EFFECTS OF SPATIAL STOCK STRUCTURE AND EFFORT DYNAMICS ON THE PERFORMANCE OF ALTERNATIVE ASSESSMENT PROCEDURES FOR THE FISHERIES OF NORTHERN AUSTRALIA

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Abstract

With the world's fisheries in crisis, most fisheries fully- or over-fished, and world catches perhaps exceeding sustainable limits, our capability to monitor and manage fisheries is uncertain. I reviewed these problems, and described ways that spatial complexity compromises monitoring and assessment. Monitoring/ management combinations that are robust to fine-scale dynamics are needed. I developed a closed-loop simulation framework, using the disc equation to distribute fishing effort. A suite of small, spatially-complex fisheries were simulated, and fishery performance was measured, under different monitoring/ management arrangements. Spatial dynamics interacted with monitoring/ control systems, engendering fine scale effects such as biomass erosion, and serial depletion. Performance depended upon control and monitoring information quality. It deteriorated as capacity and hyperstability increased. Poor information / control combinations (CPUE/ TAC) produced poor performance, especially where the stock and effort were concentrated. Effort control with monitoring fishing rates (F) by tagging was risk-averse, performing consistently well across all scenarios.

I used a single-stock, age-structured model to assess the Northern Territory Spanish mackerel fishery. This fishery's status is grossly uncertain: available abundance and composition data were uninformative; catch and effort history, biological parameters and stock structure were all uncertain. There was no evidence supporting any increase of the current limit reference point.

Genetic mark-recapture might overcome the limitations of conventional tagging. It is suggested for direct F measurement, for routine monitoring. This would entail *in situ* collection of tissue ("tagging") and subsequent screening of catch samples for matches ("recaptures"), using DNA-fingerprinting. I present device designs for *in situ* tissue collection. Success rates (proportions

of strikes yielding tissue) relative to design features were examined. Design and the line on the test vessel on which the tool was deployed interacted strongly. Predicted success rates of the best design are 44–85%, depending upon the line used. Industry participation, entailing daily genetic tagging a set number of fish, would ensure that all members of the fished population have a similar probability of being tagged. In further simulations, genetic tagging outperformed other monitoring methods. Performance improved with small concurrent conventional tagging programs. This methodology could be developed to monitor F in many fisheries.

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1. GENERAL INTRODUCTION

"How should we manage this fishery?" "What impact does fishing have on the fish population?" "What information should we be collecting?" These are, of course, questions that are fundamental to sustainability of fisheries. We might think they would be addressed carefully at the inception and repeatedly throughout the course of any fishery: fisheries are very valuable to us as a source of food, and for economic and social activity. However, the evidence of history and the current state of the world's fisheries indicate that we have not been good at confronting these questions, that we are frequently unable to address them in ways that ensure fisheries sustainability. Despite the increasing sophistication of assessment and management tools, our record of fisheries management is badly blemished.

1.1. The Sad State of the World's Fisheries: Paradigms Lost[1]

In the mid-19th Century, the world's fishery catch was a mere two million tonnes, and marine resources seemed limitless, a boundless providence (McGoodwin 1990). Over the course of a handful of generations, the catch increased fiftyfold; there is strong evidence that it is now in decline (Watson and Pauly 2001). Recent decades have seen optimism progressively replaced with disquiet, then alarm. Just a few decades ago, the world's fisheries were regarded as a challenge for development: resources were just waiting to be discovered, and fished to generate prosperity. Even until the 1980s, governments provided ship building bounties and other subsidies to promote the development of resources in their newly declared Exclusive Economic

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[1] An earlier, extended version of this section was published as Buckworth (1998b)

Zones. At the same time, the theory of fishery assessment and management was also being developed, to ensure that benefits were maximised sustainably. But, over recent decades, that optimism about the extent of resources, and our faith in our ability to effectively assess and manage fisheries, have been progressively eroded.

Gathering disquiet at the demise of major fisheries (eg California sardine, North Sea herring, *Peruvian anchoveta*) through the mid-century built, by the mid-1990s, to outright alarm at continued collapses (eg northwest Atlantic cod and ground fish stocks) and, as evidence mounted, the realization that overfishing was a widespread phenomenon. While fishing was approaching theoretical limits, the potential for harvest was already eroded: globally, fisheries were and remain in a crisis state (eg McGoodwin 1990; Garcia and Newton 1997; Botsford *et al.* 1997; Buckworth 1998b; Pauly 1996). Global food security is threatened (McGoodwin 1990; Pauly 1996; Garcia and Newton 1997; Botsford *et al.* 1997; Buckworth 1998b; Pauly *et al.* 2002).

The world's fisheries are teetering on, and may have exceeded, the limits imposed by oceanic production. Many are over-exploited, with attendant social, economic and ecological disruption. We are at a point in our history where fisheries management must therefore perform at its best, yet our record of fisheries management is poor, despite the increasing sophistication of analytic, assessment and management tools. We need to improve, to rethink, our approaches to fisheries assessment and management.

World fishery production increased exponentially between the 1950s and the 1990s. A 300% harvest increase was achieved, so that at around 100 million tonnes, the fishery supports around 200 million people and represents considerable world trade (Garcia and Newton 1997). The catch cannot be limitless. However, the actual total catch is inexactly known. Over-reporting by

China – where state edicts that catches should increase drove recorded statistics, rather than real catch information – has meant that the increasing trend of world catches of the last decade has probably been spurious (Watson and Pauly 2001). By-catch, discarded, is estimated at an additional 27 million tonnes (Alverson and Larkin 1994) while further undeclared catches might be as high as 20-30% of those recorded (Pitcher 1995). The geographic expansion of fisheries, generating the great increase in catches to the 1970s (Alverson and Larkin 1994; Myers and Worm 2003) cannot be repeated. Any small growth achieved by expanding into under-used resources is likely to be offset by losses to overfishing elsewhere.

Estimates of sustainable world fishery production were critically reviewed by Pauly (1996). The only credible estimates were of 100 million t (Gulland 1970) and 120-150 million t (Moiseev 1994). It is possible then, that fishing has, already, pushed beyond the limits that would provide maximum sustainable yield globally. The proportion of aquatic primary production captured by fisheries, 8%, is deceptively small: most primary production is in the extensive but relatively unproductive open ocean from which fisheries appropriate just 2% of primary production (Pauly and Christensen 1995). Meanwhile, fisheries appropriate 24-35% of primary production from the freshwater, continental shelf and upwelling systems, the very areas most subject to environmental damage and reduction in ecosystem diversity. Garcia and Newton (1997) noted that the total world effort in 1989 was near their (roughly) estimated $F_{0.1}$ level and cautioned that such a global assessment disguises the status of individual stocks.

Only about a third of the stocks examined by FAO in 1994 (Garcia and Newton 1997) were less than fully-exploited, while 22% were over-exploited or depleted. The situation has continued to deteriorate: a mere 25% of stocks were considered to be lightly or moderately exploited in

2000, with 47% fully-exploited and 28% over-exploited, depleted or recovering from depletion (Anon 2002).

The details and ramifications of the crisis continue to be exposed. Garcia and Newton (1997) observed a change to lower value species in catches as higher value species (eg cod) became depleted. This was evidenced in ecological, in addition to economic terms, by a substantial reduction in trophic level of fish catches (Pauly *et al.* 1998; Pauly *et al.* 2001), with landings shifting, over the period 1950 to 1994, from large piscivorous fishes toward smaller invertebrates and planktivorous fishes. This decline in trophic level has been particularly marked in the Northern Hemisphere (Pauly *et al.* 1998); the biomass of North Atlantic predatory fishes has apparently been reduced by two-thirds in the same period, and by a factor of nine over a century (Christensen *et al.* 2003). Myers and Worm (2003) have demonstrated that, on a global scale, predatory fish biomass is around 10% of pre-industrial levels – a pronounced decline across whole communities, in a wide range of ecosystems. Most of the declines in biomass happened rapidly from the inception of fisheries, in as little as 3-5 years in seamount and continental slope fisheries. At an ecosystem-wide scale, removal of most top predators, even the extinction of local populations, may engender depensatory effects (Walters and Kitchell 2001) and may have correspondingly widespread, and unpredictable ecological consequences. Strong top down effects are already apparent in some systems (Daskalov 2002; Myers and Worm 2003).

Paradoxically, the cost of fishing exceeds the revenue from harvest. This deficit reflects over-capitalisation and over-capacity and effectively, is offset by complex systems of government subsidies (Sissenwine and Rosenberg 1993). Where can this over-capacity go? This is one of the critical factors driving the overfishing problem (Mace 1997).

Economically, it is more efficient to fish down low productivity stocks, even over-fished stocks. Capacity is redeployed when alternative opportunities arise (Clark 1990; May 1994). This means that entrenched overcapitalization and politico-social effects will continue to drive fisheries even when the resource on which they depend has collapsed (eg Peruvian anchoveta, Muck 1989). At the same time, over-capacity enables rapid redeployment, so that fisheries growth is virtually instantaneous, threatening the continued viability of fished stocks (Mace 1997).

However, there are other factors at play. There is a food fish deficit that with population growth and changing world demographics will drive demand (Alverson and Larkin 1994; Garcia and Newton 1997; Anon 2002). This will create pressure to develop new fisheries and increase and intensify those fisheries that are not yet over-fished. In addition, wherever there is economic opportunity, it will create pressure for the shift of capacity from developed to developing nations (Bundy and Pitcher unpublished; Garcia and Newton 1997). The risk is extraordinary for countries that do not have the capacity of the US or Canada to stand the social and economic consequences of a large fishery collapse.

1.1.1 What can our responses be?

There are some patent responses to the world fisheries problem – precautionary approaches with better stock assessments based on better science and better data, implemented with better understanding of the non-biological side of fisheries management. These responses are mostly now encoded within the Codes of Conduct for Responsible Fisheries (FAO 1996) and varying government policies and legislation. However, can we be optimistic? Given the past record, we cannot – we must try to improve on all fronts.

There have been strong advances in technology and statistical methodology available for fisheries assessment, technology for collecting, accessing and managing data, and our understanding of basic population processes is improving (Richards and Megrey 1994). Schnute and Richards (1994) speculated on the evolution of the fisheries statistics required to meet future stock assessment needs, and emphasised the need for new data sources. New developments in ecological modelling have opened the way for better understanding of ecological processes and assessment of alternative policies (eg Walters *et al.* 1997; Pauly *et al.* 2000; Walters and Kitchell 2001; Pauly *et al.* 2002). There are even new monitoring approaches based on trophic level assessment (Pauly *et al.* 2001).

The need to examine uncertainty, its recognition and quantification, was emphasized by several workers in the previous decade (such as Hilborn and Walters 1992; Rosenberg and Restrepo 1994). The evaluation and presentation of such uncertainty in management strategy evaluations is becoming common practice (eg Francis and Shotton 1997; Sainsbury *et al.* 2000). There can be little argument that better information subjected to better analysis and used more carefully in management will be to advantage. Nevertheless, there are many fisheries for which the management and scientific infrastructure does not exist. For many fisheries, revenues and sampling power might not support a large investment in research, monitoring or compliance. There is a need for new monitoring and management tools that are suitable for a wide range of fishery contexts.

It must be a warning that the most notable and tragic of fishery collapses have occurred while there was maximal scientific and managerial input. The Peruvian anchoveta fishery collapsed from more than 12 million t in 1972 (Boerema and Gulland 1973), a final catastrophe exposing a need to replace the Maximum Sustainable Yield paradigm (Larkin 1977). The production of

both North Sea herring and North Atlantic cod dwindled from several hundred thousand tonnes to commercial extinction. This was despite quantum leaps in the sophistication of fishery models, data collected and management invested (Saetersdal 1980; Hilborn and Walters 1992; Hutchings and Myers 1994; Walters and Maguire 1996). These historic examples represent the full range of failures due to assumptions about the adequacy of data, experience or methodology. Sometimes the problems lie with motivations: economic and politico-social forces meant that whales were knowingly over-fished (Allen 1980; Clark *et al.* 1985). Clearly assumptions on which assessments and management strategies are based need to be continuously examined and upgraded, just as much as methods need to be improved.

There is an important quandary in developing fisheries, even when there is determination to invest whatever might be needed in science for sustainability. The most productive population level for sustaining catches from a fished stock can really only be ascertained by observing the responses when a stock is fished below that level (Hilborn and Sibert 1988; Ludwig *et al.* 1993). Critical changes can happen in fisheries at rates to which management and social structures cannot respond. A typical scenario is thus that fisheries have to be cut back when the participants are already suffering reduced revenues from declining catches (Hilborn and Walters 1992). Uncertainty in assessments provides fuel for argument against reducing the catch, and risk-prone fisheries management decisions thus follow, engendering further cycles of declining catches and revenues and exacerbation of economic, biological and social consequences (Sissenwine and Rosenberg 1993).

This inability to predict, assess and control fishery impacts has prompted the growing emphasis on precautionary systems (eg FAO 1996; Restrepo and Powers 1999; Chesson *et al.* 1999; Hilborn *et al.* 2001) and recognition and management of fisheries as integral components of

dynamic ecosystems. This implies approaches, which rebuild fish populations within large

marine protected areas (MPAs) (Pauly *et al.* 1998; Pauly *et al.* 2002). In concurrence with

observations on fisheries likely to remain sustainable (Walters 1998b), MPAs impose an upper

limit to exploitation rates, at the same time providing a hedge against ignorance of ecological

interactions (Lauck *et al.* 1998). Some authors have even advocated reversal of the usual

paradigm, so that areas open to fishing are the exception rather than the rule. This ensures that

only a small fraction of fish stocks and the ecosystems in which they are components are

exposed to fishing (Walters 1998a).

