do impact upon marine habitats (see Auster and Langton 1999 for an overview), and this needs to be accounted for in economic analysis. In this paper a standard bioeconomic marine reserve model is expanded upon by allowing habitat enhancing effects, depending on the size of the reserve (see Halpern 2003 for a discussion of reserve size and biological effects). These effects come in the shape of changes in carrying capacity (as suggested in Roberts and Sargant, 2002), and could

² Much of the bioeconomic literature on marine reserves allows for differing growth and carrying capacity, but little attention is given explicitly to this issue prior to Schnier (2005).

well be deleterious (as described by Skonhofs and Armstrong, 2005 and Armstrong and Skonhofs, 2005). The intuition behind positive effects is that closing areas to fishing allows improved habitat for protection of young and spawning, as well as increased prey abundance (Auster and Malatesta, 1995, Garcia-Charton and Perez-Ruzafa, 1999, Roberts and Sargent, 2002).

This paper first presents the ecological-economic research on marine reserves, and shows the lack of ecosystems or habitats in these models. A bioeconomic habitat enhancing model of a marine reserve is presented in the second section, followed by the results and comparison between different management options. In conclusion the paper discusses the results and reflects more generally over the integration of ecological and economic knowledge in models of marine reserves.

Bioeconomic modelling and marine reserves

Ecological research on marine reserves is generally more positive to this management option than the economic research is, with Bohnsack (1993) summing up much of the former groups' optimism, stating that marine reserves will protect resources, enhance fisheries and even solve conflicts. Economists question all these issues as well as others in different studies which will be reviewed below.

Discounting the future

One of the earliest problems that economists have pointed to as regards marine reserves is the issue of *time*. Also biologists have underlined the trade-off between short term profits from fisheries versus possible long term gains from risk reduction of marine reserves (Dayton *et al.* 2000). This is however explicitly studied in Holland and Brazee's (1996) dynamic bioeconomic analysis where the discount rate of those affected by a reserve implementation, and their minimum production requirements, are critical with regards to policy decisions on marine reserves. If society discounts the future to a large degree, possible gains from marine reserves a long time ahead will not pay for the loss in economic activity today. The consequences for policy-making are well known from the global warming debate surrounding the Kyoto

³ Studies of terrestrial reserves and increased carrying capacity have however been made (see Carroll *et al.* 2003, Song and Li 1995 and Warkentin *et al.* 1995)

Protocol. Hence the “bounce-back” propensity of chosen marine reserve areas compared to society’s rate of discount, will determine whether marine reserve policy as a fisheries management option will successfully win through.

Uncertainty

Natures’ ability to “bounce back” naturally brings us to the issue of uncertainty, which is also of interest from an economic viewpoint. It has been hypothesised that marine reserves could insure against environmental shocks, or function as a “hedge against inevitable management limitations” (Lauck *et al.* 1998). Sumaila (2002) applied a Beverton-Holt bioeconomic simulation model and found that the implementation of a reserve may indeed protect discounted economic rent. This model assumes, however, that the shock only occurs in the fishable area. Conrad (1999), using a Gordon-Schaefer model assumes a more general shock to the system, and finds that marine reserves do succeed in reducing biomass variation, but also reduce harvests and thereby economic rents as compared to a private property management without a reserve.

Management options outside the reserve

Since Gordon’s (1954) seminal work, economists have criticised open access fisheries, as the effect of profits to attract fishers in unmanaged fisheries leads to excessive effort investment, which decimates the profit, even in the presence of marine reserves. In the analysis of marine reserves, many economists have pointed to this fact (Holland and Brazee, 1996), and also shown that perfectly applied private property management without reserves will better take care of both stocks and harvests, unless marine reserves are exceedingly large (Hannesson, 1998). It is clearly not fair to assume that in fisheries with a high degree of sophisticated management, the implementation of marine reserves will eradicate all other management except perhaps access (see references to limited-entry management and marine reserves below). Much work still remains, therefore, with regards to the analysis of different management options than solely open and limited access outside marine reserves.

