

**A framework for evaluating spatial closures as a
fisheries management tool**

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Spatial closures are a potentially useful fisheries management tool. There are a variety of attributes that any one closure may have (size, shape, location, permanence, gear specificity) and alternative arrangements of individual closures to form a closure regime. For a given fishery, different closure regimes are likely to have different performances with respect to management objectives and these alternative regimes should be evaluated before they are implemented.

As with other fisheries management controls, experimental evaluation of spatial closures is difficult due to the large spatial and temporal scale over which they act. However, simulation modelling provides a means of evaluating alternative closure regimes at the appropriate spatial scale by integrating the available information on the population and spatial dynamics of the fishery. Previous approaches to modelling the effects of spatial closures on fisheries have generally ignored five key aspects: (i) the effects of other management controls, (ii) spatial heterogeneity in the environment, (iii) the spatial dynamics of fishers, (iv) uncertainty in model parameters, and (v) performance indicators other than total yield. This report describes a framework for evaluating spatial closures as a fisheries management tool that addresses these five aspects.

The framework developed is based upon applying spatial extensions to existing stock assessment models (Section 2). The resulting integrated model is, therefore, consistent with the current stock assessment and thus able to evaluate spatial closures in unison with existing management measures. The framework reviews models for describing the spatial dynamics of fish and fishers in terms of their assumptions, approaches, and parameterisation (Sections 3 & 4). The integration of the spatial extensions with the existing assessment model is discussed in Section 5, including the issues of spatial resolution, spatial dimensioning of population dynamics parameters, and the estimation of parameters. Finally, steps for using the integrated model to evaluate closures are described (Section 6). The steps are (i) defining management objectives and performance indicators, (ii) choosing closure regimes for assessment, and (iii) incorporating uncertainty into evaluations.

The framework was developed to be broad enough to evaluate spatial closures for most fish stocks in New Zealand. However, the degree of certainty in evaluations of spatial closures will be dependent on the amount of information available on the spatial dynamics of both fish and fishers.

1. INTRODUCTION

1.1 Closures as a fisheries management tool

Recent failures of conventional management controls in many fisheries have generated a renewed interest in the application of closures as a fisheries management tool (Bohnsack 1996, Clark 1996, Roberts 1997, Dugan & Davis 1993). Closures are principally recognised in a fisheries management context for their potential to provide a buffer against uncertainties in stock assessments, variability in recruitment, and catastrophic events by providing a refuge in space (Beverton & Holt 1957, Davis & DODRILL 1980, Roberts & Polunin 1991, Guénette et al. 1998). There are two main mechanisms by which closures may provide fisheries management benefits: (a) movement of post-settlement fish from closed areas to fished areas – the 'spillover effect', and (b) protection of a component of the spawning stock from fishing, thus increasing egg production (Rowley 1992).

Closures can potentially increase yield per recruit through the spill-over effect by providing a growth refuge from fishing in space, in much the same way as size limits do through size. The degree to which an increase in yield is realised will depend on the ratio of the open:closed areas, the level of fishing mortality, and the movement behaviour of the target species. Some simulation studies suggest the potential benefits from the spill-over effect are likely to be small and localised (Polacheck 1990, DeMartini 1993).

Marine reserve theory predicts an increase in the abundance of larger more fecund individuals within closures compared to fished areas (Bohnsack 1990, Roberts & Polunin 1991, Carr & Reed 1993), thus potentially increasing the spawning output of the protected population. This increase in spawning output has the potential to maintain, or even increase, yields relative to a non-closure scenario, through increased production and dispersal of planktonic larvae from the closure to fished areas. Given the extended duration of the larval phase in most marine species, the spatial extent of this effect is likely to be considerably larger than the localised spill-over effect (Rowley 1992).

Through these mechanisms, spatial closures would be expected to benefit the management of fisheries where there are no other controls on exploitation rates. However, in fisheries where controls already exist, the impact of closures may be less pronounced. Indeed, spatial closures could counteract the effects of other controls. For instance, if a fishery is managed by a limit on catch and closures are introduced without reducing catch limits, then depending upon the rates of movement of fish, the areas open to fishing could be overfished and the overall biomass of the stock decline (Bohnsack 2000).

The effects of closures will depend upon the spatial and population dynamics of the fish species, the spatial dynamics of the fishing fleet, and the other management controls in place (Walters 2000). Like other fisheries management controls, spatial closures need to be properly evaluated before they are implemented and this may be done through empirical studies or models.

1.2 Assessing closures with empirical studies

One way to assess spatial closures for fisheries management is to directly measure their effects. Several studies have shown increases in abundance and size of animals within closures compared with adjacent fished areas (Buxton & Smale 1989, MacDiarmid & Breen 1993, Cole 1994). However, it is more difficult to measure the extent of a spill-over effect. Rates of movement of tagged individuals from a closure to fished areas have been estimated in some studies (Davis & DODRILL 1980, Bryant et al. 1989, Bohnsack 1990, Zeller & Russ 1998), but, very few studies have measured the effect of export of adult biomass on local fisheries yields (Alcala & Russ 1990, Russ & Alcala 1997).

An increase in the abundance of larger more fecund fish within closures has also been documented (Davis & DODRILL 1980, MacDiarmid 1989, Bohnsack 1990, Cole et al. 1990). Although total egg production may be increased within a closure, there is very little empirical evidence of enhanced recruitment within closures or adjacent fished areas. This is probably because, for most target species, highly dispersed larvae means that local recruitment is decoupled from local adult fecundity. Enhanced recruitment is extremely difficult to demonstrate empirically, because of the high variability in the relationship between egg production and

recruitment to a fishery and the lack of comprehensive long-term studies of fisheries following the introduction of closures.

Much of the empirical evidence on the effectiveness of closures as a fisheries management tool is correlative and thus equivocal. As with other fisheries management controls, spatial closures have effects over much larger temporal and spatial scales than are amenable to experimental studies.

1.3 Assessing closures with models

To assess closures in the absence of large scale experiments, simulation experiments may be done by integrating information on the spatial and population dynamics of the fishery (Polacheck 1990, DeMartini 1993, Man et al. 1995, Guénette & Pitcher 1999, Maury & Gascuel 1999, Nowlis & Roberts 1999). Most of the models used to assess closures have been generalised, examining factors that may influence the potential of closures to be an effective tool for fisheries management. In general, these models have not been used to provide specific recommendations on closures for actual fisheries (Guénette et al. 2000). For modelling to be a pragmatic tool for the assessment of closures in fisheries management we believe the following five key areas need to be addressed.