The pressure on fishery resources to provide food and socio-economic benefit means that new

fisheries will develop, and existing fisheries will be under pressure to expand. Each additional

fishery management failure will compound the global fishery crisis. How, then, should these

new and developing fisheries be managed to avoid the catastrophes that have marred the history

of fisheries management? Better information gathering (Martell and Walters 2001; Walters and

Martell 2002) and better stock assessments are needed, as is their better application in

management systems that are robust and reversible to failure of basic assumptions about stock

characteristics and fishery system responses.

1.2. Spatial structure and dynamics of stocks affect parameter estimation and management

Fishery management is a problem in controlling processes at a mixture of scales of spatial,

temporal and ecological organization. Fishery biologists and managers realize that fishery

systems are heterogeneous and patchy across these scales. The management process, to be

successful, needs to somehow accommodate effects that may occur at and interact between

different scales (Mason and Brandt 1999). The general solution to this problem has been to identify, assess and manage "unit" stocks. These are basically, arbitrary groups that are self-reproducing, closed to migration, and where the members of each group have similar life history characteristics such as recruitment and mortality attributes (Hilborn and Walters 1992; Schneider *et al.* 1999). Consequently, there is a substantial research effort dedicated to spatial stock definition and identification (Begg *et al.* 1999) This is not a simple endeavour. Especially in maritime situations, it is rare that there is discrete stock delineation – structure is complicated by migration and mixing, temporal variation in stock locations, political and social considerations that may vary from the stock spatial structure, and so on (Stephenson 1999): from the conceptual level through to the applied, spatial effects not typically included in assumptions about structure lead to bias in stock assessments, and complicate and constrain opportunities for safe regulation of a fishery.

At the scale of stocks and fisheries, management of multiple stocks or substocks as a single discrete stock has several well-known effects (eg Ricker 1973). There are problems for any mixed stock fishery with spatially varying population characteristics (Collie *et al.* 1990). For multiple stocks with differing stock productivities, treated as a single stock, stock recruitment analyses will typically over-estimate the optimum harvest rate and under-estimate optimum escapement levels (Hilborn 1985). Growth may show substantial variation over the distribution of a management unit (Campana *et al.* 1995), so that again the productivity of some stocks or substocks may be over-estimated. The consequence is that less-productive stocks tend to be overfished (Ricker 1973; Larkin 1977; Supriatna and Possingham 1998; Cadrin and Friedland 1999; Clark 1999; Holland and Maguire 2003), or even extirpated (Stephenson 1999).

At within-stock scales, spatial dynamic effects have undermined assessment and management of important fisheries. A number of spatial effects at such a scale may have contributed to erroneous assessments of the Canadian Northern cod stock. These include shifts in spatial distribution that exposed the stock to greater fishing pressure (Rose 1993) and caused misleading survey estimates (Hutchings 1996). Migration can create bias in survey abundance estimates (McAllister 1998), one effect being to create false impressions of large natural variability in abundance (Samb and Pauly 2000). A consequence of meta-population structures may be that even without reduction of genetic diversity, localized overfishing may erode spatial stock structures through loss of opportunities for recruiting fish to learn spatial components of population dynamics, such as migration routes or spawning sites (McQuinn 1997; Secor 1999). This may delay recovery of depleted populations (Smedbol and Wroblewski 2002). Spatial variation in age composition may lead to loss of more productive elements of the spawning stock (Cardinale and Arrhenius 2000).

There are also strong effects on smaller spatial scales. For pelagic species, aggregating or shoaling behaviour can produce the effect that densities at local scales remain high while the area occupied by the stock, and the stock as a whole, declines (Paloheimo and Dickie 1990). The consequence of this is that search efficiency by fishers increases, and produces one of the more dangerous spatial dynamic effects that managers need to contend with – escalating catchability (fishing mortality rate per unit fishing effort) as a fish stock declines (Paloheimo and Dickie 1964). This is against a background of industry observation that catch rates remain high. Rose and Kulka (1999) describe this effect for Canada's northern cod fishery. Stock assessment methods may also be compromised in these situations, misinterpreting the decline in the proportion of older age classes as a relative increase in younger fish, i.e. as increased recruitment when in fact the recruits are just being harvested more severely. These problems

are exacerbated if the analyses rely upon CPUE (National Research Council 1998), or surveys which have not adequately accounted for changes to the stock area (Walters and Maguire 1996).

The hyperaggregation of pelagics is an extreme of a process that occurs in all fish stocks: fish densities are spatially heterogeneous, as a result of aggregation to various habitat features (Essington and Kitchell 1999; Mackinson 1999). Availability or reactivity to fishing gear may change rapidly (hours-weeks) in time (Clark and Mangel 1979; Walters and Bonfil 1999; Cox 2000), or even due to random processes, at scales of metres, through to the spatial scales of shoals or aggregations and grounds, to that of the area occupied by the stock, over temporal scales of hours to seasons. These are also the temporal and spatial scales of fishing operations. Fishers do not usually fish randomly, but in response to experience (locations that provide better catches at certain seasons, tides, weather conditions etc), and the likely value of catches. They thus respond to the fine scale distribution of fish densities (~ relative catch rates) and such socio-economic factors as distances from ports (Hilborn and Ledbetter 1979; Hilborn 1985; Allen and McGlade 1986; Hilborn and Kennedy 1992). Thus it is largely fisher behaviour that drives catch rates (Gordon 1953; Hilborn and Kennedy 1992).

This non-random distribution of effort potentially degrades many of the predictions made by assessment and management models that do not represent the detailed dynamics at similar scales. Thus Caddy (1975) warned that, for shellfish populations, peak mortality occurs progressively earlier in fishery development with the degree of spatial aggregation. This was despite predictions of the gear selection ogive, emphasising that selection was a property of the interactions between fisher behaviour, gear, and the target species, rather than a property of the gear. He also established that, when MSY was exceeded, yields would decline much more rapidly than predicted under a simple yield per recruit analysis. Increasing capacity, or fishing

power, that acts over a scale of years, may be confounded with the effects of non-random

distribution of fishing, acting on a time scale of days to months. Thus, probable overfishing on

tiger prawns in Australia's Northern Prawn Fishery was recognized in 1986, with steadily

increasing fishing power and lack of standardization of effort being recognized as a contributing

assessment and management problem (Buckworth 1987). A continuing management response

has been to track fishing power increases (Robbins *et al.* 1998) and to remove capacity with the

goal of containing total effort to below that predicted to achieve an MSY goal. At the same

time, search in the fishery is highly non-random, and most of the annual catch is taken (and

most of the fishing mortality imposed) within the first few weeks of each season. Management

of the fishery must certainly deal with increasing technical efficiency. However the main effect

of effort reduction by removal of capacity has most likely been the removal of the (late season)

component of the annual fishing effort. This component imposes the least incremental fishing

mortality, and least impacts on stock reproductive capacity. The tiger prawn component of the

fishery is, probably, still overfished (Dichmont *et al.* 2001). The ability of fisheries to switch

from species to species and location to location in response to changing abundance and values

has driven the serial depletion of Alaskan crustacean resources (Orensanz *et al.* 1998).

The use of catch per unit effort as a measure of relative abundance in stock assessment has

attracted strong criticism. Obviously, unless fishing is randomly distributed with respect to the

stock distribution, CPUE must be a function of both fisher targeting behaviour and the

distribution of abundance. Warnings about this problem have been in the fishery literature for a

long time. Paloheimo and Dickie (1964) suggested that nonrandom fishing and correlation of

biomass and range could alter catchability, with their predictions subsequently confirmed for the

Georges Bank haddock stock by Crecco and Overholtz (1990). Rothschild (1972) suggested

that catchability could be variable, while Mangel and Clark (1983) demonstrated that

distribution of densities will largely determine the form of the relationship between catch rates and abundance. Several studies in diverse fisheries (eg Rose and Leggett 1991, Walters and Maguire 1996, Atlantic cod; Wallace *et al.* 1998, western rock lobster) have supported these observations. In reviewing the relationship between CPUE and abundance, Hilborn and Walters (1992, Chap 5) point out that the "spatial pattern of abundance, the spatial pattern of fishing effort, and the relationship between abundance and capture all combine to produce the aggregate CPUE". Hyperstability, in which catch rates decline at a slower rate than abundance, is a very common pathology. Nevertheless, for many fisheries, the only long term information on fished stocks are time series of catch and effort data (eg Punt *et al.* 2000; Dichmont *et al.* 2001).

Although spatial structuring is, therefore, problematic for fishery assessment and management, there can be generic advantages, also, since the structure can provide opportunities for innovative management regimes (Walters 1986; Walters 1995). Protected areas are a conceptually simple example, typically protecting part of a spawning stock or nursery area, containing fishing mortality, and providing resilience to overfishing and to unforeseen events (Lauck *et al.* 1998). These attributes, coupled with the potential for concurrent protection of system biodiversity, have generated an enormous amount of investigation (see review by Guénette *et al.* 1998). Temporal closures may provide a refuge from fishing for new recruits, to enhance yield, value or survival to spawning (eg Die and Watson 1992), or to protect other seasonally vulnerable segments of the population (eg spawning aggregations). Although not a panacea, incorporation of protected areas in management strategies can impose an upper limit to exploitation, acting as a control in a fixed harvest rate strategy. Such strategies are robust to changes in carrying capacity (Hilborn *et al.* 1995; Walters and Parma 1996), and the closure would guarantee the safety of at least a portion of the population.

Rotating harvest management strategies for sedentary species can help contain fishing mortality (and even indirect fishing mortality on non-target species) by alternately opening and closing areas within a fishery (for example, Botsford *et al.* 1995; Myers *et al.* 2000). For example, complex schedules of temporal and area closures have been extensively used to maximize the economic yield of penaeid prawn stocks in Australia (eg Carrick 1982; Somers 1985). These species grow rapidly as they undergo offshore migrations, and as value per unit weight also increases with size, the benefit of making use of the temporal and spatial separation of sizes is transparent.

Recognition of spatial structuring in a population also enables assessments that are more effective. Spatial replication makes better use of available information than is possible if the observations are combined, assuming that information on some population parameters is shared between substocks (Collie and Walters 1991). Polovina (1989) used a short time series of data for slipper lobsters from three isolated areas to fit a spatially structured biomass dynamic model, demonstrating that fishing effort levels were unsustainably high. Fletcher (1992) used a spatial model, again to examine a short time series for a developing fishery, providing a more precise estimate of possible stock biomass and recruitment. Moreover, he was able to predict scenarios that could be addressed experimentally by relatively inexpensive egg survey methods. Adaptive management (eg Walters 1986; Hilborn and Sibert 1988; Walters and Collie 1989; Collie and Walters 1991) carries this approach further: by deliberately maximising the spatial contrast in exploitation rates on a fished stock, the dynamics of the stock can be quantified more accurately.

Managers clearly face many problems, and some opportunities, arising from the spatial dynamic problems described above. Detailed knowledge of fishery dynamics (ie of fishers and of the fish) might be available for some fisheries, but for developing fisheries and fisheries that

generate low revenues, this is not likely. Spatial complexity is clearly one of the reasons that fishery management is often subject to difficult choices. The basic problems of choosing monitoring methods and instituting suitable management arrangements, invariably under uncertainty, remain.

1.3. Thesis overview

In this thesis , I focus on aspects of the fundamental questions with which I began this chapter. Given that spatial dynamic effects might devalue information and management approaches, I ask how different monitoring and management control combinations might perform under a suite of differing spatial stock dynamic and administrative structures. As a case study, I address how the results of that examination might be applied to the fishery for narrow-barred Spanish mackerel, *Scomberomorus commerson*, (Lacepède 1800, Family Scombridae) in Australia's Northern Territory (NT). This examination leads to the development of a methodology that could have broad application.

In Chapter 2, I present the machinery to examine management and monitoring of spatially structured fisheries, including an improved application of the disc equation for the distribution of fishing effort. In Chapter 3, I use simulations to examine which monitoring/ management control combinations might be likely to be effective for fisheries like the Spanish mackerel fisheries of northern Australia, given differing degrees of fishery capacity and non-linearity in catchability. This complexity is compounded by various spatial stock and administrative structures. Several interesting generalizations arise from this part of the study. Most important is the utility of mark-recapture based direct monitoring of fishing mortality rates and catchability over the collection of catch, effort, and survey information that is more commonly used in stock assessment. Also of interest is the observation that multiple spatial management

units may perform better than single management units - even when these administrative structures differ spatially to stock structure.

I focus on the Northern Territory fishery for Spanish mackerel in Chapter 4. I conclude that the status of the fishery is very uncertain, and that a good means of monitoring fishing mortality rates or abundance is required to overcome substantial uncertainty in its status. The least informative of monitoring methods examined in Chapter 4 (catch rates and mean weight) are frequently the only ones available for this species: the size, mobility and pelagic habit of *S. commerson* make them difficult to survey. They are also difficult and dangerous yet fragile to handle, so that tagging programs that might be used to estimate harvest rates or abundance are very difficult and expensive.

To overcome the problems associated with tagging, I suggest that genetic tagging be developed as a monitoring methodology for *S. commerson*. In Chapter 5, I describe the development of a simple device for *in situ* collection of genetic samples. Given the demonstrated feasibility of the components of genetic tagging for Spanish mackerel, Chapter 6 is a simulation of this and alternative monitoring methods that might be applied in the Northern Territory fishery. It concludes that genetic tagging is a viable option for monitoring NT Spanish mackerel. In the conclusion, Chapter 7, I argue that for many fisheries, collection of monitoring information for stock assessment might be replaced by direct monitoring of fishing mortality rates and catchability. For this purpose, genetic tagging might find broad application by overcoming many of the problems associated with conventional tagging.