Reduced management costs

It has been claimed that marine reserves in industrialised fisheries may be cheaply monitored via satellite systems, thereby reducing management costs. Bioeconomic

analysis including management costs (Armstrong and Reithe 2001) has indeed shown that the introduction of marine reserves has the potential to reduce the reserve size that will give equal stock and harvest results as that of private property management without a reserve. However, existing reserves are most common in tropical waters with greater prevalence of subsistence fisheries, where unless there is community acceptance, the monitoring costs are prohibitive. There are however very few existing reserves in fisheries where one could expect reduced management costs, such as for instance in highly industrialised fisheries with for instance satellite monitoring. We do however see increasing pressure for the implementation of marine reserves in areas with high degree of industrial fishing activity, where remote sensing and closed areas could potentially reduce management costs.

Reduced conflict

It has been claimed that marine reserve implementation will somehow reduce conflicts at sea (Bohnsack 1993), presumably by zoning, and separating different interest groups in the ocean. The many conflicting opinions that have arisen *prior* to reserve implementation in for instance California may not be an indication of the climate *after* a reserve implementation, but it definitely begs the question of how a shrunken area of use will reduce conflict among a given group of agents. Sumaila and Armstrong (2006), using a one-species cohort model with two harvesting groups fishing on different age groups within the stock in question, show how there may be clear disagreement between the two groups as regards marine reserve size and fisheries management implementation. This result is underlined in Holland's (2000) applied model of multi-species and multi-agent fisheries on Georges Bank. Via simulations it is shown that some groups of fishers obtain increased yields from permanent marine reserves, while others obtain reduced yields.

"Double Payoff"

So far we have mainly discussed work that focuses on simple two-patch models. Sanchirico and Wilen (1999, 2001) introduce greater degree of spatiality into bioeconomic models of marine reserves. They show that economic results are highly dependent upon the type of interaction between different patches, and which patch is closed. This due to complex spatial and intertemporal effort redistribution effects. Hence both economic and biological factors affect the results of reserve

implementation. The authors (*op. cit.*) identify circumstances which would afford what they call a “double payoff”, namely increased aggregate biomass *and* aggregate harvest by closing one or more areas to exploitation. They find that, given open access outside reserve patches, biological benefits may be forthcoming from reserve implementation, but few combinations of biological and economic parameters give *both* increased aggregate harvests and aggregate biomass.

Human behaviour

A central critique made by economists of biologists’ work is how the latter include human behaviour. In most biological models of marine reserves, effort is assumed exogenously given and constant (Man *et al.* 1995, Carr and Reed 1993). Economists see this as an overly strong assumption and underline the incentive effects that fishers react to. Smith and Wilen (2003), use an applied, spatially and behaviourally explicit, dynamic bioeconomic model of a specific fishery, the sea urchin fishery in northern California, which they compare to a pure biological model with standard simplifying assumptions regarding human behaviour, i.e. in effect ignoring economic incentive behaviour. They demonstrate that the optimistic conclusions regarding reserves found in ecological work may well be a result of the simplified assumptions ignoring economic behaviour. Furthermore, the preferred areas to close vary greatly for the two models analysed.

Which areas should be closed?

What does the economic research say about which fisheries and areas to close in the case of reserve implementation? Holland and Brazee (1996) show that only fisheries with a high degree of effort already present will gain from reserves. This is also a result found in Holland’s (2000) applied work, as well as biological modelling (see the review in Gerber *et al.* 2003). Hence fisheries that manage to control effort by other means are best left alone. Sanchirico and Wilen (2001) find that given open access conditions, closing the most profitable high productivity patch provides the greatest chance of increased aggregate harvests. This is due to the fact that under open access, the most profitable area is overexploited, and hence fits into Holland and Brazee’s (1996) definition of high effort levels. Introducing limited-entry management changes these results. In this case Sanchirico and Wilen (2002) show that the closure of lower productivity areas gives higher values in the fishery, as the

high productivity areas under limited-entry already without reserves give large returns. The opportunity cost of closing such areas is thus high. Hence closing low productivity and high cost areas may lead to increased profitability in a limited-entry fishery. This underlines the importance of the status quo in a fishery prior to reserve implementation, and also points a finger at one of the most usual determinants for reserve area choice, namely natural productivity. Once the economics of a managed fishery is taken into account, areas less attractive are more advantageously closed. This is often in direct conflict with what we see in actual reserve management as well as suggestions from biologists (Garcia-Charton and Perez-Ruzafa 1999).