1.3.1 Integration of management

Previous approaches to modelling have demonstrated benefits from closures in the absence of other controls (Polacheck 1990, DeMartini 1993). In reality, however, many fisheries closures will be implemented as part of a management regime that includes other controls. Closing a proportion of the fished area reduces the exploitation rate on the entire population, which in turn increases the average size of stock, egg production, and total biomass. However, it is not clear that the reduction in exploitation rate caused by closures is any more effective than reductions in exploitation rates caused by other management measures, such as effort controls, catch controls, or size limits (Bohnsack 2000). It is essential when assessing closures, therefore, that they be modelled in unison with other strategies that are already, or likely to be, implemented in the fishery.

1.3.2 Spatial realism

Almost all models used to assess closures have assumed a homogenous environment (Maury & Gascuel 1999) making some proportion of it unavailable to fishing. There are very few examples of models that have examined the effect of closures in actual physical locations. However, the peculiar combination of habitat distribution and hydrodynamics in a particular area will have important consequences for the optimal placement of closures. For instance, placing a closure in an area of higher fish density could cause overfishing of other, less productive areas. In order to provide practical advice on where to place closures, models need include a higher degree of spatial realism.

1.3.3 Fisher dynamics

Most models of closures have focussed on modelling fish movements and have ignored the spatial dynamics of fishers. While this suffices in homogenous environments, in the real world fishers allocate their fishing effort in space in response to the distribution of fish and costs associated with fishing an area. How fishers redistribute their effort when an area is closed to fishing will have important ramifications for the placement and configuration of closures.

1.3.4 Including uncertainty

Most of the previous approaches to modelling closures have not accounted for uncertainty in the effects of closures in a systematic way (Mapstone et al. 1996). Firstly, parameters of the population and spatial dynamics have generally been assumed rather than estimated from actual data. Secondly, variability in the environment, such as spatial and temporal variation in recruitment, has generally not been incorporated. Thirdly, the regulation of the closure is assumed to be perfect, that is, complete compliance, with zero fishing mortality in closures. These sources of uncertainty can, however, be systematically incorporated using techniques commonly used in modern fisheries assessments.

1.3.5 Performance indicators

Marine protected areas are often seen as a catch-all conservation measure, supporting a wide range of often poorly defined and conflicting objectives (Jones et al. 1993, McNeill 1994). Further, objectives are rarely defined quantitatively, making assessment of the performance of closures problematic. Even when the objectives of closures are limited to fisheries management there have been a number of ways in which performance has been measured. For example, most modelling of closures has used long-term yield to measure the benefits to the fishery, but yield is a crude measure for assessing benefits from any management control. Other performance indicators may be more appropriate for assessing closures, for example, one that incorporates the economic cost that may arise from consequent changes to fishing patterns as a result of a closure. Determining specific and quantitative management objectives will be essential in assessing the performance of closures in meeting their objectives.

1.4 This study

In this report a framework is developed for assessing spatial closures as a complementary tool for fisheries management in New Zealand. This is the first activity of New Zealand Ministry of Fisheries project ENV1999/04: *Use of areas closed to fishing as a diversification strategy to reduce the risk of overfishing*. The overall objective of this project is 'to evaluate the feasibility of determining the size and number of areas closed to fishing needed to buffer against the results of unsustainable catch limits or other management measures in NZ waters.' The feasibility of applying the framework to a New Zealand fishery is tested in a later stage of this project.

We have aimed to develop a framework that can be used in real fisheries to provide management advice on closures. It addresses the five key areas described above which we contend should be an integral part of any evaluation of the potential utility of closures for fisheries management. We have placed considerable emphasis on making the framework generic so that it can be applied to a variety of fisheries with different fish and fisher dynamics, and regulations.

Section 2 describes the general approach taken in the framework. Sections 3 & 4 deal with spatial population dynamics and spatial effort dynamics, respectively. Both sections provide a brief overview of alternative approaches, underlying assumptions and constraints, followed by a more detailed parameterisation of the approach selected for the framework. Section 5 identifies the issues that relate specifically to the integration of the spatial and populations dynamics. In particular, the questions of appropriate scale, the degree of spatial dimensioning of the population model, and parameterisation of the stock-recruitment function are addressed. Section 6 outlines an approach to evaluating the potential benefits of closures in concert with more conventional fisheries management strategies, such as catch and effort controls and technical measures. There is a focus on explicitly incorporating important sources of uncertainty into the assessment of alternative closure regimes and how to present the outcomes so the trade-offs between alternatives are transparent. Key issues associated with the feasibility of formally evaluating closures in a fisheries management context are reiterated in Section 7 and areas requiring further development are highlighted.

2. FOUNDATION OF THE FRAMEWORK

Rather than attempt to develop a generic spatial stock assessment model, we base the framework on spatial extensions to existing stock assessment models. For a particular fishery, the current stock assessment model should contain the best representation of the current knowledge of the population dynamics of the stock. It should also include the management measures currently in place for the fishery and the principal sources of uncertainty to be taken into account. Thus, rather than develop a new, spatially explicit model for assessing closures, it is more appropriate to extend the existing model.

Space can be treated just like any other index in a fisheries model. For example, size-structured models represent the stock in a number of size classes (levels of the size index) and specify the transition of fish among size classes through growth. Space-structured models represent the stock in a number of areas (levels of the space index) and specify the transition of fish among areas through movement.

In a model of a fishery, the dynamics of fishing is as important as the dynamics of the fish population (Hilborn et al. 1995). A size-structured model defines how fishing mortality is distributed over size classes, usually using a selectivity curve. Similarly, in a space-structured model it is necessary to describe how fishing mortality is distributed across areas using an effort allocation model.

Models may be structured by more than one index, for example age and sex. Each additional index replicates the population in each of its own levels. Extending a stock assessment model spatially thus requires the population to be subdivided in each area and for the spatial dynamics of the fish and fishers to be specified. The existing model, thus extended spatially, provides the basis for investigations of the relative performance of closure options.

Alternative models for the spatial dynamics of fish populations and fleets are briefly reviewed in the following sections. Preferred alternative models, based upon considerations of generality of application, are then selected.

3. MODELS FOR THE SPATIAL DYNAMICS OF FISH

3.1 Assumptions

(i) Aggregate movement. Individual-based models of movement have been used to model spatial dynamics (Ault et al. 1999). However, aggregate models can usually be derived from probabilistically defined individual behaviour. Such models are less computationally intensive and allow for better estimation of movement parameters. Examination of models for the framework is restricted to those that describe aggregate movement of populations.

(ii) Markovian movement. Only Markovian models of fish movement are considered in the framework, i.e., movement determined by the position currently occupied by the individual and not by any past movement. This assumption is unsuitable for homing species, such as salmon or eels, but is probably reasonable for most marine species.