2. A SIMULATION FRAMEWORK FOR EVALUATING MONITORING/ MANAGEMENT CONTROL SETS FOR SPATIALLY COMPLEX FISHERIES

2.1. Introduction and model overview

In the introductory chapter, I indicated the need to address basic fisheries management questions that acknowledge spatial dynamics of fisheries. The purpose of this chapter is to describe a modelling system built to address such questions; results of applying this system will be described in Chapter 3. In particular, the system allows for the investigation of the effectiveness and attributes of different monitoring and management approaches when the spatial stock structure of fisheries is poorly known, or when political or administrative arrangements (such as provincial/ state or regional boundaries) dictate the spatial structure of management. It also provides for the examination of effects of fishery dynamics that are manifest on finer temporal and spatial scales than most fisheries can be managed. It is rare that there will be detailed knowledge of spatial dynamics from the outset of a developing fishery, so that choices about management regimes should be sustainable and robust to a wide range of potential dynamic structures.

The aim of this section of the work was to develop a closed-loop simulation framework for considering the effects of spatial complexity with different monitoring/ management arrangements on fishery performance (Smith 1993). In this chapter I describe this spatial framework, which includes a reference population dynamics model, spatial fishing effort
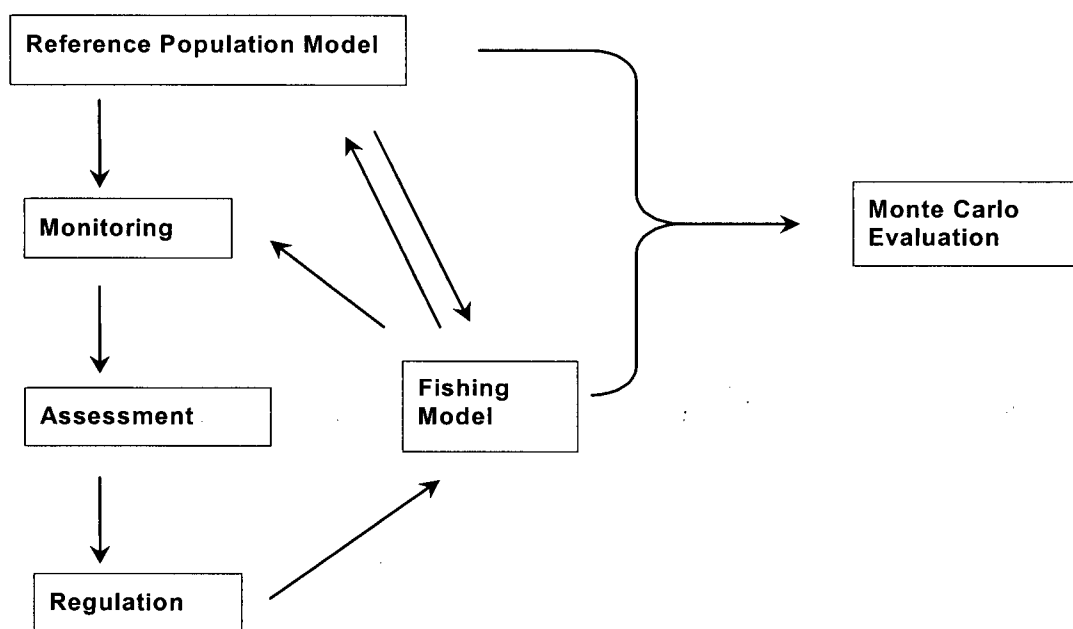
Figure 2.1. Schematic representation of model structure for evaluation of monitoring/management control combinations for spatially complex fisheries.

dynamics, management information gathering, simulated use of the information for assessment, and management control (Figure 2.1). Monte Carlo simulations can be used to compare various scenarios – combinations of different spatial population structures, recruitment variation, sampling levels in alternate sets of monitoring measures (catch and effort information, surveys, length measuring, tagging), assessment approaches and management controls. As part of the simulation of the uncertainty faced with spatial stock structure, performance can be compared between scenarios that vary in the extent that their spatial structures agree with those specified in assessment and management arrangements. A number of performance measures are used to make comparisons.

To capture realistic dynamics of spatial fishing effort, the interactions between search and stock abundance are simulated using an approach based on the disc equation (Holling 1959; Hilborn and Walters 1987, 1992). This equation is used to relate catchability (the proportion of the fish

stock removed by a unit of effort) to local abundances. The predicted distribution of effort responds to changes in the fished population at fine time and spatial scales, and to management rules (Hilborn and Walters 1987). I elaborate an efficient methodology for doing this.

Simple monitoring and management approaches are implemented. Several criteria are used to measure performance of different types of information/ management control combinations (McAllister and Kirkwood 1998b), including relative to an optimal catch series (Walters and Parma 1996; Walters 1998b, Martell and Walters 2001).

The simulation system provides for the evaluation of different information gathering/management control approaches within any spatial scenario and so is the basis of comparison among sets of scenarios. Specification of the model with realistic parameter inputs would provide for management strategy evaluation (eg Polacheck *et al.* 1999) given various hypotheses on spatial structure of stocks.

The model as parameterised here roughly approximates the fisheries for narrow-barred Spanish mackerel, *Scomberomorus commerson*, in northern Australia and is in the first instance a policy tool for these and similar fisheries. Nevertheless, with alternative specification requiring minimal input, the model could be applied to a broad range of fisheries and species.

2.2. Spatial fishery model

2.2.1 Reference population

For the reference population simulation indicated in Figure 2.1, I used the FISHMOD system, a VisualBASIC shell developed by Professor C. J. Walters (Fisheries Centre, University of British

Columbia). This system provided a spatially-referenced population model, in a framework that allowed convenient development of the other model components shown in Figure 2.1, which I developed. These were the fishing, monitoring, assessment and regulation sub-models and the Monte Carlo evaluation system. FISHMOD's spatial referencing enabled the consideration of various population and management spatial structures. My objective in this section is to describe the pertinent features of that system.

Based on standard, single species age-structured accounting (Hilborn and Walters 1992), the FISHMOD system explicitly models dispersal and migration across a series of spatial cells. This set of spatial cells represent a map of the area over which fish populations and fishing are distributed. The system accommodates multiple substocks, for which there is separate age structure accounting. The substocks are identified by specifying groups of cells as nursery areas, to which adults return for spawning. Abundance and age structure information for each spatial cell is represented in each monthly time step, providing for allocation of fishing mortality by substock and age. For simplicity of interpretation, age specific weights, relative vulnerabilities and fecundities are treated as identical between substocks. Attributes with spatial components, such as habitat suitability, fishing costs, closures and management/administrative boundaries, that have relevance to population dynamics or management, are all referenced at each spatial cell.

Population dynamics

In the absence of movement, the number of fish $N_{s,a,ij,t}$ of substock s and age a in cell ij at time t is dynamically represented by

$$N_{s,a+1,ij,t+1} = N_{s,a,ij,t} \cdot \exp(-v_a F_{ij,t} - M),$$

with M being the natural mortality rate and v_a the relative vulnerability at age a. The fishing mortality rate $F_{ij,t}$ represents the annual sum over the monthly fishing rates generated by effort allocated to the cell ij, which sum to the annual fishing effort $E_{ij,t}$ (prediction of fishing mortality from effort is described below). Harvest and movement calculations are calculated monthly but, to save computation, natural mortality is imposed only annually, at the end of each 12-month sequence.

Movements

Movements over spatial cells by substock and age are described by a sub-model in which, for each simulated month, fish can be redistributed among neighbouring spatial cells (providing that cell has suitable habitat). For each such step, the fish $N_{s,a,ij,t}$ are divided into five proportions p: p_0 is the proportion that remain in the cell over the step, and p_1 to p_4 are the proportions moving to adjacent cells such that p_1 is the proportion that move to cell $i,j-1$, and $p_2, p_3,$ and $p_4,$ are the proportions that move to cells $i+1,j$, $i,j+1$, and $i-1,j$, respectively, if habitat is suitable in those destination cells.

Then $N_{s,a,ij,t}$ is recalculated for movement as the number of fish remaining, $(p_0 N_{s,a,ij,t})$ plus movements into cell ij from the (up to) four adjacent cells. These are the sum of proportions times numbers $p_k N_{s,a,i'j',t}$ moving, from the surrounding $i'j'$ cells. The p values are calculated by assigning a movement "weight" W_k to each of the five possible moves k.

Setting a high weight W_0 on not moving represents low dispersal rate, even weights $W_k,$ $k>0$ represent random dispersal to adjacent cells, and biased weights W_k on particular k values represent increased likelihood of "migrating" toward the cell represented by k. The proportions moving, p_k, are calculated as $p_k = W_k / \Sigma W_k$.

To represent migration, the FISHMOD user can specify stock-age specific "target cells" $ij_{a,s,m}$ by month of the year m; then at each movement step, the model places proportionally higher weights w_k on moving into those cells that are in the spatial direction of $ij_{a,s,m}$, with increasing distance of ij from $ij_{a,s,m}$. This weighting approach causes animals in cells ij "near" $ij_{a,s,m}$ to disperse randomly, but to cause much higher proportions of animals in more distant cells to move toward $ij_{a,s,m}$. The user can also specify age dependency in the non-movement weight W_0, to represent limitations in movement speed (proportion of fish able to leave each cell per movement step) of smaller fish.

Extensive movement may be simulated by running the sub-model more than once per month i.e. with repeated dispersal-migration computation steps.

Specification of population parameters

The annual sequence of population and fishery events in FISHMOD is summarized in Figure 2.2. The reference population is specified by entry of the values in Table 2.1 and Table 2.2. Multiple entries for the reduction in recruitment compensation parameter, r_s, in Table 2.1, indicate values for different substocks, while multiple values for the standard deviation in recruitment, u_r, specify alternative states of nature represented in the simulations (details in respective sections below). Although the unfished population biomass was chosen to roughly approximate the NT *S. commerson* population, it could conceivably be in error by a substantial amount as this value is uncertain. For the purpose of this part of the study, the value input is only to provide scaling that is reasonably realistic. The survival rate was derived in a procedure analogous to Hoenig (1983), by predicting that, in the absence of fishing, 1% of animals would survive to 12 years. The greatest age of *S. commerson* observed in catch samples from the NT has been 11 years of age (Buckworth 1998a). Older animals have been observed in both the

Queensland (McPherson 1992) and Western Australia (Mackie *et al.* 2003) so this value is quite uncertain. The recruitment compensation parameter (see below), is a moderate choice from values inferred from the metanalysis of Myers *et al.* (1999).

Each entry in the weight at age schedule, (Table 2.2) was derived from the growth equations and length-weight relationships for Queensland Spanish mackerel, (McPherson 1992), by generating a series of weights at age for each sex and then finding the mean of the two. Growth observed in the Northern Territory (NT) fishery is consistent with the growth model parameters described by McPherson (1992) but is quite variable between individuals (Buckworth 1998a).

```
┌─────────────────────────────┐          ┌─────────────────────────────┐
│  Population events          │          │  Fishery &                  │
│                             │          │  management events          │
└─────────────────────────────┘          └─────────────────────────────┘

    ┌─────────────────────────┐              ┌─────────────────────────────┐
    │ Natural mortality       │              │ Pre-season survey           │
    │ Growth                  │─────────────▶│ Update assessment           │
    │ Recruitment             │              │ model (data from t if survey,│
    │                         │              │ otherwise t-1)              │
    └─────────────────────────┘              │ Calculate control targets   │
                                             └─────────────────────────────┘
```

Monthly events

Update $N_{s,a,i}$ for movement	Calculate regulatory controls
	Distribute fishing subject to controls
Update $N_{s,a,i}$ for harvest ◀──	Record catch and effort

Tag, weight and CPUE monitoring information	Storage of performance information

Annual Loop*

Performance measures
Calculation and storage after I*
iterations
Storage of performance

Figure 2.2. The annual sequence of events in the closed loop simulation. The number of annual loops* and iterations for each fishery scenario I* are inputs.

Knife edge age selection was specified by setting relative vulnerability at age (the relative exposure to fishing mortality) to 0 for fish ages 1 to 4, and 1.0 for older fish, to ensure that the specifications of the reference population model and the assessment model were as close as possible (the delay-difference age-structured model used for assessment required knife-edge recruitment). In the fishery for *S. commerson*, fish are not fully selected until ages 4-5 (Buckworth 1998a).

Fecundities at age (Table 2.2) were derived from Mackie *et al.* (2003) and are the product of observed proportions mature at age and batch fecundities at age, and have been scaled by a factor of 10^{-8}. It is not known for *S. commerson* whether there is an age/size related difference in the extent of batch spawning, and whether this may differ with latitude or other locational components. The extent of spawning appears to differ with latitude (McPherson 1993; Mackie *et al.* 2003).

The movement weight specified in Table 2.2 specifies age dependency in the non-movement weight W_0, as described above. The larger values for older fish represent their greater capacity for movement relative to younger, smaller fish. As little is known of the movement rates of Spanish mackerel, the values entered represent no more than an intuitive set of values.