Bycatch fisheries

Bycatch fisheries are by many economists seen as the type of fishery that will best gain from marine reserve implementation. This is however a topic that has not been sufficiently studied⁴, though Reithe (2006) shows numerically in a two-species model that the type of ecological interaction between the two species influence the possibility of actually protecting a bycatch species through the use of a reserve. The ecological interaction also affects the possibility of obtaining a win-win situation when implementing a reserve, and also determines the optimal patch to close. Ecosystem modelling is not prevalent in bioeconomic research as of yet, but marine reserves as a fisheries management tool seems like a natural place for such research to emerge. Bonceour *et al.* (2002) apply a two-species, two-area model of marine reserve implementation. One stock is a commercially interesting prey species while the other stock is a non-commercial predator. They show that interactions between species may reduce benefits to the fishing industry of reserve implementation, but on the positive side create opportunities for eco-tourism development.

Applied bioeconomic modelling work on marine reserves is limited, but one of the few examples that does exist, Holland's (2000) multi-species, multi-fishery bioeconomic model of Georges Bank brings out a critique of the perhaps most often supposed positive result of marine reserve implementation, namely increased stock sizes (see however Parrish (1999) for a similar argument from a biologist). In this work he shows that in a situation where fisher's react to area closures in an

⁴ Also on the biological side, models with more than one species are scarce (Gerber *et al.* 2003).

economically sound fashion, resulting fishing may actually increase the danger of depletion for some fish stocks. This is clearly a warning regarding how we design marine reserves.

So far, however, almost all modelling of marine reserves in economics rests upon the driving force of density dependent migration. Density dependent migration is highly probable, but nonetheless not demonstrated (Gell and Roberts 2003). This begs the question of possible alternative advantages of marine reserves, such as how possible habitat improvement as a result of closed areas will affect the harvests and profits. Issues such as the “shifting baseline syndrome” (Pauly 1995), and the question of whether applied carrying capacities are large enough, or should be revised for marine reserves is clearly an issue that would affect some model results. Here is a clear ecological input which may affect the bioeconomic results with regards to marine reserves. This issue is discussed in the following.

Modeling habitat effects of marine reserves

In economic analysis of marine reserves so far, studies have had a single-species focus (see however Bonceur *et al.* 2002 and Reithe, 2006), and issues of biodiversity have not been given economic weight. A model that takes inherent values of ecosystems into account could be one way to remedy this, for instance by giving species biomass or density economic weight and value (Skonhøft and Johannesen, 2000, used a similar model in a study of reindeer herding). The object would be to maximize the total value of habitat use:

$$\text{Max } \Pi(h,S) = \omega\Pi(h,S) + (1-\omega)\Pi(S)$$

where $\Pi(h,S)$ are profits from harvesting h , which also depend on stock size S , and $\Pi(S)$ are profits and valuations tied to the biodiversity or density S of production in the relevant area. $\Pi(S)$ could be profits connected to tourism, which would be dependent upon the biodiversity or the density of natural production in the area of interest. Alternatively, or additionally, $\Pi(S)$ could incorporate non-use values (Skonhøft, 1999). ω is a weighting parameter giving the relative weight attributed to the two forms of value emanating from the habitat in question. The issues of determining ω could be solved in a way similar to Munro’s (1979) method of finding

the weighting parameter between the profits of two countries harvesting a common fish stock, i.e. applying for instance the Nash bargaining solution.

A few bioeconomic papers have discussed habitat quality and reserves. Schnier (2005) models heterogeneity of habitats, but does not include how the implementation of reserves can affect this quality. Rodwell *et al.* (2003) study positive and negative fisheries effects as a function of time with a reserve in place; the positive effects emanating from reduced natural mortality and the negative being reduction in spatial movement out of the reserve, both due to improved habitat within the reserve. Upton and Sutinen (2003) model reduction in the habitat growth dependent upon fishing, which feeds into intrinsic growth and carrying capacity of the fish species in question. A different model that takes into account habitat effects of marine reserves could be analysed by modifying the carrying capacity K of the logistic growth function⁵.

In the following we will study a simple model of carrying capacity being affected by reserve size⁶, in keeping with existing bioeconomic models on marine reserves, hence allowing comparison.