(ii) Spatial scale. We examine models of fish movement that are appropriate to the scale at which closures are likely to be implemented, (i.e., 1–100 km).

3.2 Approaches

3.2.1 Proximate models

Movement between areas can simply be specified by the proportion (or number) of individuals that move from each area to every other area. Such models are proximate, rather than mechanistic, in that they describe the movement of fish rather than the reasons for it. The model does not explicitly attribute movement of fish to any characteristic of the areas, including their relative position in space. For instance, movement to a distant area may be greater than movement to a nearby cell.

In a matrix of probabilities of movement from one area to another, each off-diagonal element is a separate parameter of the model, $\theta_{i \rightarrow j}$, and the diagonal elements equal,

$$\text{Equation 1} \quad \theta_{i \rightarrow i} = 1 - \sum_{j \neq i} \theta_{i \rightarrow j}$$

This is the most flexible model for movement, but is suitable only when there are only a few areas (2–10) under consideration (Maunder 1998, Cordue 1999). When there are many areas, the number of parameters that need to be estimated increases rapidly; for example, with 20 areas up to 380 parameters could be estimated. The level of parameterisation can be reduced by making assumptions about movement, for instance by restricting which areas have a transfer of fish. This type of modification begins to imply a spatial structure between areas (Fournier et al. 1998).

3.2.2 Distance models

An alternative to proximate models is to provide an explanatory model for the different rates of movement between areas, the most obvious such explanatory variable being distance. A simple model of movement could use a linear relationship between the distance between two areas and the probability of movement.

Models used in physics to explain the diffusion of particles may also be useful. The simplest diffusion model of fish movement is that they have Brownian random motion at a rate that is constant over space (Holmes et al. 1994). The resulting change in numbers in an area is proportional to the gradient in numbers at that point. The resulting diffusion model in one dimension X is,

$$\text{Equation 2} \quad \frac{\partial N}{\partial T} = \frac{\partial \left(\kappa \frac{\partial N}{\partial X} \right)}{\partial X}$$

where N is the number of fish, T is time, and κ is the diffusivity. (For simplicity this and subsequent differential equations are in one dimension only, but can be extended to two dimensions by repeating the terms in the Y dimension).

Random undirected movement is likely to be uncommon for fish. Directed movement can be modelled through the addition of an advection term to the diffusion model,

$$\text{Equation 3} \quad \frac{\partial N}{\partial T} = \frac{\partial \left(\kappa \frac{\partial N}{\partial X} \right)}{\partial X} - \frac{\partial (uN)}{\partial X}$$

where u is the velocity in the X dimension. In a two dimensional model, κ is the same in both dimensions but there is a perpendicular velocity acting in the Y dimension, v . The combination of the two velocities determines the resultant direction and velocity of movement in the two dimensions.

Such advection-diffusion models are ideal for modelling the movement of passively drifting larvae where the velocity in the advective term is related to water currents. However, it can also be applied to adult fish. Sibert et al. (1999) used this model for adult tuna with different north-south and east-west velocities in a number of regions. A similar model can be derived by replacing the perpendicular velocity terms with a single bearing and velocity of movement (Deriso et al. 1991).

Advection-diffusion models are a useful tool for modelling movement. However, they do not attribute a cause to movement and to effectively capture spatial heterogeneity in the rates and direction of movement it may be necessary to estimate a large number of velocities in different areas. Consequently, these models may require large amounts of high quality data on movement such as those derived from tagging studies.

3.2.3 Preference models

A simple advection-diffusion model uses a velocity term to describe how fish move. It does not model why fish move. In a spatially heterogeneous environment a better model might incorporate explanatory variables that more closely describe why fish move from one area to another. Certain areas are more suitable for fish than others because of their physical and biological characteristics. Fish would be expected to move towards areas that are the most suitable with a certain velocity. To model this, the velocity term in Equation 3 can be replaced by the gradient in habitat preference, P , and a coefficient of responsiveness η that describes the speed of fish movement in response to that gradient,

$$\text{Equation 4} \quad \frac{\partial N}{\partial T} = \frac{\partial \left(\kappa \frac{\partial N}{\partial X} \right)}{\partial X} - \frac{\partial \left(\eta \frac{\partial P}{\partial X} N \right)}{\partial X}$$

This preference model is the basis of movement in MacCall's (1990) 'Basin Model' of fish spatial dynamics, in which population growth rates are also related to habitat suitability. It has been found to be an effective model of fish movement (Bertignac et al. 1998, Maury et al. 2001) and has been applied to simulating the impact of fisheries closures (Maury & Gascuel 1999).

The advantage of preference models lies in their ability to model spatial heterogeneity in movement without increasing parameterisation. Rather than using a large number of parameters to describe movement in each area, they use additional information to derive a suitability index that inherently captures spatial differences among areas and movement of fish between them. Although parameters are involved in building the index, these are relatively few in number. Because of this power of preference models to describe fish movement with minimal parameterisation, we emphasise their use in our framework.

3.3 Parameterisation

The choice and parameterisation of movement models will depend upon the movement characteristics of the species and information available. The parameterisation of models for passive dispersal and active movement in response to a preference gradient is discussed below.

3.3.1 Passive movement

For passively dispersed phases of the life-cycle, a simple advection-diffusion model is ideal for modelling movement. In two dimensions the advection-diffusion model is,

$$\text{Equation 5} \quad \frac{\partial N}{\partial T} = \frac{\partial \left(\kappa \frac{\partial N}{\partial X} \right)}{\partial X} + \frac{\partial \left(\kappa \frac{\partial N}{\partial Y} \right)}{\partial Y} - \frac{\partial (uN)}{\partial X} - \frac{\partial (vN)}{\partial Y}$$

Various degrees of complexity of passive dispersal can be modelled with alternative values of u and v . The following three examples of parameterisation illustrate some possible alternatives, depending on the information available on water currents.

When there is no information available on currents or there is no net water movement in any direction, then a simple diffusion model is appropriate. This can be achieved by setting u and v to zero. Diffusion results in eggs from a single point having a normal distribution with a mean at that point and with a mean squared displacement of $4T\kappa$, where T is the time since release (Holmes et al. 1994). Thus the value of the diffusivity, κ , can be used to alter the degree of random passive dispersal.

When all areas experience the same unidirectional current, then u and v are constant across all areas. In these cases the combination of u and v determines the average direction and velocity of movement in the X and Y directions. This will act in addition to diffusion such that eggs from a single point will have a normal distribution but with a mean determined by the resultant velocity. An example is a coastline with a consistent current flowing in one direction along it.

Where there is knowledge of current velocities in each area, such as from hydrodynamic models or current measurements, then area specific values of u and v can be specified (Chiswell & Booth 1999). This allows for more realistic simulation of spatial variations in movement velocities.