Table 2.1. Input values for population parameters for the spatial reference model

Parameter	Input values	Description
B_0	5000	Unfished total population biomass
ζ	0.68	Natural survival rate (exp(-M), where M is instantaneous natural mortality rate)
K_r	6	Recruitment compensation: increase in survival of recruits at very low population levels
r_s	0.8, 0.6, 1.0	Reduction in K_r for substocks $s = 2, 3, 4$
u_r	0.2, 0.6	standard deviation in recruitment

Table 2.2. Values for age-specific parameters for the population reference model

Age	Weight	Relative vulnerability	Fecundity	Movement weight
1	2.16	0	0.005	0.0
2	3.87	0	0.052	0.1
3	5.77	0	0.148	0.3
4	7.71	1	0.255	1.0
5	9.57	1	0.353	1.0
6	11.30	1	0.456	1.0
7	12.86	1	0.559	1.0
8	14.24	1	0.660	1.0
9	15.45	1	0.754	1.0
10	16.50	1	0.842	1.0
11	17.40	1	0.921	1.0
12+	18.17	1	0.993	1.0

Recruitment

FISHMOD generates recruitment annually with the Beverton-Holt (Beverton and Holt 1957) stock-recruitment relationship (S-RR), in a form described by Hilborn and Walters (1992),

$$R = a'E /[1 + (a'/b'')E \qquad \text{Eq 2.1}$$

where R specifies recruitment, E is the total egg production, a' is the maximum R/E for low stock size and b'' is the maximum number of recruits when E is very large. The information for specification of the curve was entered as the unfished population biomass for the substock and the relative recruitment compensation at low stock sizes, K_r inferred from the analyses of Myers *et al.* 1999. This required calculation of the parameters a' and b'' using the approach of Walters and Bonfil (1999):

$$a' = K_r R_0 / E_0, \hspace{2cm} \text{Eq 2.2}$$

with R_0 being the recruitment, and E_0 being the egg production, at unfished equilibrium. Substituting for Eq 2.2 and the initial equilibrium values allows translation of the expression a'/b'' into terms that are readily calculable from entered values (or may be leading parameters in an estimation), so that Eq 2.1 can be re-expressed:

$$R = \frac{K_r (R_0 / E_0) E}{1 + [(K_r - 1)/E_0]E} \hspace{1cm} \text{Eq 2.3}$$

Elaboration of the S-RR with Eq 2.3 thus involved prior computation of several values at the unfished equilibrium. These are conveniently expressed using the analytic approach for equilibrium age-structured fisheries of Botsford (1981). Given the values for initial population biomass, B_0, natural survival ζ ($= exp\ (-M)$), (Table 2.1) and the fecundity and weight at age schedules (Table 2.2), the following are predictions for an unfished population at equilibrium:

$$l_{a0} = \varsigma^{a-1} \hspace{4cm} \text{unfished survivorship to age } a,\ a_r < a < a_{max}$$

$$l_{a\ 0} = \varsigma^{a_{max}-1} /(1-\varsigma) \hspace{2.5cm} \text{unfished survivorship to age } a = a_{max}$$

$$\varphi_{e0} = \sum_a l_{a0} f_a \qquad\qquad \text{unfished eggs per recruit}$$

$$\varphi_{b0} = \sum_a l_{a0} w_a \qquad\qquad \text{unfished biomass per recruit}$$

$$R_0 = B_0 / \varphi_{b0} \qquad\qquad \text{unfished recruitment}$$

where a_r is age at recruitment, w_a is the weight at age a. Note that the age $a = a_{max}$ is the plus-group, all fish of age a_{max} or older. The proportion of the recruits allocated to substock s, r_{ps}, was defined as the proportion of nursery cells for s out of the total nursery cells used by the population.

Annual total egg production for the stock s was defined by the numbers at each age a (those remaining after the 12 month sequence of fishing and natural mortality) multiplied by fecundity at age, f_a, summed across all cells i used by that stock, ie

$$E_s = \sum_i \sum_a f_a N_{s,a,i}$$

Additionally, in some scenarios, I altered FISHMOD so that the relative recruitment compensation differed between stocks; this required that an additional variable, r_s, representing differences in recruitment curve slopes between stocks, relative to the substock 1, was also entered (Table 2.1). The number of year 1 fish entering substock s was calculated as

$$N_{s,1} = \frac{r_s K_r (R_{0s} / E_{0s}) E_s}{1 + [(K_r - 1) / E_{0s}] E_s} \exp[u_r (\varepsilon - u_r / 2)] \qquad \text{Eq 2.4}$$

where u_r sets the magnitude of simulated recruitment variation and ε is a random normal variable of mean = 0 and variance = 1. Recruitment and total egg production in the unfished

substock, R_{0s} and E_{0s} respectively, are subscripted here to emphasise that these parameters are specific to substock s. The $N_{s,I}$ recruits were evenly distributed among the i cells that were defined as nursery cells for substock s.

2.2.2 Distribution of fishing effort

Elaboration of the disc equation as a density-dependent catchability function

Holling's disc equation was chosen as the basis for the distribution of fishing effort because it can be used to simulate the concentration of fishers on aggregations. These are fished down in turn as fishers rapidly distribute effort among sites, while repopulation at any site is over a larger time scale. In this study, effort was distributed each month, spatial biomass was usually distributed by movement also on a monthly scale. Expressing catchability as inversely related to biomass, the disc equation represents the hyperstable (catch rates remain high as population abundance declines) nature of fisheries based on aggregations (Hilborn and Walters 1987; Hilborn and Walters 1992). The "value" of fishing at a particular site can be modelled so as to include a variety of elements, but here I have included population density and costs at location.

The basic equation (Holling 1959; Hilborn and Walters 1987; 1992) can be derived by considering the time budget of a day's fishing operation. Fishing time, T, is the sum of two components, searching time (T_s) and handling time (T_h), respectively defined as time spent searching for and testing aggregations, and the time spent in taking and processing the catch from those aggregations. Thus,

$$T = T_s + T_h \qquad\qquad \text{Eq 2.5}$$

With the assumption that aggregations are randomly distributed in the area searched, the number of aggregations, A, searched and tested during T_s, is calculated as:

$$A = aT_sD,$$

with a being the effective area searched per unit time and D, the density of aggregations.

Handling time is the product of the average time taken to catch and process each aggregation, h and the number of aggregations,

$$T_h = hA.$$

After substituting these relations into Eq 2.5 and rearranging, the number of aggregations fished in time T is expressed

$$A = aTD/(1 + ahD) \qquad \text{Eq 2.6}$$

Local abundance (as biomass), B is the product of density D and average aggregation biomass, b, so that $B = bD$. Recognizing that catch is the product of the number of aggregations taken (A) and the biomass per aggregation, b, and substituting for D in Eq 2.6 allows catch and daily catch rate, respectively, to be expressed as functions of local biomass:

$$Catch = aTB/(1 + ahB/b) \text{; and,}$$

$$c = aB/(1 + ahB/b). \qquad \text{Eq 2.7}$$

Note also that local biomass can be predicted from catch rate:

$$B = 1/(a/c - ah/b) \qquad \text{Eq 2.8}$$

Furthermore, given the standard fishery equation for catch rate (the catch removed by a unit of fishing effort),

$$c = qB \qquad\qquad \text{Eq 2.9}$$

where q is catchability, then by comparison of coefficients in Eq 2.7 and Eq 2.9, the catchability is expressed

$$q = a/(1 + ahB/b) . \qquad\qquad \text{Eq 2.10}$$

For a given h, q is predicted to increase with school biomass, b Note also that q is at a maximum when $h = 0$.

Relating effort to catch, catch rate and abundance

Recognizing catchability as a function of biomass, rather than a constant, has the consequence that catchability can no longer be used in a straightforward manner to predict the incremental fishing mortality rate imposed or catch removed by a given unit of effort. Because catch rates are not proportional to abundance, then it is no longer feasible to predict the spatial distribution of effort simply in proportion to local abundance (biomass).

This challenge in simulation of fisheries was addressed previously by Hilborn and Walters (1987) with an iterative approach. I develop below the basis of a less computationally-intensive iterative method for predicting short-term spatial effort distributions. It is first necessary to assume that biomass dynamics over any short period while there is fishing are described approximately (ignoring natural mortality and growth during that short period) by

$$dB/dt = -cf , \qquad\qquad \text{Eq 2.11}$$

where f is the number of fishing units (eg boats or operations). Separating variables

$\int (1/c)dB = -\int f dt$, and integrating gives

$$\frac{1}{a}\ln B + \frac{h}{b}B + C_1 = -E \qquad \text{Eq 2.12}$$

E, the integral of f, is the effort required to drive the biomass down to level B (and corresponding catch rate, c), and C_1 is an arbitrary constant. At the start of fishing, $E = 0$, and let biomass be B_{start}. Let the biomass remaining after a period of fishing be B_{end}, then E is the effort required to drive biomass and corresponding catch rate, c_{end}, down to this level. Substituting these values into Eq 2.12 provides simultaneous equations

$$\frac{1}{a}\ln B_{start} + \frac{h}{b}B_{start} + C_1 = 0 \quad \text{Eq 2.13}$$

$$\frac{1}{a}\ln B_{end} + \frac{h}{b}B_{end} + C_1 = -E \quad \text{Eq 2.14}$$

Subtracting Eq 2.14 from Eq 2.13

$$E = (1/a)[\ln(B_{start}/B_{end}) + (ah/b)(B_{start}-B_{end})]. \qquad \text{Eq 2.15}$$

and recognizing that catch is the difference in biomass due to fishing

$$E = (1/a)\{\ln[(B_{start}/(B_{start}-Catch)] + (ah/b)(Catch)\} \qquad \text{Eq 2.16}$$

These equations generate the infamous relationship between catch rates and abundance for hyperstable fisheries (Figure 2.3).

Note from Eq 2.5 and Eq 2.8 that the catch rate corresponding to B_{end} is

$$c_{end} = aB_{end}/(1 + ahB_{end}/b) \qquad \text{Eq 2.17}$$

and predicted B_{end} given a c_{end} target is

$$B_{end} = 1/(a/c_{end} - ah/b) \qquad \text{Eq 2.18}$$
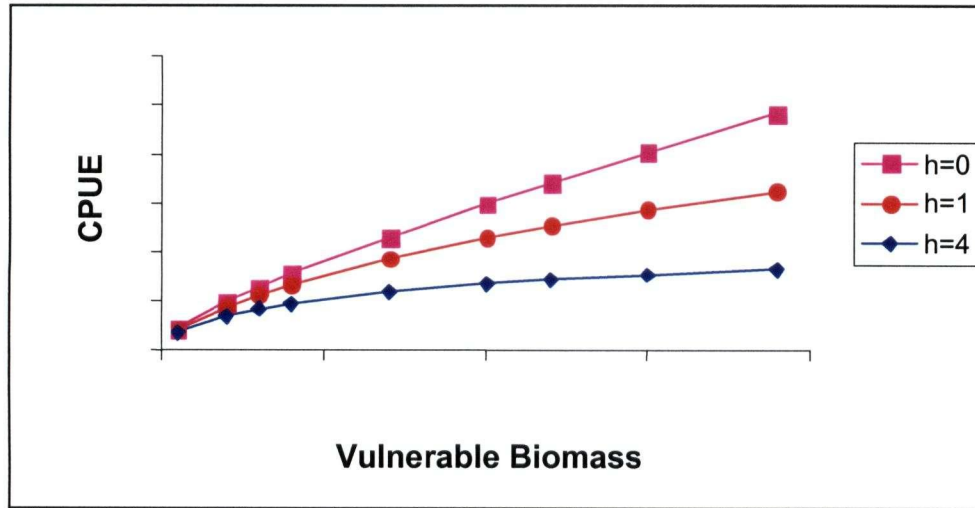
Figure 2.3. The relationship between catch rates (CPUE) and biomass for hyperstable fisheries. Units for the handling time *h* are relative.

This set of equations (Eq 2.16 – Eq 2.18) can be manipulated to predict the effort required, given a starting local biomass or catch rate, to produce a particular catch or to drive the catch rate or local biomass down to a particular level. It is also worthy of note that, if the effort occurs over a short time period, then $\ln(B_{start}/B_{end})$ approximates F, the instantaneous fishing mortality rate and when this is substituted into Eq 2.15,

$$E = F/a + (h/b)(B_{start} - B_{end}) \hspace{2cm} \text{Eq 2.19}$$

the term $(h/b)(B_{start} - B_{end})$ is the penalty in terms of extra effort, imposed by handling the catch, $(B_{start} - B_{end})$. This of course tends to zero with h where, with $q = a$, Eq 2.19 produces the usual expression, $E = F/q$.

If the assumption is further made that fishers will distribute their effort on the basis of catch rates (or value) at each location (Gordon 1953) so that catch rates at the different locations are

brought down to similar levels by successive applications of effort (so-called Ideal Free

Distribution or IFD assumption), the model produces the curvilinear trend in effort with catch of

Figure 2.4. This is diagnostic of targeted fishing.
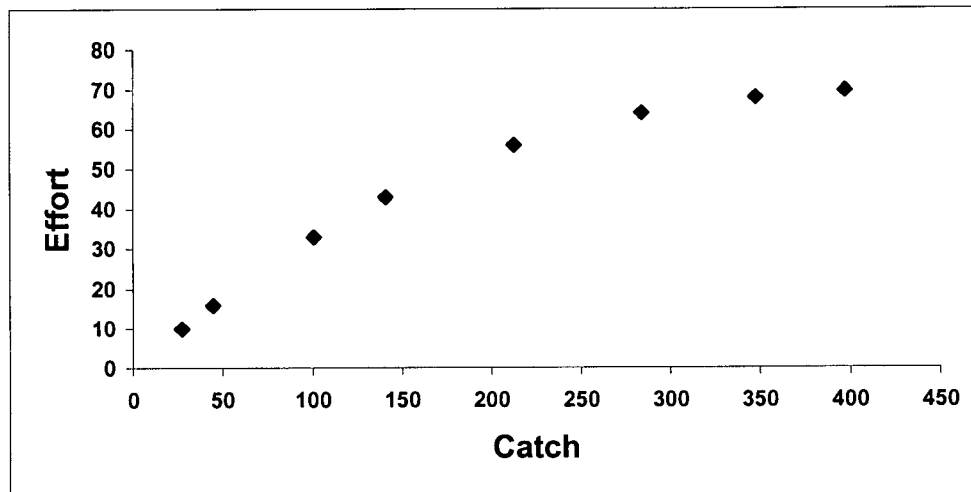
Figure 2.4. Trend in effort with catch due to spatial distribution of fishing effort.
Units are nominal values.