The model is based on Hannesson's (1998) marine reserve model, though in this paper stock entities are studied, rather than densities as Hannesson (*op. cit*)⁷. To this basic model is added a possible habitat effect of marine reserve implementation. One type of habitat effect is opened for, namely that carrying capacity in the reserve depends on marine reserve size. The following four management regimes are studied;

- Open access
- Private property
- Marine reserve
- Marine reserve with habitat effects

⁵ Alternatively the intrinsic growth rate of the logistic function could be modelled as being habitat dependent, as this will have a different effect upon long run stock size and optimal harvesting effort as compared to that of the carrying capacity.

⁶ Armstrong *et al.* (2004) analyse a hypothetical marine reserve for the North-East Atlantic cod stock applying a more complex version of the carrying capacity habitat effect presented below. They show that given modest habitat effects, a marine reserve for a migratory species such as cod may well be economically advantageous.

We will start by defining the model for the marine reserve, assuming that it is always combined with open access outside the reserve, i.e. effort enters where allowed as long as there exists positive profits in the fishery. The private property regime ensures profit or rent maximization, without a reserve. On the other end of the scale is a pure open access regime. Hence these two standard management options are compared with marine reserve management with and without habitat effects. Habitat effects as the result of the implementation of a marine protected area will be described in the following.

The resource in question is uniformly distributed over a homogeneous area. K is the carrying capacity in the total area. A fraction m , $0 < m < 1$, of the total area determines the reserve size, making $(1-m)$ the fraction of the total area found outside the reserve. Hence the basic carrying capacity within the reserve is mK , and the carrying capacity outside the reserve is $(1-m)K$. The rate with which fish move between the two areas, or in this context the dispersal parameter, is defined as z . It is assumed that for a positive m , i.e. reserve, there is an addition to the basic carrying capacity within the reserve, as total carrying capacity in the reserve is affected by the *carrying capacity habitat effect* $g(m)$ ⁸. We assume that $g' > 0$, and $g'' \leq 0$. Thus, total carrying capacity increases for increased reserve size, but at a constant or decreasing rate⁹.

This makes the total carrying capacity in the reserve equal to;

$$mK + g(m)$$

⁷ Anderson (2002) and Conrad (1999) use models similar to the basic model applied here.

⁸ We assume that only the carrying capacity in the reserve is affected by the size of the area closure. Hence there is an underlying assumption that the outside area's habitat is either in such a condition or of such a character that any change in effort that may occur from reserve implementation does not affect it to any greater degree. This seems like an acceptable assumption in cases where there is extensive prior fishing activity, or where the habitat outside the reserve is of a kind that is not affected substantially by fishing effort.

⁹ Carrying capacity could also be dependent upon the reserve shape, in the sense that large dispersal z may describe a reserve shape that gives a large reserve boarder line (as described by Flaaten and Mjølhus, 2005), while low z defines a more compact reserve. This may clearly be an issue with regard to protection of genetic base, in the sense that a reserve shape that allows for large dispersal out of a reserve, irregardless of size, will not to the same degree protect attributes of individual fish that may make them desirable from a commercial point of view, or for sustaining large stock levels (Trexler and Travis 2000).

The aggregate resource biomass, if we have a biodiversity perspective, or single stock size in the simpler context, is described by X_i , with $i=M,O$, denoting the marine reserve and the outside area respectively. The rate of change in biomass in the fishable area becomes:

$$\frac{dX_o}{dt} = rX_o \left(1 - \frac{X_o}{(1-m)K_o}\right) + z \left(\frac{X_M}{mK + g(m)} - \frac{X_o}{(1-m)K_o}\right) - Y, \quad (1)$$

While the rate of change in biomass in the reserve area is:

$$\frac{dX_M}{dt} = rX_M \left(1 - \frac{X_M}{(1-m)K + g(m)}\right) + z \left(\frac{X_o}{(1-m)K} - \frac{X_M}{mK + g(m)}\right) \quad (2)$$

The first term on the right hand side of (1) describes the growth in the fishable area, while the second term expresses the net migration to the fishable area, and Y is the harvest. The net migration expressions are determined by the relative densities of fish in the two areas, hence a density dependent migration between reserve and outside area. The growth in the reserve area is described in (2), and is equivalent to (1), except that in the reserve there is no harvest.