3.3.2 Preference indices

In two dimensions the preference model of active (post-settlement) movement is,

$$\text{Equation 6} \quad \frac{\partial N}{\partial T} = \frac{\partial \left(\kappa \frac{\partial N}{\partial X} \right)}{\partial X} + \frac{\partial \left(\kappa \frac{\partial N}{\partial Y} \right)}{\partial Y} - \frac{\partial \left(\eta \frac{\partial P}{\partial X} N \right)}{\partial X} - \frac{\partial \left(\eta \frac{\partial P}{\partial Y} N \right)}{\partial Y}$$

In this case the diffusivity, κ , represents the degree of random movement, that is, movement that is not associated with the gradient in habitat preferences.

The model requires the specification of an index describing preference for an area. The preference index is calculated for each area from some function, $f(\cdot)$, of a number of variables that describe the current state of that area,

$$\text{Equation 7} \quad P_i = f(x_i^1, x_i^2, x_i^3 \dots x_i^4)$$

This section proposes a general approach to specifying the preference function. This involves specifying the response to each variable and how to combine the individual responses.

3.3.2.1 Units of preference

Each variable, x^j , in the preference function has a preference, P^j , associated with it which is determined by the preference function for that variable, $P_i^j = f^j(x_i^j)$. We consider the case in which the response to each variable is unimodal, that is, there is a single optimum value for each variable and we arbitrarily set the index to one at that value, $P^j = f^j(\phi^j) = 1$. The preference at values other than the optimum will depend upon the tolerance for that variable. We propose two ways of specifying this tolerance depending on whether the variable is continuous or categorical.

3.3.2.2 Continuous variables

A range of unimodal functions is available for continuous variables such as temperature, salinity, and depth. The Gaussian response curve can be used but assumes symmetry in the response to high and low values of the variable. A compound normal curve which allows for asymmetry in the response to a variable is more flexible. This model can be parameterised so that it is expressed in terms of the ‘optimum’ and ‘tolerance’ of preference to the variable:

$$\text{Equation 8} \quad P_i = \begin{cases} \exp\left(\ln(0.1) \frac{(x_i - \phi)^2}{(\nu - \phi)^2}\right) & \text{for } x_i \leq \phi \\ \exp\left(\ln(0.1) \frac{(x_i - \phi)^2}{(\omega - \phi)^2}\right) & \text{for } x_i > \phi \end{cases}$$

where ϕ is the optimum, ν is the lower value at which preference is 0.1, and ω is the upper value at which preference is 0.1. The range $\omega - \nu$ can be used as a measure of the tolerance for a particular variable. The greater this range, the greater the tolerance.

This function has the flexibility to approximate a number of shaped responses. A normal response curve is achieved when ν and ω are equal. A logistic response curve (Bertignac et al. 1998, Maury et al. 2001) is approximated when ω is very large (Figure 1).

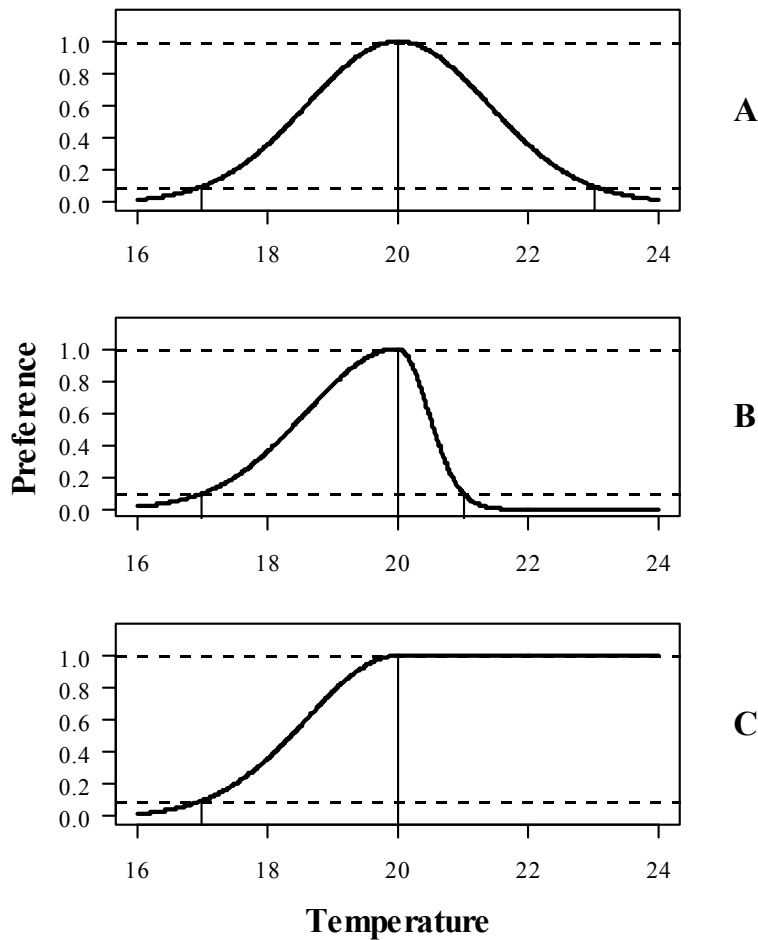


Figure 1. Examples of preference functions for water temperature using Equation 8. In all cases $\nu = 17, \phi = 20$; (A) $\omega = 23$; (B) $\omega = 21$; (C) $\omega = \infty$.

3.3.2.3 Categorical variables

The relative preference for categorical variables, such as substrate type or relative exposure, needs to be specified by a preference value for each level of the variable. For example, for a variable representing four types of substrate, $x = \{1,2,3,4\}$, a parameter is defined having a level associated with each type, $\alpha_1, \alpha_2, \alpha_3, \alpha_4$, and in area i the preference depends upon the substrate type,

$$\text{Equation 9} \quad P_i^s = \alpha_{x_i}$$

Tolerance for categorical variables can be measured simply using the mean of $\alpha_1, \alpha_2, \dots, \alpha_n$. A mean of 1 indicates no preference, whereas a mean of $1/n$ indicates no tolerance, for example, that the species is only found in one substrate type.

3.3.2.4 Density dependence

The number of conspecifics in an area may alter the preference for a particular variable. For example, the preference of fish for the abundance of prey species is likely to be affected by the number of conspecifics

competing for the food source. Such effects may be particularly important when considering marine reserves because they determine the degree of spill-over from a preferred, yet densely populated, area.