2.2.3 Scaling and parameterization of catchability and handling time

In considering the impacts of different capacity and hyperstability levels on fisheries, it is

valuable conceptually to express them in terms of the F that they may impose on fished stocks,

relative to F_{opt}, a constant fishing mortality rate that will maintain a fishery near optimum levels

over time (Walters and Parma 1996). The scenarios examined in this thesis are chiefly of

interest where a fishery has sufficient capacity, available effort, for overfishing to occur, ie it is

possible for F to exceed F_{opt}. Similarly, in the context of this work, it is the degree to which the

targeting of aggregations – the interaction between fish and fisher behaviour and technology –

impacts on the value of information gathered and the efficacy of management measures.

Hyperstability, is a symptom of this targeting. One of the important criteria for good fishery

monitoring and control combinations is that they keep a fishery near optimum levels even when

the catch rate is hyperstable and there is substantial effort capacity above that needed to achieve F_{opt}. Thus it is again the impact of the function onfishing mortality rates and catchabilities through time that are of interest, not particular values of the handling time parameter.

Therefore I included excess capacity by relating the fishing mortality rate that the fishery could impose, in the absence of hyperstability effects, with a given effort, E, to F_{opt}, by $R_c = F_0 / F_{opt}$. Given this ratio, if the base catchability at the start of fishing is q_0, then recalling the usual relationship between the fishing mortality rate and effort as $F = qE$, q_0 can be parameterized by entering R_c and calculating, respectively, the base fishing mortality rate and catchability as

$$F_0 = R_c F_{opt} \text{ and}$$

$$q_0 = R_c F_{opt} /(E / i / t) \qquad \text{Eq 2.20}$$

(here i, the number of spatial cells in the simulated fishery, and time unit, t, scale q_0 to unit area and time; $t = 12$ in this study, reflecting monthly distributions of effort and fish through movement).

Equations Eq 2.10 and Eq 2.15 to Eq 2.17 can be parameterised by allowing entry of relative handling time via another ratio of F values. This is achieved by recognising that as B is reduced to zero in Eq 2.10, q tends to $q_{max} = a$, F tends to F_{max}, and then making the substitution $h_s = ah / b$ into Eq 2.10. Note that a and b are constant across scenarios, so that changes in h_s are relative changes in h, the specific values of which need not be known in the context of this study. This allows for the simplification

$$q = q_{max} /(1 + h_s B) \qquad \text{Eq 2.21}$$

Let $q = q_0$ at $B = B_0$, so Eq 2.21 can be rearranged:

$$h_s = (q_{max} / q_0 - 1) / B_0. \qquad \text{Eq 2.22}$$

Here B_0 is the mean unfished biomass per spatial cell, calculated from the input value of unfished biomass. h_s can be parameterised by entering the catchability ratio in Eq. 2.22. This is equivalent to an F ratio, representing the relative growth in F, or in catchability, as biomass is reduced but if effort is held constant:

$$R_h = F_{max} / F_0 = q_{max} / q_0$$

Procedure for allocation of fishing effort across spatial cells

Fishers tend to distribute effort in such a way that the perceived value of catches from all spatial cells will be equalized (Gordon 1953; Hilborn and Kennedy 1992). A small variation to the above model, then, is to consider that fishers will moderate their fishing activities on the basis of value of their catch rather than just the catch rate. This can be by incorporating, for example, operating costs or prices that might be associated with particular localities (such as fuel used to fish sites distance from port). They will thus leave a fishing location when their catch rates decline to a certain value, V – that includes costs, prices etc - rather than at a specific catch rate c_{end}.

Accordingly, subject to management implementation described below, total effort available within any month, E_0, (standard effort, as a model input) was distributed amongst spatial cells so that, after fishing in each monthly time step, the marginal value after fishing in each cell was approximately equalised over all fished cells to the same value V (V is assumed to change among time steps with changes in fish distribution and abundance and with changes in total

effort). The basic effort allocation procedure involves finding that value of V such that the total

predicted effort over all cells sums to E_o. The marginal value for each cell i was the catch rate

predicted for that cell, using Eq 2.17, parameterised as above and penalised by the scaled cost

assigned to that cell, $Cost_i$

$$V_i = c_{end} - Cost_i \qquad\qquad \text{Eq 2.23}$$

and $V = \sum_i V_i$.

The effort in cell i, e_i, was estimated for any V using Eq 2.15 to predict c_{end} in Eq 2.23 and

solving the resulting equation for e_i. However, determination of e_i using Eq 2.15 required prior

calculation of the quantities B_{start} and B_{end}. The B_{start} was estimated as the vulnerable biomass,

B_v, in the cell, which was calculated as

$$B_v = \sum_a N_a w_a v_a ,$$

within the spatial cell, the sum across all ages of the number of fish N_a, multiplied by the mean

weights w_a at age, by the relative vulnerabilities v_a at age, all referenced to age a.

Substituting for c_{end} in Eq 2.18 using Eq 2.23 allowed B_{end} to be calculated as a function of V:

$$B_{end} = 1/[q_{max}/(V + Cost_i) - h_s]$$

meaning that the biomass after fishing in each cell is predicted by the value for that cell after

fishing. Each e_i could then be predicted from Eq 2.15, as

$$e_i = (1/q_{max})[\ln(B_v / B_{end}) + h_s (B_v - B_{end})], \text{ and } E_{est} = \sum_i e_i .$$

To find that value of V leading to $E_{est} = E_0$, V was chosen with Newton's method, i.e. updating

$V_{n+1} = V_n - (E_0 - E_{est})/E'_{est}$, where $E'_{est} = d(E_0 - E_{est})/dV = dE_{est}/dV$, (as E_0 is a constant), and

iterating to satisfy the constraint $|E_0 - E_{est}| \leq 0.01 E_0$.

The derivative E'_{est} was

$$E'_{est} = \sum_i q_{max} / (V_n - Cost_i)/(hV_n - q_{max} + h_s Cost_i)^2$$

For initial months of the simulation of each fishery, a seed value of $V = 099\ V_{max}$ for the

Newton's method search was calculated using Eq 2.7 and substituting for c_{end} using Eq 2.23,

$$V_{max} = \max[q_{max} B_v / (1 + h_s B_v) - Cost_i].$$

2.2.4 Calculation of optimum fishing rate

Application of the outputs of the model for the distribution of fishing effort as management

controls under a harvest rate strategy first requires definition of a target fishing rate. The

optimal target fishing rate, F_{opt} was determined using an equilibrium age-structured model based

on the approach of Botsford (1981), and as described for the reference population model.

To estimate F_{opt}, the following equilibrium calculations were made:

$$l_a = \varsigma^{a-1} \prod_{i=1}^{a-1} (1 - v_{i-1} U)$$
survivorship to age a, where $0 < a < a_{max}$

$$l_a = \varsigma^{a-1} / [1 - \varsigma(1 - v_a U)] \prod_{i=1}^{a-1} (1 - v_i U)$$
survivorship to age $a = a_{max}$

$$\varphi_{e0} = \sum_a l_a f_a$$

eggs produced per recruit given $U = 0$

$$\varphi_f = \sum_a l_a f_a$$

eggs produced per recruit given $U > 0$

$$\varphi_b = \sum_a l_a w_a$$

biomass per recruit

$$\varphi_{bv} = \sum_a l_a v_a w_a$$

vulnerable biomass per recruit

$$R = (a'\varphi_f - 1)/[(a'b)\varphi_f]$$

predicted equilibrium recruitment

$$Catch = R\varphi_{bv}U$$

catch given U

where a_r is age at recruitment, v_a is the relative vulnerability at age a, $U = 1 - \exp(-F)$ is a harvest rate. The age a_{max} represents a plus group to accumulate the small number of animals of that age and older, as for the reference population.

Also as for the reference population, the calculation of recruitment R is based on the Beverton–Holt S-RR. Given that the equilibrium fished population produces φ_f eggs per recruit, and thus population egg production is $R\varphi_f$, then the equilibrium stock-recruit condition is expressed as $R = a'R\varphi_f/(1 + a'bR\varphi_f)$, which reduced and solved for R is the equation above. Using a similar approach to that for parameterization of the S-RR for the reference population, and noting that r_s is the relative difference in recruitment curve slope for stock s relative to the substock 1 (as for the reference population S-RR calculations; Eq 2.1, it can be shown that recruitment equation above can be expressed:

$R = (r_s K_r \varphi_f - \varphi_{e0})/(K_r - \varphi_f)$, with the value for R reset to zero if this calculation produced a negative value.

Given R, the equilibrium catch can be expressed as the product of the equilibrium recruitment, vulnerable biomass per recruit, and harvest rate, as above. For each stock, $F_{opt\,s} = -ln(1-U)$ was then identified by choosing the U which maximised *Catch* when calculated over a fine grid of U values. I calculated a fishery target F_{opt} as an average over stocks, weighted by unfished stock size,

$$F_{opt} = \sum_s F_{opts} B_{0s} / \sum_s B_{0s}$$

with B_{0s} being the unfished biomass of stock s.

2.2.5 Optimum catch series

It is conceptually convenient to describe here a small dynamic age-structured model, used to describe an optimum catch series, against which fishery performance is later compared. The optimum catch series was generated by use of an age-structured model with the fishing rate fixed at F_{opt}. The population dynamics were represented by propagating a vector of survivorships, with numbers at age a in year t as $N_{a,t}$, and initiated :

$N_{1,1} = 1$

$N_{a,1} = \zeta N_{a-1,1}$ for $a = 2$ to a_{max-1}

$N_{a,1} = \varsigma/(1-\varsigma)N_{a-1,1}$ for $a = a_{max}$, where a_{max} is again a plus group to accumulate the small number of animals of that age and older. The survival rate ζ is as defined for the reference population.

Relative vulnerable biomass and egg production (each per recruit) were annually updated as

$$B_{vt} = \sum_a N_{a,t}v_aW_a \text{ and } Eggs_t = \sum_a N_{a,t}f_a \text{, with } v_a, W_a \text{ and } f_a \text{ each as previously defined.}$$

The optimum catch for year t was then computed as $C_t = [1 - \exp(-F_{opt})]B_{vt}$. Given that $N_{1,1} = 1$, the quotient, B_0 / φ_b, which is the unfished biomass of the reference population divided by the biomass per recruit, scales the catch to the reference population,

$$C_{opt,t} = [1 - \exp(-F_{opt})]B_{vt}B_0/\varphi_b \qquad \text{Eq 2.24}$$

for B_0 being the unfished biomass and φ_b defined as above. Numbers in the model were updated as

$$N_{a+1,t+1} = N_{a,t}\exp(-v_aF_{opt} - M), \text{ for } a > 1,$$

with recruitment to the first year class being described by a Beverton-Holt SRR:

$$N_{1,t+1} = \frac{K_r(R_0/E_0)Eggs_t}{\{1+[K_r(R_0/E_0)-1/\varphi_{e0}]Eggs_t\}}, \text{ with all inputs as previously defined.}$$

2.2.6 Implementation of management controls and harvest

The underlying approach to harvesting fish in the model is to predict a monthly fishing rate for each cell, based on the fishing effort allocated as above, then to apply this fishing rate to the

vulnerable biomass, by age and stock, within that cell. Alternative regulatory options are effected by multiplying the effort allocated to each cell within a statistical area, by a regulation factor. This factor is calculated to modify the effort allocated to each cell, according to the control rule in force. This means that if, in any month, the control target for a statistical area would be exceeded by the effort initially calculated, total effort allocated to that area is reduced.

Two management control options, acting at the level of statistical area, are included in the simulation framework. The objective of each control is to maintain the annual fishing rate near the optimum by containing fishing effort. The first control implements an annual total allowable catch (TAC), based on the cumulative catch for the year and the predicted catch for the coming month. The second implements an effort control based on the estimated fishing rate for the year (calculated from the cumulative annual fishing effort) and the predicted fishing rate for the month, relative to the target, F_{opt}. The controls rely on information either from the annual stock assessments, or the direct estimate of catchability from mark-recapture, as described below.

Calculation of regulation factors

I first describe the rules for most years of the fishery simulation (years 6 to 40). I then describe modifications to the control rules for early fishery development periods (simulated years 1-5) of each fishery, before much information would normally be available from stock assessments.

TAC Control Option

In all but the first year of each fishery year, given the TAC calculated for the statistical area k, (TAC_k,) in the assessment at the end of the previous year, the regulation factor for each statistical area was calculated each month as

$$reg_k = (TAC_k - C_{cum}) / C_{pred},$$

with the constraints that if $reg_k > 1$, it was reset so that $reg_k = 1$, while if $reg_k < 0$, then it was reset so that $reg_k = 0$. The term C_{cum} was the sum of catches, reported without error, from the statistical area over previous months within the year, and the catch predicted for the statistical area in the current month calculated as the sum of predicted catches by cell

$$C_{pred} = \sum_i [B_i \{1 - \exp[-q_i e_i / (1 + h_i B_i)]\}].$$

where i references the spatial cells.