Assuming p is the unit price and c/X_o is the unit cost of harvest, the profit function becomes:

$$\Phi(X_o, X_M) = \left[rX_o \left(1 - \frac{X_o}{(1-m)K}\right) + z \left(\frac{X_M}{mK + g(m)} - \frac{X_o}{(1-m)K}\right) \right] \left(p - \frac{c}{X_o}\right) \quad (3)$$

Since we assume there is open access outside the reserve, (3) is set equal to zero, giving the standard open access stock size outside the reserve

In the following we will assume that $g(m) = vm$, with v being a positive constant, i.e. the habitat effects of reserve size are directly proportional to the reserve share m of the total area. For the marine reserve case without habitat effects, v is set equal to zero.

The open access and private property regimes can be modeled as follows. The biomass is here defined as X , as we only have one area, and the other parameters are

as for the marine reserve case. This gives us the objective function for the private property case:

$$\Psi(X) = rX \left(1 - \frac{X}{K}\right) \left(p - \frac{c}{X}\right) \quad (4)$$

which is maximized subject to

$$\frac{dX}{dt} = rX \left(1 - \frac{X}{K}\right) - Y$$

giving standard private property results for the whole area.

For the open access case, we set (3) equals to zero, giving standard open access results for the whole area.

Data

Carrying capacity is normalized; i.e. $K = 1$, as is unit price, $p=1$. Intrinsic growth rate r is set equal to 0.6, and cost c is equal to 0.05.

Results

The results show the open access and private property regimes to be lower and upper bounds, respectively, for yield in the marine reserve case. But, once habitat effects in reserves are allowed, the private property case no longer functions as an upper bound for yield at all closure levels.

As we see in Figure 1, the habitat effect is positive as one would expect, given that v is positive. When compared to the marine reserve case without habitat effects, as the size of the marine reserve increases, the increasing habitat effect increases the stock. Figure 1 and Figure 2 are for a low habitat effect of $v=0.2$. This value is chosen as it gives the minimum value of v for which the marine reserve can match the optimal management regime, when $z=0.3$. For $v=0.2$ the reserve must have a size of approximately 75% of the total area in order for the harvest to be as large as under the private property regime. This means that the increase in the carrying capacity in this

case is 15%. By increasing ν , the minimum reserve size that matches the harvest in the optimal management case is reduced, as shown in Figure 3. In this figure $\nu=0.4$ and the minimum reserve size that gives harvest matching the case of optimal management is approximately 60% of the total area, implying an increase in the carrying capacity of 24%.

Applying $\nu=0.4$ and a 60% reserve, we observe in Figure 4, as expected, that the stock situation is improved with carrying capacity improvement due to reserve size. This improvement wears off, however, as migration z increases. Looking at the equivalent for catch in Figure 5, we see that harvest is not affected as drastically as the stock by increased migration, as migration allows the harvest quantity to remain high as the stock declines.

Conclusion

Comparing to conventional models with purely density dependent gains from marine reserves, it is shown that relatively limited carrying capacity improvements as a result of reserve size give increases in stock and harvest. Private property results are emulated for reserve sizes of approximately 75% of the total area when carrying capacity is increased by 15%. For increasing habitat effects, the smaller is the needed reserve to match the private property harvest and stock levels.

Though it is unsure whether large carrying capacity increases as described above actually appear in marine environments, terrestrial reserves have been shown to have carrying capacities more than 20% greater per ha. than outside their borders (Song and Li 1995). Higher densities of fish within reserves than outside are found in many cases (see Attwood *et al.* 1997 and Halpern 2003 for overviews). It is however unclear whether this is due to reserve implementation or the fact that reserves are often established in more attractive habitats (Garcia-Charton and Perez-Ruzafa 1999), as data has to a large degree not been available. Furthermore, when closing heavily harvested areas, the consequences for the areas outside may be detrimental, hence unless managed, reducing the carrying capacities outside the reserve. And even when increases in density within reserves are shown to follow reserve implementation, this does not necessarily imply that carrying capacity has increased, as this would depend

on how large the population was originally relative to carrying capacity. Nonetheless ecological research increasingly stresses ecosystem effects on single species, and underlines the fact that these issues should be taken into account. The results given here support this, as positive habitat effects as a result of marine reserve implementation are shown to have a potential to match private property harvest. This underlines the need to take into account broader ecological effects in the ecological-economic modelling of marine reserves.