Models commonly used to estimate density dependence in growth rates of populations can be used to describe density dependence of habitat preference (May & Oster 1976, Bellows 1981). A model that is intuitively appealing assumes that preference is 1 up to a threshold population size beyond which it decreases at a rate determined by a sensitivity parameter,

$$\text{Equation 10} \quad P_i^N = \begin{cases} 1 & N \leq N^* \\ 1/\left(\frac{N}{N^*}\right)^\gamma & N > N^* \end{cases}$$

where N is the population size, N^* is the threshold populations size, and γ is the sensitivity parameter. If $\gamma = 1$, then the rate of decrease is on a per capita basis. For example, preference will be 0.5 when the population is twice the threshold. The rate of decrease above the threshold increases as γ increases (Figure 2).

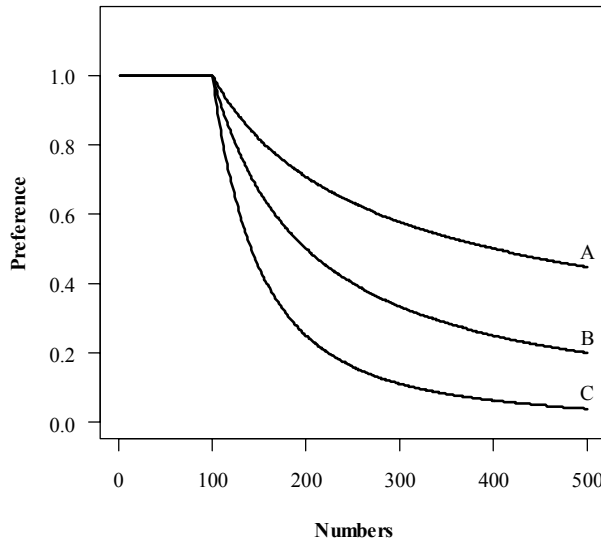


Figure 2. Examples of the density dependent function in Equation 10. In all cases $N^* = 100$; (A) $\gamma = 0.5$; (B) $\gamma = 1$; (C) $\gamma = 2$.

3.3.2.5 Combining indices

The preference for several variables can be combined into a single preference index using a multiplicative model, which means that if the value of preference for any one variable is zero then the overall preference will be zero. Using a geometric mean to combine preferences provides an index that is not dependent upon the total number of variables used (Rubec et al. 1999),

$$\text{Equation 11} \quad P_i = \left(\prod_{j=1}^n P_i^j \right)^{\frac{1}{n}}$$

The maximum possible value of the overall preference index is 1. This occurs in an area where all factors are at their optimum.

It is not necessary to weight each of the individual preference indices because the movement of fish is determined by the gradient in the preference index between areas, not by the magnitude of the index. The relative effect of a particular variable on the index will depend upon the tolerance for that variable and on the variation in that variable among areas. That is, if fish are unresponsive to a variable over the range observed then it will contribute little to the gradient in preference indices.

4. MODELS FOR THE SPATIAL DYNAMICS OF EFFORT

Modelling fleet dynamics is not well developed relative to the sophisticated theory and methods available for modelling dynamics of fish populations (Hilborn & Walters 1992, Holland & Sutinen 1999). The widespread neglect of the dynamics of exploiters has been described as 'perhaps the biggest failure' of natural resource management (Hilborn et al. 1995). It is usual for stock assessment models to assume that fishing effort is distributed evenly over the range of the stock. Clearly, this is not the general case. Most fish stocks exhibit some degree of spatial structure over their range, and it is unrealistic to assume that fishers will not respond to this spatial heterogeneity in abundance.

Understanding the factors which influence the spatial distribution of effort provides the basis for predicting how fishers will respond to changes in the status of the resource and the regulatory environment. Given that closures directly alter the distribution of the available resource, forecasting the change in distribution of effort is a key component of assessing the effects of different closure strategies (Holland & Sutinen 1999, Walters & Bonfil 1999).

4.1 Assumptions

(i) Fixed total effort. Most models of spatial dynamics assume fixed total effort, or use historical levels of annual effort for the fishery to simulate inter-annual variation in total effort (but see Allen & McGlade (1986) who use a Volterra-Lotka predator-prey model to model total effort). As spatial closures will result in an increased exploitation rate in the areas remaining open to fishing, it will be particularly important to include the response of total effort to different closure regimes in evaluations. The simplest way of doing this would be to run scenarios with alternative fixed levels of total effort. A more dynamic alternative would be to set arbitrary or empirically based minima for catch rates, or profit, below which effort leaves the fishery according to some function (Hilborn & Walters 1987).

(ii) Perfect knowledge. Most models of spatial dynamics of effort are based on the concept of maximising utility, that is, effort will be allocated among areas, according to the preference of fishers for characteristics of each area, so that they maximise the utility realised from fishing. Most attempts of modelling spatial effort allocation have assumed 'perfect knowledge' of this utility across areas. This implies that fishers have perfect information on the distribution of fish, other fishing vessels, and resource abundance for each alternative ground. In reality, this is unlikely to be the case. There is almost always going to be some degree of uncertainty associated with the various forms and sources of information individual skippers use to make their decisions on when and where to fish. Techniques for incorporating imperfect knowledge, information flow and learning in modelling of fleet dynamics, however, are largely undeveloped. Allen & McGlade (1986) explicitly modelled the level and direction of information flow within and among two fleets fishing the same grounds using an information matrix in a simulation study of the dynamics of discovery and exploitation. Their results clearly indicated that the level of information flow between the two fleets will have a substantial effect on the distribution of effort and resulting pattern of exploitation. We do not consider the 'perfect knowledge' assumption further. However, it is evident from our review of fleet dynamics literature that approaches to explicitly incorporate 'imperfect knowledge' and 'learning by experience' in effort dynamics models are poorly developed and would be a fruitful area of research.

4.2 Approaches

A model is required that will describe how total effort, \dot{E} , will be distributed across j areas. When the utility of an area does not depend upon the amount of effort allocated in that area, and given perfect knowledge, then all the effort will be allocated to the single area with the highest utility. However, at the spatio-temporal scales likely to be used for modelling effort dynamics in evaluating closure regimes, the utility of an area will depend upon the number of units of effort already allocated to it. If we consider the sequential allocation of effort, given a utility function that includes as a variable the amount of effort allocated to the area, then the first unit of effort will be allocated to the area with the highest utility (Hilborn & Walters 1987). Effort will continue to be allocated to the same area until the utility of that area declines to the same level as the area with the second highest biomass. Effort will then be allocated to that area and so on until utility is equal in all areas. We refer to the distribution of effort among areas at this point as the equilibrium distribution of effort. Given perfect knowledge, effort will distribute itself so that the utility derived from each fished area is equal.