Effort Control Option.

Effort control regulation in any month was based on the "remaining" F for the statistical area, the difference between F_{opt} and predicted F based on the most recent estimate of catchability (see below) and effort already accrued for the year, relative to the predicted F that would be imposed by the expected effort for the coming month. The regulation factor was calculated:

$$reg_k = (F_{opt} - Q_k E_{cum,k}) / Q_k E_{m,k}.$$

Here Q_k is the most recent estimate of catchability for area k, via stock assessment, or directly from tagging. $E_{cum,k}$ is the cumulative effort, reported without error, of previous months of the same year and $E_{m,k}$ is the projected effort for the statistical area, the sum of the e_i predicted for the area k, above. With the same rules as for the TAC control, reg_k was constrained so that $0 \le reg_k \le 1$.

Modification during Fishery Development Phase.

The development phase was considered to be the first 5 years of operation for each fishery simulation. The objective during this phase was assumed to be to provide a slow initial development of the fishery. As most monitoring measures required information gathered during the year to provide information for assessment, all fisheries were managed during the first year by a TAC determined as the total optimum catch for year 1 (as above), divided by the number of statistical areas (k), divided by 5, ie

$$TAC_s = C_{opt1} / k / 5,$$

and the reg_k set as for TAC in the description above. Initial trials indicated that for high catchability fisheries, substantial biomass reduction even in the first year could completely dominate fishery performance if fishing were uncontrolled.

For years 2 to 5 of each fishery, the reg_k was simply reduced by the factor $y/5$, where y is the year of operation of the fishery.

Harvest

Catch was calculated on a by-cell basis. The numerical catch was simply the product of a monthly harvest rate, the number of fish at each age a, and the schedule of the relative vulnerability at age. The harvest rate for each month for each spatial cell i (within a statistical area k) was

$$U_i = 1 - \exp[-q_{max} reg_k e_i /(1 + h_s B_{vi})]$$

where q_{max}, reg_k, e_i, and h_s are as previously defined, and B_{vi} is the vulnerable biomass, also as defined above, for the cell i. Note that the disc equation is employed at the cell level as a

catchability function. Numerical catch was defined by applying this harvest rate to the number

of vulnerable fish by each stock *s* and age *a* within the cell *i*, ie

$$CatchN_{s,a,i} = U_i N_{s,a,i} v_a ,\qquad\qquad Eq\ 2.25$$

and monthly catch by weight was simply

$$CatchW_{s,a,i} = CatchN_{s,a,i} w_a\qquad Eq\ 2.26 .$$

Catches by cell, age, statistical area or by stock, for subsequent use in population dynamics or

monitoring calculations, were simply calculated as appropriate sums of the quantities produced

in Eq 2.25 and Eq 2.26, across the cells occupied by a stock, or across cells comprising a

statistical area.

2.2.7 Monitoring information

Some of the more standard monitoring methods available to fisheries management are included

in the simulation framework. Mark-recapture (tagging), abundance surveys, sampling to

estimate mean weight, and aggregate and spatially weighted CPUE data, represent a suite of

methods that might be available in different fisheries, producing data series for assessment each

year, feeding to management control targets for the subsequent fishing year. Although a fishery

manager might use two or more of the measures concurrently, I used just one measure in any

scenario, with the intention of exposing the relative performance of each measure.

The level of precision in each measure was set for each scenario. Each of the monitoring

approaches included some measure of the investment in information gathering, manifested as

precision in sampling. For mark-recapture and mean weight monitoring this was different levels

of sampling - the number of tags released and the number of fish weighed, respectively. For the survey measure, precision was determined by the coefficient of variation (CV) in the survey abundance index; for spatial versus aggregate CPUE, the comparison was the investment by the fishery in collection and analysis of logbook data at different levels of spatial and temporal resolution.

In comparison of spatial management scenarios, monitoring effort was divided among the statistical areas so that the monitoring investment for the total fishery was the same for all scenarios. Thus a fishery managed as three statistical areas and a fishery managed as one area each received similar total monitoring input. For example, if monitoring was by mark-recapture or mean weight at a level of 600 tags released or 600 fish weighed, each of the statistical areas in a 3-area fishery would receive 200 tags, or 200 fish weighed, while a 1-area fishery would receive 600 tags or weigh 600 fish. However the CV for survey did not differ between comparable single and multiple statistical area scenarios, under the rationale that a survey in each area would achieve a similar precision by a smaller survey over a smaller area.

Mark-recapture

The annual harvest rate for a statistical area was estimated from a tagging program,

$$\hat{U} = tags_c / tags_r \, .$$

The number of marks released each year, $tags_r$ was provided as a data input. The number of recaptures, $tags_c$ was obtained by simulated sampling from a binomial distribution, given $tags_r$ (the number of trials) and the probability of recapture, which was assumed equal to the

simulated annual harvest rate U_k for the statistical area k. This was calculated as the annual catch weight divided by the vulnerable biomass (at the start of the fishing year) for the area:

$$U_k = CatchW_k / B_{vk} \qquad \text{Eq 2.27}$$

given catch weight for the area was

$$CatchW_k = \sum_{m,s,a,i} CatchW_{m,k,s,a,i} \qquad \text{Eq 2.28}$$

ie the sum of catches by weight across all cells i, ages a, substocks s and months m for the year, within the statistical area k.

The vulnerable biomass was the sum of the numbers of fish of each age multiplied by the relative vulnerability at age by the weight at age, for each stock and each cell within the statistical area, at the start of the fishing year

$$B_{vk} = \sum_{s,a,i} N_{k,s,a,i} v_a w_a \qquad \text{Eq 2.29}$$

with all notation as previously described.

Survey

It was assumed that a survey-based index of abundance would sample from a log-normal distribution, so that the index for a statistical area k was calculated by

$$Y_k = g_y B_{vk} \exp(\gamma a_s),$$

where B_{vk} is again the vulnerable biomass for statistical area k, (Eq 2.29), γ is a random normal error with zero mean and unit variance, and a_s approximates the coefficient of variation (CV) in

the survey estimate. The error γ was sampled separately for each of the statistical areas but with a_s kept constant across those areas. A smaller a_s would represent higher precision and thus a greater investment in a survey. In this simulation context, the catchability coefficient for survey is arbitrary and was set at $g_y = 0.1$. Notionally, the survey might represent a trawl survey, gillnet or aerial survey – any approach that provides an estimate of the stock biomass and its associated error of this form.

Spatially-referenced CPUE

Within any statistical area k, the spatially-referenced CPUE was the mean of the monthly m CPUE values for all cells i in the area:

$$SCPUE_k = \sum_{i,m}(Catch_{k,m,i} / Effort_{k,m,i}) / Obs_k,$$

given monthly area catches $Catch_{k,m} = \sum_{s,a,i} CatchW_{k,m,s,a,i}$, effort $Effort_{k,m,i} = \sum_{s,a,i} e_{k,m,s,a,i}$ and Obs_k is the number of cells over which catch per effort was averaged in area k. Note that this index uses a ratio estimator for catch per effort within each cell, but averages these estimators over cells (sampling strata) without assigning greater weight to cells with higher fishing efforts (which would result in hyperstability in the index if effort were concentrated in cells with higher fish abundances).

Aggregate CPUE

This was calculated simply as

$$ACPUE_k = CatchW_k / Effort_k,$$

the ratio of the total annual catch, $CatchW_k$ (Eq 2.28) to total annual effort,

$Effort_k = \sum_{m,s,a,i} e_{k,m,s,a,i}$ for statistical area k. Note that this ratio estimator would be expected to

show hyperstability due to non-random distribution of effort among cells both within and among

statistical areas.

Mean weight

Sampling to establish mean weight of fish in the catch was simulated using an additive

observation error model,

$$W_{obs} = CatchW_k / CatchN_k + a_k \sqrt{\delta_w^2 / N_w},$$

with $CatchN_k = \sum_{m,s,a,i} N_{k,m,s,a,i}$ $CatchN_{k,n}$ being the numerical annual catch for year n for the

statistical area across all months, stocks, ages and cells. The elements of the error term are a_k, a

random normal error with zero mean and unit standard deviation, N_w, the number of fish

randomly sampled each year to estimate the mean weight, and δ_w^2, the variance in weight of the

unfished vulnerable population. The variance was calculated as

$$\delta_w^2 = [\varphi_{bv2} - (\varphi_{bv})^2 / l_{av}] / l_{av}. \quad \text{Eq 2.30}$$

I again used the Botsford (1981) equilibrium analysis approach to calculate the terms for

calculation of Eq 2.30 from the reference population model:

φ_{bv} \qquad\qquad vulnerable biomass, as previously described,

$\varphi_{bv2} = \sum_a l_a v_a w_a^2$ \qquad\qquad corresponding square term, and

$$l_{av} = \sum_a l_a v_a$$ survivorship of the vulnerable population.

Use of variance from the unfished population provided computationally efficiency relative to an annual calculation of the variance in weight from the fished population. The error introduced would be minor.

2.3. Fishery Assessment

Assessment of the fishery was by two distinct methodologies. Firstly, in some scenarios, mark-recapture data were used to provide direct estimates of fishing rates, and to update the estimated catchability for the fishery annually, employing a Kalman filter (Walters 1986; Schnute 1994; Reed and Simons 1996). In the alternative methodology, full population assessment was necessary to make use of the indices of abundance provided by simulated monitoring survey, CPUE, mean weight information or mark-recapture data.

For the population assessments, I adopted the "parallel populations" approach (Walters 1994; Walters 1998b). This entailed simultaneously simulating a range of delay difference populations, initially differing in their unfished population biomass, B_0, and the various parameters that are calculated from it, but otherwise identical in biological and relative production characteristics. Corresponding parameters of interest for each hypothesized population, such as current population biomass and number, mean weight values, and catchability, were calculated at each annual update step. Assessment each year used the simulated monitoring information, calculating a likelihood for each B_0 hypothesis given the time series of biomass indices, and so identifying the maximum likelihood hypothesis (Walters and Ludwig 1994).

For the CPUE, mean weight and tag monitoring methods, population assessment at the beginning of year *t+1* comprised identification of the most likely B_0 hypothesis, given information collected previous to and during *t*, so estimating the biomass at the beginning of year *t*, ie $\hat{B}_{ht|t}$. The TAC or TAE was then calculated with the biomass prediction for year *t+1*, under that B_0 hypothesis, $\hat{B}_{ht+1|t}$. This entailed updating for each B_0 hypothesis by accommodating catch during *t*, biomass growth and predicted recruitment in the delay difference model.

I treated the survey-based assessment differently, however, on the rationale that survey timing would be controlled so as to focus on the population at the start of each fishing year, but subsequent to the annual recruitment event: it was a pre-season abundance survey. Thus for survey-based assessment, controls for fishing in year *t+1*, as well as the survey abundance index, and assessment, were all based on information or estimates for year *t+1*, ie TAC and effort control targets for *t+1* were calculated with the biomass prediction for year *t+1*, based on data for *t+1*.

2.3.1 Population assessment model description

The primary criterion for choice of an assessment framework was that any shortcomings in assessment or development performance be ascribed to the combinations of monitoring and management controls relevant to spatial dynamics, rather than model specification. At the same time, this project required a very large number of population simulations. The Deriso-Schnute delay difference model (Deriso 1980; Schnute 1985) was chosen, being both computationally compact and, although a biomass dynamic model, implicitly including the core dynamics of

age-structured population models. It retains, in particular, the lags related to recruitment. The assessment model thereby generates very similar dynamics to the fully age-structured reference model.

I applied a modified form of the implementation described by Hilborn and Walters (1992). The initial biomass value for each unfished stock size hypothesis was derived by entering maximum and minimum values of the unfished biomass B_o, and the number of B_o hypotheses into which this grid was to be divided (Walters 1998b). The biomass dynamics are described by

$$B_t = G_{t-1}B_{t-1} + w_n R_{t-1}$$

where w_n, is the mean weight at the age (n) of recruitment to the fishery and R_t is the number of recruits in year t. Inputs were B_0 for each biomass hypothesis and w_n. The growth-survival factor, G_t for year t is

$$G_t = \rho s + s\alpha / \overline{w}_t$$

with ρ and α respectively the slope and intercept of the plot of weight at consecutive ages (ie a Ford-Walford plot, w_{t+1} on w_t), s is the annual survival rate (after fishing and natural morality) and $\overline{w}_t$ is mean weight of the vulnerable population in year t. The number in the vulnerable population at the beginning of any year was calculated as

$$N_t = sN_{t-1} + R_{t-T_R} \qquad\qquad \text{Eq 2.31}$$

where T_R is the time lag between spawning and the age at recruitment. The Beverton-Holt (1957) stock recruitment relationship was again used to describe recruitment, with the R_t recruits related to spawning stock size S_t, by

$$R_t = a'S_t / (1 + b'S_t) \qquad \text{Eq 2.32}$$

with a' being the maximum recruitment per spawner as the number of spawners approaches zero, and a'/b' being the maximum number of recruits produced when S_t is very large (Hilborn and Walters 1992). The recruitment model parameters are derived given an input of the compensation in recruitment rate at low population size, K_r, (Myers *et al.* 1999),

$$a' = K_r R_0 / S_0 \qquad \text{Eq 2.33}$$

and assuming unfished equilibrium conditions at the start of the fishery, substituting Eq 2.33 into Eq 2.32, it can be shown that

$$b' = a'/(1-s)/N_0 - 1/S_0 .$$

Defining $S_0 = B_0 - w_n N_0$, as the spawning biomass and prior calculation of the starting mean weight under initial equilibrium conditions,

$$\overline{w} = \frac{s\alpha + w_n(1-s)}{1 - \rho s},$$

initial numbers in the stock, $N_0 = B_0 / \overline{w}$ allowed calculation of the S-R R parameters from the inputs B_0, K_r, and w_n:

$$a' = K_r(1-s)/[\overline{w}(1 - w_n / \overline{w})], \text{ and}$$

$$b' = a'/R_0 - 1/(B_0 - w_n N_0) .$$

2.3.2 Parameter specification and initiation

Apart from the parameters related to initial population size (the B_0 for each hypothesis), delay difference model input parameters (Table 2.5) were equal to or derived from those input to the reference population model. In most respects this ensured that any assessment failures or problems could not be ascribed to differences in components of the modelling systems. However, in all assessments the recruitment compensation parameter was equivalent to the base (the most optimistic) value of K_r for the reference population.