Summarising the economic research so far, it is safe to say that economists in general underline the dangers of assuming marine reserves to be a panacea for fisheries management. Marine reserves must be seen as one of the many tools of fisheries management, and the how, where and for what fisheries, reserves are implemented is of great concern. Economists underline the political problems of reserve implementation, due to diverging economic interests in the relevant areas, and illustrate the sometimes conflicting economic and biological goals in the use of marine reserves in fisheries management. However, the economic analysis of marine reserves is still in its infancy, demanding much further study in a wide range of topics, where economic analysis may illustrate the shortcomings of the purely ecological modelling, and ecological modelling can illustrate where more encompassing ecological models, compared to those applied by economists, may affect the results with regard to marine reserve implementation.

Looking at fisheries management, we can see an evolutionary path leading from input controls such as gear and time restrictions to output controls such as TAC and more disaggregated quota limitations. Economists have for years (starting with Gordon, 1954) pointed to the fact that management focus purely on the biological and technological side of fisheries will result in inefficient fisheries, where the potential economic rent is eaten up by increased fishing capacity. Economists have traditionally espoused economic incentive systems for management, such as output taxation, or more commonly individual transferable quotas (ITQs). However, experience shows that such management options have not eliminated stock decline (see Iceland's cod stock decline a few years back, and the recent experiences with New Zealand's Hoki as well as other species). Indeed this was not the aim of ITQs as such, as efficiency was the main focus, and in this ITQs may well have succeeded (Hannesson, 2004).

The next evolutionary step, however, seems to be a return to input control in managing the stock, but this time in a more perfect form; permanent area closures, or marine reserves. There is however a danger of marine reserves faring as poorly as ITQs. Not because either management option necessarily is detrimental, but that knowledge, control and degree of implementation is imperfect, hence leading to poor results which colour perceptions of the management option negatively (Dayton *et al.* 2000). Hence there is a great need for increased knowledge within a broad range of academic disciplines in order to better understand the many consequences of marine reserves.

Here it seems clear that research that is truly multidisciplinary, i.e. also with regards to the researchers collaborating would be advantageous. Economists working in the field of bioeconomics accept criticism from ecologists for their use of simplistic ecological models, such as the one applied here for expositional purposes. The economist defence is however that with the additional complexity of economics on top of for instance cohort models, the creation of a “big black box” is imminent, rendering constructive analysis minimal¹⁰. Hence most bioeconomic models of marine reserves are so-called Gordon-Schaefer biomass models (with some exceptions such as Holland and Brazee 1996 and Sumaila 2002) and also most often one-species models (here Boncoeur *et al.* 2002 and Reithe 2006 are the only exceptions as of yet), and are usually designed to analyse specific general issues. In fisheries, multi-species management issues brought to a fore that economics was needed in conjunction with biology, in order to somehow weight the different interacting species. The complexity of the biology involved, increased by the addition of economics may be why there is so little multi-species bioeconomic analysis carried out in fisheries. It is however clear, in the same way that multi-species approaches require greater interaction between the disciplines, ecosystem approaches have the same requirement. When bringing several, sometimes competing species, as well as habitats, into the analysis, some form of valuation is required for determining optimal human utilisation, hence economics is required. However, a limited presentation of the ecology may mask the full effects of human interactions with the ecosystem, hence requiring greater

¹⁰ This defence does however only seem to be valid for economists, as ecologists have to a greater degree accepted that model construction and analysis cannot be done using simple Gordon-Schaefer type models.

ecological involvement. It remains to be seen whether the multi-disciplinary research which one would have expected to emanate from multi-species management issues coming to a fore, will now appear with the increasing focus upon ecosystem management.