A number of models use this approach to describe how fishing effort distributes itself with respect to itself and the abundance/biomass of the resource. The most commonly used methods are the proportional allocation model (Caddy & Seijo 1998), the gravity model (Walters et al. 1993, Walters & Bonfil 1999) and the Ideal Free Distribution (Fretwell & Lucas 1970, Fretwell 1972, Abrahams and Healy 1990, Gillis et al. 1993, Gillis & Peterman 1998). These models can all be derived from a simple utility function determined by the abundance of the resource and effort, such that each additional unit of effort allocated to an area causes a proportional decrease in utility,

$$\text{Equation 12} \quad U_i = \frac{B_i}{E_i}$$

where B_i and E_i are the biomass and the amount of effort present in area i .

Given this utility function, if we assume that effort can be allocated in infinitesimally small units, then all areas would be expected to be allocated at least some effort predicted by,

$$\text{Equation 13} \quad E_i = \frac{B_i}{\dot{U}}$$

where \dot{U} is the equilibrium value of the utility. \dot{U} can be calculated numerically, but given the assumptions of infinitesimally small units of effort can be derived analytically as follows. Given that at equilibrium \dot{U} will occur in all areas, and we know the total effort to be allocated then,

$$\text{Equation 14} \quad \dot{E} = \sum_i E_i = \frac{\sum_i B_i}{\dot{U}}$$

which rearranging gives,

$$\text{Equation 15} \quad \dot{U} = \frac{\sum_i B_i}{\dot{E}}$$

By substituting Equation 15 back into Equation 13, the effort allocated to each area can be calculated simply as,

$$\text{Equation 16} \quad E_i = \frac{B_i}{\sum_i B_i} \dot{E}$$

This equation is the basis of the proportional allocation and gravity models described above. In addition to the perfect knowledge and the form of the utility function these models make the additional assumption that effort can be allocated in infinitesimally small units. However, the above principal of equilibrium allocation of effort can be used without this assumption to model effort allocation. In the following sections we examine alternative formulations of the utility function and demonstrate how these can be used in models of the spatial allocation of effort.

4.3 Parameterisation

4.3.1.1 Units of utility

It is most appropriate to make the unit of the utility index monetary. Profit, the difference between total revenues and total costs, is likely to be the major determinant of the utility of an area,

$$\text{Equation 17} \quad U_i = R_i - C_i$$

In addition, there is often information readily available to provide the relative weighting of variables of cost and revenue. For instance, consider a utility function that is based upon catch and distance from port. One represents revenue and the other a cost. Instead of using arbitrary coefficients, the relative weighting of the two variables comes naturally from the price of fish and the cost of travel respectively.

Empirical studies of spatial allocation of effort have demonstrated that a range of factors, such as distance from port, expected catch rates, profits, skipper skill and knowledge, exposure of the ground to weather, and risk of realising expected returns, will influence preference of fishers for different areas and determine, to various extents, the distribution of effort among areas (Hilborn & Ledbetter 1979, Holland & Sutinen 1999). Such factors have generally been incorporated into models of spatial effort allocation through the utility function as a relative benefit or cost associated with fishing each area (Walters et al. 1993).

4.3.1.2 Revenue

In most fisheries, the single most important factor determining revenue is the amount of fish caught. This depends upon the number of fish present and the efficiency of the gear, which in turn depends upon the amount of effort in the area. Catch rates for individual vessels may decline due to two forms of competition: i) interference, or ii) exploitative (Gillis et al. 1993). Exploitative competition arises due to a reduction in the available biomass due to fishing by other units of effort. Interference competition arises from the fishing activity of each additional unit of effort reducing overall fishing efficiency in the area. Distinguishing between interference and exploitative competition in reality will be difficult because the observed effects of each may be similar (Gillis & Peterman 1998).

The utility function described in Equation 12 assumes that either form of competition will lead to a proportional reduction in revenue. However, the degree to which catch rates will be reduced will depend on the nature of the fishing activity, the efficiency of the gear, the proportion of the stock that is vulnerable, and the spatial and temporal scale at which the fishery operates. It would be desirable, in the context of developing a general framework for modelling spatial effort dynamics, to accommodate a range of assumptions as to the form of competition among vessels.

Exploitative competition can be modelled explicitly. Each unit of effort takes a proportion, q , of the available biomass. The biomass remaining will be the starting biomass less the biomass removed by previous units of effort and can be described as an exponential decline,

$$\text{Equation 18} \quad R_i = B_i e^{-qE_i}$$

The relationship between utility and resource abundance can thus be expressed in terms of the catchability coefficient, q , which will be dependent upon the spatio-temporal scale used. For instance, q will be less when a large time step and large areas are used than when these are small.

Interference competition acts by reducing the catching efficiency of each unit of effort, that is, by reducing q . One way that this can be modelled is by adding an interference term to Equation 18 so that the effect of the number of vessels is increased,

$$\text{Equation 19} \quad R_i = B_i e^{-qIE_i}$$

4.3.1.3 Costs

Different areas have different costs associated with fishing them. The most obvious are travel costs, which depend upon the distance from port. However, there are other costs that are less obviously related to the financial aspects of fishing. These can be incorporated by using coefficients that place a 'dollar value' on that variable. For instance, in a trawl fishery, gear loss may be related to the type of substrate. The coefficient associated with each substrate type will reflect the risk of gear loss in that area and the cost of gear loss. Rougher substrate types will have a greater risk of losing fishing gear and thus a higher coefficient. Similar approaches can be used to include other variables associated with an area such as exposure or density of undesirable bycatch species.

5. INTEGRATING SPATIAL AND POPULATION DYNAMICS

Having defined models for the spatial dynamics of fish and fishers, it is necessary to integrate these spatial extensions with the existing stock assessment model. The result of this integration will be a fisheries stock assessment model that is spatially explicit and thus can be used to evaluate the management performance of spatial closures. This section examines important issues involved in this integration: (i) the degree of spatial resolution, (ii) the dimensioning of population dynamics parameters and variables, and (iii) the dimensioning of spatial dynamics parameters and variables.

5.1 Spatial resolution: trade-offs between ideal and pragmatic

Ideally, the spatial resolution of the model (the smallest scale at which the fishery is modelled) will be fine enough that the size and shape of actual closures can be represented. In most cases, there is a trade-off between having enough resolution to simulate the effects of individual closures and having enough resolution in the available data. Closures that are smaller than the spatial resolution of the model can be simulated by making only a proportion of the biomass in that area available to fishing. However, this may not be realistic if movement rates are slow relative to the size of the area or there is significant habitat heterogeneity within an area (Walters & Bonfil 1999).