The Ford –Walford parameters α and ρ were estimated by linear regression from the weights at age input in the reference population sequence.

Table 2.3. Input values for parameters of the population assessment model

Parameter	Value	Description
B_{0min}	500 t	Minimum of range of initial biomass
B_{0max}	10 000 t	Maximum of range of initial biomass
n_{B0}	50	Number of initial biomass values on assessment grid
α	2.226 kg	Intercept of plot of weight at consecutive ages (Ford-Walford plot)
ρ	0.9274	Slope of plot of weight at consecutive ages
ϑ	0.68 year^{-1}	Annual survival rate in the absence of fishing (=$exp[-M]$)
K_r	6	Recruitment compensation: the relative increase in the number of recruits at low stock sizes
W_n	5.772 kg	Mean weight at age of recruitment n
T_R	3 years	Recruitment time lag

Implementation

I generally adopted the approach recommended by Hilborn and Walters (1992) for simulating the hypothesis populations over time. However, in initial trials, I found that hypothesized

biomass values could change so rapidly in some high effort scenarios, that assessments would fail. It was not possible to choose a hypothesis within the grid of hypotheses (B_0 s) supplied, because the hypothesis trajectories diverged very rapidly (also noted by Walters and Ludwig 1994). At the same time, catchability q values for the reference population increased very strongly. The usual approach to estimation of catchability is to treat q as a nuisance variable and calculate it from other inputs or leading variables, for example assuming it to be linear with effort and F, annually updated as $\hat{q} = \sum EF \, /(\sum E^2)$, and where the information for all years up to t is equally valued. Clearly q could not be treated in this way for this study, where catchability changes might be large. I therefore modified the approach to track the change in catchability between years, using this value with fishing effort to predict the reduction in biomass due to harvest, under each hypothesis. That this differs from most assessment approaches, which deduct the recorded catch, and in a practical context would require that effort is well-standardized.

The annual update process for each hypothesis, for each statistical area k, entailed the following series of steps for each statistical area k in year t, and hypothesis h:

an initial estimate of fishing mortality rate, $F_{k,h,t-1} = -\ln(1 - Catch_{k,t} \, / B_{k,h,t})$, with $Catch_{k,t}$ being the catch and $B_{k,h,t}$ the predicted vulnerable biomass,

an estimate of catchability using a weighted regression

$$\hat{q}_{k,h,t} = (\lambda \sum_{k,h,t} E_{k,t-1} F_{k,h,t-1} + E_{k,t} F_{k,h,t}) /(\lambda \sum_{k,t} E_{k,t-1}^2 + E_{k,t}^2), \qquad \text{Eq 2.34}$$

with $E_{k,t}$ being the total fishing effort. In all simulations, I set the weighting on past information in the regression at $\lambda = 0.5$. (I found in initial trials that the assessment was not sensitive to the value chosen).

for $t \geq 2$, an improved estimate of fishing mortality rate employing this catchability estimate

$$F_{k,h,t} = \hat{q}_{k,h,t} E_{k,t}$$

calculation of the spawning stock remaining after fishing as

$$S_{k,h,t} = (B_{k,h,t} - w_n N_{k,h,t}) \exp(-F_{k,h,t}) \, ;$$

recruitment

$$R_{k,h,t} = a' S_{k,h,t} / (1 + b'' S_{k,h,t}) \, ; \cdot$$

total survival after fishing and natural mortality, M,

$$s_{tot} = \exp(-F_{k,h,t} - M) \, ,$$

update of population biomass (accommodating recruitment time lag)

$$B_{k,h,t+1} = s_{tot}(\alpha N_{k,h,t} + \rho B_{k,h,t}) + w_n R_{k,h,t-T_R} \, ,$$

noting that recruitment to the fished population provided by spawning in year t is lagged by T_R years.

Use of a weighted regression approach accommodates rapid change in annual q, weighting the most recent year's information. An alternative but notionally similar approach to a weighted regression would be to use a Kalman filter (Walters 1986; Schnute 1994; Reed and Simons 1996), as applied below for direct estimation of catchability from mark-recapture information.

This technique weights new information based on the ratio of process error to total (process plus observation) error, so is more explicit in requiring each source of variation be detailed. However, independent knowledge of the extent of inter-annual variation in catchability (process error) would be required. In most situations, this would not be available, and an informal approach as adopted for the regression weighting, as used here, would be necessary (Martell and Walters 2001).

Tracking catchability using fishing rate information from mark-recapture

For tracking catchability using a mark-recapture approach, the underlying model that I adopted was that catchability q deviated randomly each year from the previous year's value. This approach not only acknowledges that catchability will change randomly due to various unspecified processes, but also provides for long-term directional change (Schnute 1994).

An initial estimate of the catchability each year was simply the estimate from the previous year, i.e.

$\hat{q}_{k,t|t-1} = \hat{q}_{k,t-1|t-1}$, the notation $\hat{q}_{k,t|t-1}$ signifying "the estimate of q in year t and area k, conditional on previous observations on q made up to and including the year t-1".

An annual observation from the mark-recapture program was used to update this estimate of catchability using a Kalman filter (Walters 1986; Schnute 1994). This observation in year t, in each statistical area k of the fishery, was derived

$$\hat{q}_{k,t} = -\ln(1-X)/Effort_{k,t} \qquad \text{Eq 2.35}$$

given $X = tags_{c,k,t} / tags_{r,k,t}$ (an estimate of the annual harvest rate).

This was used to update the state estimate of catchability

$$\hat{q}_{k,t|t} = \hat{q}_{k,t|t-1} + \lambda(\hat{q}_{k,t} - \hat{q}_{k,t|t-1}) \qquad \text{Eq 2.36}$$

For use in Eq 2.36, the Kalman gain, λ (Eq 2.37), was calculated from the approximate inter-annual variance of catchability, σ_V^2, which required input as independent information, and variance of the mark-recapture estimate of catchability, (Eq 2.38)

$$\lambda = \sigma_V^2 / (\sigma_V^2 + \sigma_U^2) \qquad \text{Eq 2.37}$$

$$\sigma_U^2 = X_{k,t}(1-X)_{k,t} / tags_{rk,t} / (Effort_{k,t})^2 \qquad \text{Eq 2.38}$$

Initial trials across a range of values indicated that results were not very sensitive to σ_V, and this was set at 0.2. Given the estimate of catchability and annual catch and effort data, the vulnerable biomass for the area k at the beginning of year t was also estimated

$$\hat{B}_{vk,t} = CatchW_{k,t} / [1 - \exp(-\hat{q}_{k,t|t}Effort_{k,t})] \qquad \text{Eq 2.39}$$

The approach described here specifies a compromise each year between a new observation of q, and previous estimates based on older observations - the Kalman filter gain is merely the proportion of the total variance (process plus observation) that can be ascribed to interannual variation in the value of catchability. It is desirable that the technique should be sufficiently sensitive to pick up consistent change in catchability (weighting new information) while not being oversensitive to unaccounted variation in any new mark-recapture observations. Methodology for estimating or choosing σ_V is an area that warrants future investigation. Although I found the results in this study were not very sensitive to the specific value of σ_V,

lack of independent information about variation in catchability is a potential source of problems for estimation using this technique. Independent information might be sourced from other monitoring methods, or other fisheries.

Likelihood calculation for abundance indices

For the population assessment, each separate hypothesis about the unfished biomass (B_0) implies a separate biomass trajectory - a time series of annually predicted population biomass (B_t) - reflecting the history of harvest. The degree to which the alternative hypotheses agree with the monitoring data series is evaluated by calculating likelihood kernels and posterior probabilities for each hypothesis, given that data, using the methodologies of Walters and Ludwig (1994) and Walters (1998b). Assuming normal or lognormal error distributions for survey, CPUE and mean weight abundance indices, for each simulated year t, for each statistical area k, and for each population hypothesis h, the marginal likelihood is proportional to

$$L_t(B_h \mid Y_{k,t}) = SS_h^{-(t-1)/2}, \text{ for}$$

$$SS_h = \sum_{t=1}^{t}(z_t - \bar{z}_t)^2 \text{ (the sum of squared deviations), and}$$

$$z_t = \ln(Y_{k,t} / B_{k,h,t}), \qquad \text{Eq 2.40}$$

for survey and CPUE abundance indices, or

$$z_t = \ln(\hat{w}_{k,t} / \overline{w}_{k,h,t}) \text{ (for mean weight)}.$$

Here $Y_{k,t}$ is the year t index of abundance, $B_{k,h,t}$ is the estimated population biomass under hypothesis h, $\hat{w}_{k,t}$ is the mean weight estimated from sampling, and $\overline{w}_{k,h,t}$ is the mean weight predicted under the population hypothesis h.

The sum of squares for each hypothesis was updated each year t for each hypothesis, using the relationship

$$SS_h = S_{1h} - S_{2h}^2 / t,$$

with $S_{1h} = \sum z_h^2$ and $S_{2h} = \sum z_h$ (Walters and Ludwig 1994).

For the binomially-distributed mark-recapture data used for population estimation, a likelihood kernel was calculated annually (logged to avoid computation limit problems)

$$lL_t(B_{kht} \mid tags_{ck,t}) = tags_{ck,t} \ln(CatchW_{kt} / B_{k,h,t}) + (tags_{rk,t} - tags_{ck,t}) \ln(1 - CatchW_{kt} / B_{k,h,t})$$

then the loglikelihood for the time series was updated as

$$lL_t(B_{k,h} \mid tags_{ck}) = \sum_t lL_t(B_{kht} \mid tags_{ckt}) \quad \text{and the likelihood calculated}$$

$$L_t(B_{k,h} \mid tags_{ck}) = \exp[lL_t(B_{k,h} \mid tags_{ck})] \qquad \text{Eq 2.41}$$

The number of marks released each year, $tags_r$ provided as a data input, was divided evenly among the k statistical areas so that the number of tags released in any area k in year t was

$$tags_{rk,t} = tags_r / k.$$

The maximum likelihood hypothesis was thus identified, and was used for subsequent calculation of the TAC for the coming simulated fishing year. The approximate posterior probability for each hypothesis was defined as

$$P_t(B_h \mid H_{k,t}) = L(Y \mid B_h)P_h(B_h) / \int_{B_h = B_{min}}^{B_h = B_{max}} L(Y \mid B_h)P_h(B_h)dB_h \, ,$$

the probability in year t of the biomass hypothesis B_h given the time series of observations $H_{k,t}$ evolved by the monitoring measure.

This posterior probability was evaluated by firstly summing the L values over all hypotheses then dividing L for each hypothesis by that sum. The expectation of catchability for the statistical area was the probability-weighted sum of catchabilities across the hypotheses:

$$\hat{q}_{k,t} = \sum_h P_h \hat{q}_{k,h,t} \qquad\qquad \text{Eq 2.42.}$$

Setting of management control targets using assessment outputs

Outputs of the fishery assessments were calculated to provide values for use in simulated management control. The catch control Total Annual Catch (TAC) was calculated as

$$TAC_{k,t+1} = [1 - \exp(-F_{opt})]B \qquad \text{Eq 2.43.}$$

B in Eq 2.43 represents either $\hat{B}_{vk,t}$ from the maximum likelihood biomass predicted for year t when mark-recapture was used to track catchability (Eq 2.39), or $\hat{B}_{\max L,k,t+1}$, the predicted biomass for year $t+1$ from the maximum likelihood biomass hypothesis from the stock assessment (Eq 2.41). F_{opt} is calculated as in Section 2.2.4 and is assumed known.

For effort control, the requirement was for an estimate of catchability, Q_k. From the mark-recapture estimation, this was $\hat{q}_{k,t|t}$ (Eq 2.36), while from the stock assessments, this was $\hat{q}_{x,h,t}$ (Eq 2.34).

2.4. Performance measures of monitoring/management control combinations

Information for measures of assessment and development performance was saved from the Monte Carlo simulations (all are over the time horizon, T years):

the sum of the absolute differences between the series of optimum catch and catches each year

$$L_{abs} = \sum_{t}^{T} | \, Catch_{opt}(t) - Catch_t \, | \qquad \text{Eq 2.44,}$$

and a corresponding relative measure,

$$L_{rel} = \sum_{t}^{T} | \, Catch_{opt}(t) - Catch_t \, | \, / \, Catch_{opt}(t) \quad \text{Eq 2.45.}$$

For each of substocks 1 to 4, the number of years which the substock declined below 20% of the unfished biomass for that substock.

All loss functions are summarised as the mean and standard deviation (SD) over the I replicate simulations for each fishery configuration.