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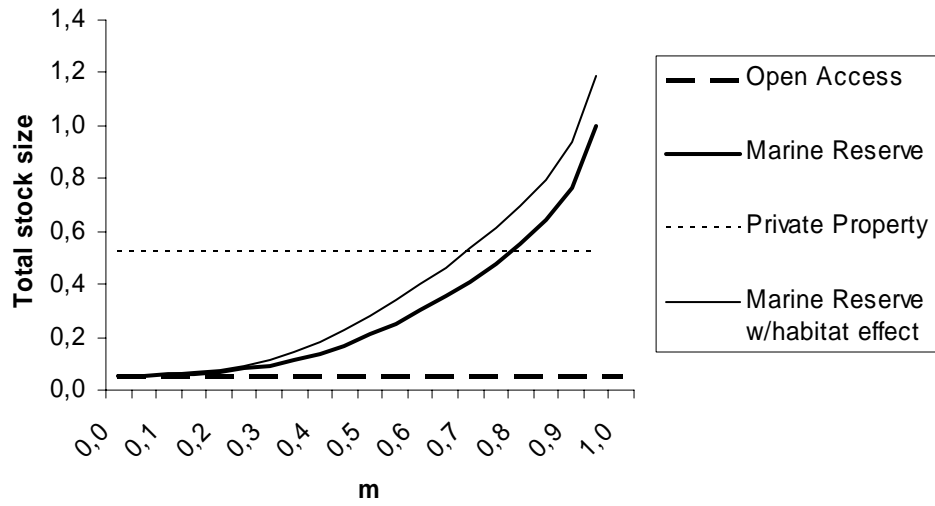


Figure 1. Effects on stock of varying reserve size m with carrying capacity positively affected by m ($z = 0.3, \nu=0.2$)

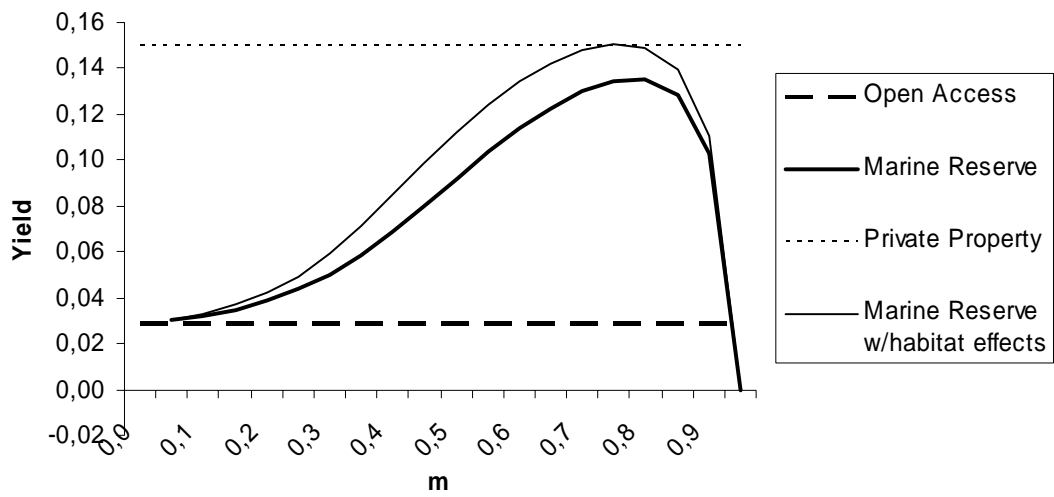


Figure 2. Effects on catch of varying reserve size m , with carrying capacity positively affected by m ($z = 0.3, \nu=0.2$)