5.2 Spatial dimensioning of population dynamics

In making a model spatially explicit it is at least necessary to keep track of the numbers of fish at age within each area. The minimum degree of spatial dimensioning in the population dynamics is to replicate the stock in each area and have the same population dynamics in each area. However, population processes such as growth and mortality may differ between areas and an important issue in developing an integrated model is determining to what degree these should be spatially explicit. In the most complex case, all population dynamics parameters would be specified for each area. However, there is unlikely to be the information available to estimate these parameters and the decision for each parameter will be dependent upon knowledge of its spatial heterogeneity. If a parameter is expected to vary significantly over a larger spatial scale than the areas defined in the model, regions (groupings of areas) can be used to define different levels of the parameter.

5.3 Population dimensioning of spatial dynamics

The converse of applying the spatial dimension to the population dynamics parameters is to apply the population dimensions to the spatial dynamics parameters. The parameters that specify the movement of fish may differ between components of the population. For example, if the population is age-structured it may be appropriate to define age-specific movement parameters and preference indices. For example, juvenile and adult fish may have different optimum depths. In other cases, these population stages may have the same optimum value for variable but different tolerances. Fournier et al. (1998) used a model for the relationship between age and diffusion rate. Preference functions can also be separately defined for different periods (e.g., seasons).

5.4 An example of model integration: stock-recruitment relationships

Many fisheries assessment models do not explicitly model the processes affecting the survival of pre-recruits. Often these processes are condensed into a stock-recruitment relationship that describes the relationship between the number of eggs produced and the number of larvae surviving through to recruitment to the fishery. In other cases, it is assumed that there is no relationship between egg production and recruitment and that environmental variability is of primary concern.

Despite what assumptions are made for the stock-recruitment relationship for the whole stock, spatial variation in the survival of eggs to recruits is likely to be important in determining the effectiveness of fisheries closures. For instance, a closure that is surrounded by areas of poor habitat for juveniles is likely to contribute fewer recruits than an area located near good juvenile habitat. This spatial variation in pre-recruit survival can be included by modelling pre-recruit habitat preference and linking it to survival. For example, a pre-recruit mortality could be assumed directly proportional to habitat preference,

$$\text{Equation 20} \quad N_i' = E_i P_i' \rho$$

where in area i , N_i' is the number of recruits, E_i is the number of eggs or larvae setted in the area, P_i' is the preference of pre-recruits for the area, and ρ is the maximum survival rate of pre-recruits.

An alternative is to include density dependent effects. This could be done by incorporating the preference index into a stock recruitment relationship. For example, the Francis (1992) parameterisation of the Beverton-Holt relationship could be modified by changing R_0 from the recruitment to the whole stock in a virgin population to recruitment to a cell in the virgin population and scaling it by P_i' . Alternatively, Equation 10 could be used with N^* scaled by P_i' .

6. ASSESSMENT OF SPATIAL CLOSURES FOR FISHERIES MANAGEMENT

6.1 Choosing closure regimes for evaluation

Spatial closures can be separated into two broad categories:

- (i) permanent spatial closures, which in turn may include gear specific spatial closures, and;
- (ii) temporary spatial closures, which incorporate rotational closures and seasonal spatial closures.

Most spatial closures that have been implemented have been, or intended to be, permanent. Generally, they have excluded all forms of fishing and other exploitation. Permanent closures are also the category of spatial closure considered in most modelling studies. Objectives, from a fisheries management perspective, include protecting juveniles and their habitat from fishing, reducing the proportion of the exploitable stock that is available for harvest, protecting habitats from the effects of particular gears (e.g., trawl), and rebuilding over-fished stocks (Roberts & Polunin 1991, Guénette et al. 1998).

Temporary spatial closures have been used to protect harvested stocks during vulnerable stages of their life history or to maximise the size or value of yield. Seasonal closures are probably the most commonly used closure type in this category, protecting breeding aggregations and juvenile nursery areas (McNeill 1994). The use of rotational closures, with or without permanent closures, is less common. This is despite their theoretical advantages, particularly for sessile benthic fisheries (e.g., Sluczanowski 1984, Bohnsack 1996).

Considering these two main categories of spatial closures and types within, there is an infinite range of possible combinations of closures. However, in this study we will be considering permanent total closures only. Regardless, even from a relatively simple spatial model, a large number of different closure regimes may be tested. For example, a 20% closure of a model with 50 areas results in over 1 000 000 000 possible combinations of closures that could be tested. If other closure types (e.g., gear or seasonal closures) were also incorporated, the possible combination becomes extremely high. It is not feasible to test all combinations. A subset will usually be chosen for testing based upon the life history strategy of the species, the spatial and temporal dynamics of the species and fishery, and the most practical configuration for enforcement.

The optimal physical design of closures (size, shape, number, and location) has been intensely debated (McNeill 1994) and much literature exists on the criteria required to design optimal marine reserves. In general, however, there has been little consideration given to the criteria required for fisheries conservation and enhancement in the design of marine reserves. Rather, as for most marine reserves, the design of fisheries closures has been ad hoc (Bohnsack 1996, Roberts 2000). To effectively conserve and enhance a fishery a closure must not only provide protection to a proportion of the stock, but also, through export of larvae or harvestable adults to fishable areas, maintain or increase productivity in the open areas. The effectiveness of a closure will, therefore, greatly depend upon the spatial dynamics and mobility of the target species. Knowledge of the scale of dispersal will help define the most beneficial size and arrangement of closures. Optimal closure design requires detailed knowledge of the life history characteristics of the target species and temporal and spatial dynamics of the fishery. There are many social and economic criteria (e.g., public support, enforcement, compliance, fishing markets, fleet investments) which should be considered in the physical design of a closure which are beyond the scope of this project and will not be discussed here. Below, a brief review of physical design considerations relevant to fisheries closures is provided.

6.1.1.1 Size

Area is possibly the most controversial aspect of closure design (Bohnsack 1996). Ultimately, the optimal size of a closure will be species specific, determined by factors such as the steepness of the stock recruitment relationship and dispersal of a species (Rowley 1992, Carr & Reed 1993, Dugan & Davis 1993, Walters 2000). However, because knowledge of the distribution and dispersal potential of life history stages is often unknown, it is often difficult to predict the optimal size of a closure. In general, closures may conserve site-attached or territorial species, or specific life history phases of migratory species but will have limited application to pelagic or highly migratory species (Nowlis & Roberts 1999).

There has been no field testing of the effect of size of closure on fisheries. Further, closures declared to date have been too small to allow a comprehensive evaluation of the effects of closures (Dayton et al. 2000). However, several studies, using models, have investigated the effect of closure size on fisheries yields (Guenette et al. 1998, 2000, Nowlis 2000). The majority of these studies investigated the effect of size by closing a proportion of the management area. Results from these studies recommended between 10 and 80% closure of total habitat of a species (Ballantine 1991). Size of a closure was species specific, dependent on attributes (e.g., transfer rates, reproductive strategy, life history) of a species, fishing intensity and habitat characteristics (Carr & Reed 1993, DeMartini 1993, Dugan & Davis 1993, Guenette et al. 1998, Nowlis & Roberts 1999). DeMartini (1993) showed that reserves could be smaller for short-lived, fast-growing species than for long-lived slow-growing ones. Whilst Polacheck (1990), concluded that the higher the movement rates the larger the closed area required to achieve the same benefit to the fishery.

6.1.1.2 Number

The debate over whether several small reserves enhance/protect a species to the same extent as one large reserve of the same total size continues (McNeill & Fairweather 1993). There have been few tests of predictions arising from this debate and theoretical results are equivocal. Some authors suggest one large reserve increases enhancement over several small reserves (e.g., Dahlgren & Sobel 2000), others believe several small reserves in a network will be the most effective configuration of closures (McNeill & Fairweather 1993, Carr & Reed 1993, DeMartini 1993). The benefits of several small closures are thought to ensure increased encounter by adults, juveniles, and larvae leading to an increase in overall recruitment, and to guard against recruitment uncertainty and natural catastrophe by spreading the risk (McNeill & Fairweather 1993).

6.1.1.3 Shape

The shape of closures may also contribute to their effectiveness. Increased perimeter:area ratios provided by small areas have been suggested to support more of recruits than one large area of smaller perimeter:area ratio (McNeill & Fairweather 1993, DeMartini 1993). Increased probabilities of interception by larvae and increased refuge area have been suggested to account for this result (De Martini 1993). Orienting a closure to take advantage of oceanographic processes may increase chances of encounter by adults and larvae, and increasing the perimeter to area ratio of a closure may increase dispersal out of a closure. The shape of a closure will often be dictated by the physical habitat of an area (e.g., rocky reef, seagrass bed, seamount). Increases in the edge provided by these physical structures have been shown to disproportionately increase the dispersal of individuals in to and out of a closure (Buechener 1987).

6.1.1.4 Location

The location of a fisheries closure will depend on the requirements of the life history stage/s to be protected (Crowder et al. 2000, Roberts 2000, Warner et al. 2000). Critical to effectively locating a closure will be knowledge of the spatial distribution of life history stages and the relationship among areas/habitats and oceanographic regimes. Several studies have shown a negative or no effect of a closure in areas of 'low quality' or 'marginal' habitat (Dugan & Davis 1993, Crowder et al. 2000). In this regard, foraging ranges and reproductive migrations may be protected through protection of critical habitats or alternatively specific locations known to be spawning, nursery, or feeding areas, or areas known to produce larvae and recruit juveniles may be targeted (Crowder et al. 2000, Koenig et al. 2000). Similarly, closures may benefit from their proximity to habitats used by different life stages (Peterson et al. 2000). Obviously, activities occurring beyond the range of the closure, such as pollution sources and mining, may ultimately affect its effectiveness (Roberts 2000, Simberloff 2000, Warner et al. 2000).

6.2 Management objectives and performance indicators

Although marine protected areas serve many conservation objectives, in this study, only the potential impacts of spatial closures in the context of fisheries management objectives are considered. This is not to say that the potential broader benefits of spatial closures for environmental management and protection of biodiversity do not warrant quantitative evaluation.

Operational management objectives and performance indicators (which measure the performance of a management strategy against specified objectives) are necessary for quantitative assessment of the relative performance of alternative management strategies (Smith 1996) Although most fisheries legislation incorporates objectives, they are usually framed in the form of general statements. It is necessary to 'operationalise' these objectives so that they may be quantitatively related to outputs of the model used for assessment.

Ideally, defining objectives in operational terms will involve an iterative process between assessment scientist(s) and stakeholders groups (Smith et al. 1999). Once objectives have been specified it is possible to

develop measurable performance indicators for each objective, which can be used to measure how well the objectives are being met by a given management strategy (e.g., Table 1.)

Table 1. Examples of specification of general objectives into quantitative operational objectives with performance indicators.

General Objective	Operational Objective	Performance indicator
Conservation of the stock	Maintain biomass > 30% of unfished level with a probability > 0.8	$\Pr(B_T < 0.3B_0)$, Probability that biomass in the final year of the forecast period is less than 30% of unfished level.
Maximum yield	Maximise yield over period of projection	Average total yield over the forecast period.
Social benefit (e.g., employment)	Minimise no. years in which effort is zero	Average number of zero effort years over forecast period.

6.3 Incorporating uncertainty in evaluations

Stock assessment advice is often provided as a single recommendation on the optimal level of harvest given a general fisheries management objective, such as maximising long-term yield or profit. Presenting such advice as a single optimum disguises the many uncertainties that characterise our knowledge of the dynamics of fisheries systems and the inherent tradeoffs among objectives (Hilborn & Walters 1992). Acknowledgment of this has led to the development of methods that explicitly include measures of uncertainty in the stock assessments (Francis 1992, Hilborn & Walters 1992) and the advice provided to stakeholders and managers (Francis & Shotton 1997).

Previous assessments of the potential of spatial closures for both fisheries management and conservation have often not acknowledged the different forms of uncertainty associated with the models used. Parameters have often been assumed, as opposed to being estimated from the available data. The result may be unjustified confidence in the accuracy and precision of the assessments of closures and the decisions based on them. A number of techniques are available for incorporating uncertainty into assessments and are increasingly feasible given modern computing power (Francis 1992, McAllister et al. 1994, Punt & Hilborn 1997).

7. CONCLUSIONS

Assessing the effect of spatial closures on fisheries management can, and should, be done in the same way as, and in conjunction with, other fisheries management controls. The framework developed in this study provides a means of evaluating closures as one of a suite of possible management measures, including input and output controls. The framework, based upon extending an existing stock assessment model with models for the spatial dynamics of fish and fishers, is a relatively new approach to modelling the effect of spatial closures on fisheries management. It is a more realistic and specific approach to evaluating the effect of closures as it uses existing information used to currently manage the fishery and incorporates additional information on the spatial dynamics of the target species and fishing effort. The framework developed is, however, generic enough to be applied to most New Zealand fisheries and to a range of available information.

The framework addresses the relative level of certainty of effectiveness of closures by systematically including sources of uncertainty into assessments. This is important when assessing fisheries closures as an alternative to other controls, and choosing between management controls. For example, for some stocks there will be very little information for the estimation of parameters of the spatial dynamics of fish and fishers. In these cases, there may be large amounts of uncertainty in the effects of closures and other management controls may provide better certainty of meeting management objectives.

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