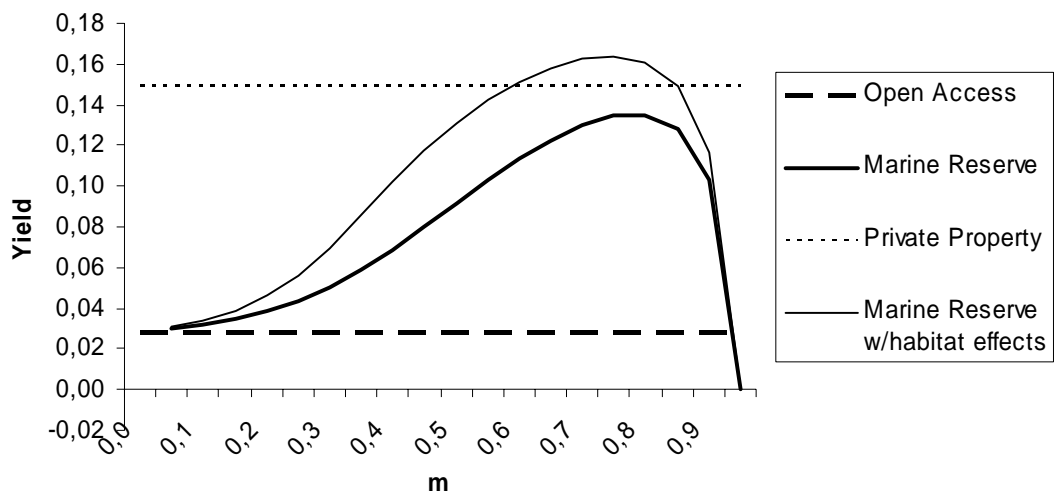


Figure 3. Effects on catch of varying reserve size m , with carrying capacity positively affected by m ($z = 0.3, \nu = 0.4$)

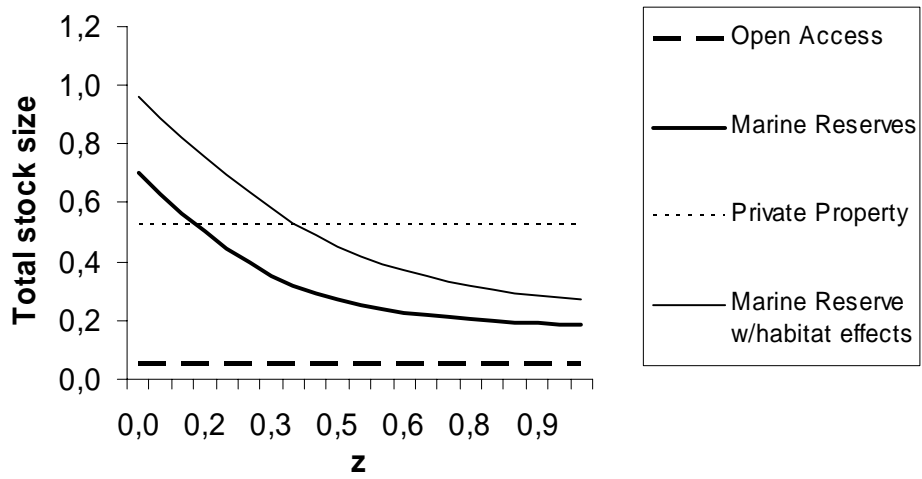


Figure 4. Effects on stock of varying migration rate z , with carrying capacity positively affected by m ($m = 0.6, v=0.4$)

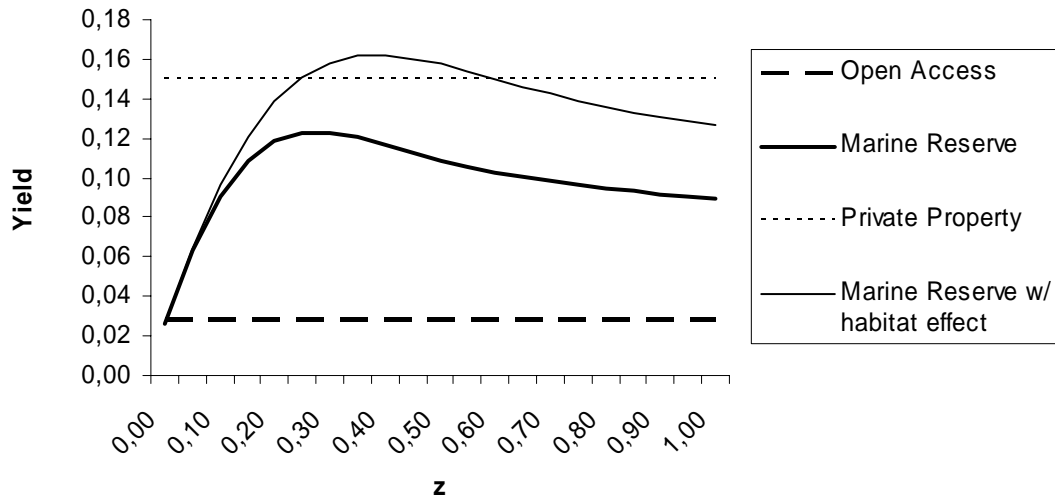


Figure 5. Effects on catch of varying migration rate z with carrying capacity positively affected by m ($m = 0.65$, $v=0.4$)