A wide range of performance measures have been used in fishery assessments (e.g. McAllister and Kirkwood 1998a; Punt and Smith 1999). Francis and Shotton (1997) found 39 different measures used in just 20 published risk assessments. The choice of measures seems to be context sensitive. In this study, the management strategy was to control harvest near a target

fishing rate, $F_{opt.}$, under varied spatial dynamic conditions. Thus it is desirable that the primary performance measures evaluate how well/badly different monitoring and management combinations keep fishing levels near F_{opt}. It is appropriate, then, to use the two Walters (1998b) loss measures (Eq 2.44, Eq 2.45) to compare the series of fishery catches, to the optimum catch series generated by F_{opt}, $Catch_{opt}$ (t), elaborated in Section 2.2.4. Both measures make the assumptions that deviations of annual catches from $Catch_{opt}$ (t) are equally undesirable whether above or below the optimum, and that there is no discounting of future values - future deviations are just as important as current ones. Although similar in appearance, the measures in Eq 2.44 and Eq 2.45 differ in that the absolute measure tends to favour policies that assume B_0 is somewhere in the middle of its probable range (= "certainty equivalence", Walters 1986), whereas L_{rel} tends to favour cautious policies, that do not risk over-harvesting by assuming that B_0 is small (Walters 1998b).

It is important to consider that taking substantially greater catches than $Catch_{opt}$ (t) will impair long term capability to take catches at that level: combinations of spatial dynamic structures and management/control sets that tend to result in overfishing will be penalized provided the time horizon T is long enough for the losses due to overfishing to become evident (Walters 1998b). Given the structure of the modelling system and the nature of the management controls (targeted at F_{opt}), severe or consistent overfishing is likely to arise from assessment errors or bias due to the spatial dynamic structures, and would be manifested early in the time series, and correspondingly penalized. Given this reasoning, low discount rates are also unlikely to have much impact on comparisons between spatial dynamic structures or monitoring/ management sets. Low discount rates did not significantly affect policy comparisons in the work of Walters (1998b).

The measures of frequency of a substock's biomass being driven to low levels are included as an adjunct to the loss functions above, to add shades of meaning. The widespread shift to more precautionary legislation for fisheries has been evident in Australia with the introduction of the *Environment Protection and Biodiversity Conservation Act* (Commonwealth of Australia 1999, www.ea.gov.au/epbc/index.html). This Act requires that all fisheries based on export of marine species undergo assessment to determine the extent to which management arrangements ensure ecologically sustainable management. During negotiations with the Federal Government over assessment of NT-managed fisheries, there were consistent queries of the form of "what is the risk of the population biomass falling below x% of the unfished biomass?" Similarly, the emphasis on sustainability embodied in FAO (1996) suggests that the frequency with which biomass becomes "dangerously low", as well as a management system's ability to detect this, are also of interest. Measures such as those used here have received widespread, but frequently inconsistent use (Francis and Shotton 1997). They have been strongly criticised for their arbitrary choice both of thresholds and probabilities at which the risk of such depletion is acceptable (Walters 1998b). They are included here as a simple means of additionally illustrating any differential impacts that might occur between substocks.

3. EFFECTIVE MANAGEMENT OF SPATIALLY COMPLEX FISHERIES REQUIRES GOOD INFORMATION AND FINE SCALE CONTROL [2]

3.1. Introduction

In the development of a new fishery, a manager is confronted with a set of decisions about which management controls might be applied, and which monitoring methods would be the most informative. Although these choices should be those that lead to optimum performance (typically this means sustainability as well as best economic and social benefit), they are constrained by economic and administrative feasibility, as well as political acceptability. In a new fishery, there may be little knowledge of the biology of the target species, or of other fisheries for it. At the same time, the small size of a fishery might limit many information-gathering activities, so that it may not be afforded management agency priority, particularly during early fishery development. Various operational characteristics of the fishery and attributes of the fished species also affect the feasibility and sampling power of different monitoring methods. Together, these constraints mean that there is a real danger that the fishery develops more rapidly than biological and operational knowledge can be acquired and applied in management (Perry *et al.* 1999).

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[2] Preliminary results of this chapter have been presented at a national symposium (Buckworth 2001) and an international conference (Buckworth and Martell 2003).

My aim in this chapter is to establish which of the choices of monitoring/ management sets a manager might make are safe, given various levels of prior information of the spatial stock dynamics of a fishery, and fleet characteristics (operating characteristics, search efficiency, cost structures and so on). I therefore specified a variety of spatial scenarios as inputs to the model developed in Chapter 2, to generate some of the complex interactions at different spatial and temporal scales that are observed in real fisheries. I provide an overview of the outputs of these simulations, and then examine some aspects of the simulated fisheries and outputs in detail.

I show that among the various fishery configurations produced, the best performance was in fisheries with the most informative monitoring, and provided that that monitoring information was accurate, both management controls simulated here (output control of TAC versus input control of fishing effort) led to similar performance. However, management failure was most likely to occur where stock spatial dynamics led to concentration of the stock: in such conditions, knowledge of stock status and ability to control effort became critical; the CPUE/TAC combinations tended to fail in these situations. These problems increased as capacity and hyperstability were increased. Poor knowledge of stock status and rapid change in fleet dynamics (through, for example, deployment of capacity, learning, and development and acquisition of new equipment) are most likely during the developmental phase so that, in a rapidly developing fishery, this is when management failure is most likely to occur. I show that many spatial dynamic effects are at fine spatial and temporal scales and suggest that use of information on these scales may improve management.

3.2. Methods

3.2.1 Simulations

Using the model framework described in Chapter 2, I simulated fisheries with a variety of spatial structures, with a series of monitoring/management control combinations and fleet operational characteristics, as well as recruitment variation. Fishery performance under these scenarios was then examined with the performance measures described in the previous chapter. Additional simulations and analyses were conducted to illustrate and examine observations and questions arising from the broader analysis.

For each spatial scenario, for each monitoring/ management control combination specification, I used high and low recruitment variation (u_r =0.2 or u_r =0.6), and an increasing range of capacity, R_c and catchability increase at low biomass (R_h) inputs (each ranged 1.0 – 4.0, in increments of 0.5). Each Monte Carlo simulation for each fishery configuration produced outputs of mean and standard deviation for each performance measure over 100 40-year simulations.

Monitoring methods and levels within these are indicated in Table 3.1. Levels within monitoring methods were chosen to supply a degree of effective monitoring, and that might be achievable for fisheries of the size specified (unfished biomass = 5000 tonnes), yet still illustrate the effects of different levels of investment in monitoring.

3.2.2 Spatial fishery scenarios

Ten basic spatial scenarios were employed (Table 3.2). These differed in the spatial dynamics and productivities of substocks, the number of administrations managing the fishery, and spatially-related gradients in the cost of fishing. Stock spatial dynamics were characterized by the number of substocks, the location of their nursery areas, and movements by each stock. Substock productivities were input as a reduction in the slope of their stock recruitment relationship (S-R R), relative to substock 1, as described in Chapter 2. In each fishery scenario there were either one or three statistical areas. Monitoring, assessment and management controls were applied on the basis of these statistical areas, regardless of stock structure.

Basic features

All spatial scenarios (Table 3.2) were built on fisheries composed of 12 rows by 2 columns of grid cells (Figure 3.1). Cells in the first column were treated as "inshore", while those in the second column were "offshore". This means that all directed movements of fish greater than one year old were in the offshore area; however, this stopped neither random movement components nor fishing in the inshore area. The nursery area for the first and fourth substocks was defined as the very top left grid cell (row 1, column 1) and those for the second and third substock nursery cells were (6,1) and (12,1), respectively (Figure 3.1). Although substocks 1 and 4 were equally productive, the slopes of the S-RRs for substocks 2 and 3 were respectively 80% and 60% of the slope of the S-RR for substock 1. Movement of fish from nursery areas was to the cell immediately offshore of the nursery cell. As indicated in Chapter 2, movement was controlled by several parameters. More extensive movement/ dispersal was effected by increasing the number of movement simulations per month ("Moves/ month" in Table 3.3). The amount of random movement was relative to the values entered for "Random move weighting",

while directed movement (migration) was relative to values for "Weight on directed movements" as per Table 3.3. Settings for these variables in each scenario are included as Table 3.3. All migratory substocks followed the route and timing shown in Figure 3.1b.

Table 3.1. Sampling levels within monitoring methods.

Monitoring method	Description of levels	Levels
Survey	CV of survey estimate	A_s = 0.2,0.4,0.6
CPUE	Spatial or aggregate	
Mean weight	Number of fish weighed	100, 2000, 5000
Mark-recapture for abundance estimation and stock assessment	Number of tagged fish released	2000, 5000
Mark recapture for F estimation and tracking catchability	Number of tagged fish released	500, 1000

Table 3.2. General features of spatial fishery scenarios

Scenario	General description	Sub-stocks	Stat. areas	Fishing cost
1	Unit stock fishery, Pool	1	1	Homogeneous
2	Single migratory stock	1	1	Minimum at spawning ground
3	Single migratory stock	1	3	As for Scenario 2
4	Single migratory stock	1	3	Strong gradient with maximum at spawning ground
5	"metastock" –multiple substocks low movement levels,	3	3	Gradient with 'weaker' stocks cheaper to fish.
6	"metastock" –as Scenario 5 in a single statistical areas	3	1	As for Scenario 5.
7	A salmon-like scenario, with multiple substocks, a high rate of diffusion and mixing, over multiple statistical areas	3	3	As for Scenario 5.
8	As with Scenario 7, with a moderate rate of diffusion and mixing	3	1	As for Scenario 5.
9	A mixed 'metastock': 3 sedentary substocks and a migratory substock	4	3	As for Scenario 5.
10	A mixed 'metastock': The same as Scenario 9 but with 3 statistical areas.	4	1	As for Scenario 5.

Management in spatial scenarios comprised either one or three statistical areas (Figure 3.1a). These did not necessarily match the spatial stock structure. Each statistical area thus corresponded to a separate political unit (~state, province) or statistical management area within such a unit. A management agency might, for example, use such a device to prevent all capacity in a fishery being deployed in just one area, or in an attempt to match biological and management spatial structures. However, statistical divisions here did not constrain the distribution of fishing effort except in the limits imposed by the regulation factor in the effort or TAC controls. Within any input configuration, monitoring method, level and management control did not differ between statistical areas.

Inshore   Offshore

1,4				Jan	
				Feb Dec	
Boundary, statistical areas 1, 2				Mar Nov	
2				Apr Oct	
Boundary, statistical areas 2, 3				May Sep	
				Jun Aug	
3		**a**		Jul	**b**
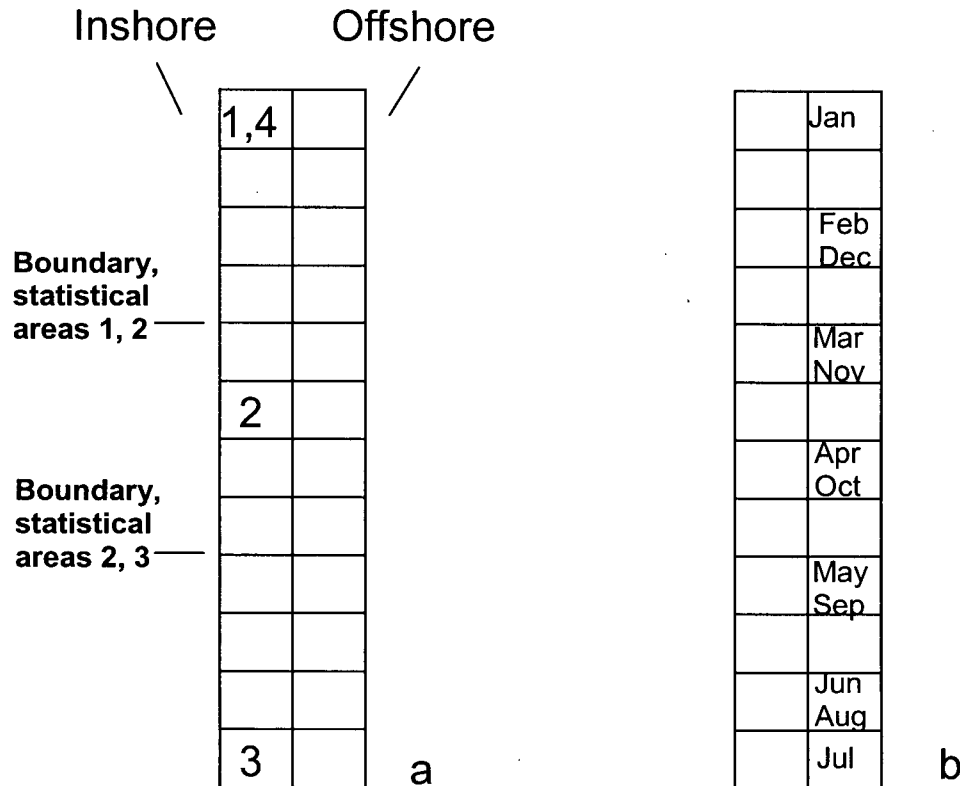
Figure 3.1. Basic maps for spatial fishery scenarios, indicating **a**, Nursery cells for each substock, designated by numbers; and **b**, annual migration route for migratory substocks, with "target cells" as described in the movement model description, indicated by the month in which each was the target call.

Table 3.3. Movement variables for spatial scenarios

Scenario	Moves/ month	Random move weighting	Weight on directed movement (i, j directions)	Annual migration
1	1	20	5,5	no
2	1	20	5,5	yes
3	1	20	5,5	yes
4	1	20	5,5	yes
5	2	20	5,5	no
6	1	20	5,5	no
7	2	50	5,5	no
8	1	20	5,5	no
9	1	20	5,5	Substock 4 only
10	1	20	5,5	Substock 4 only
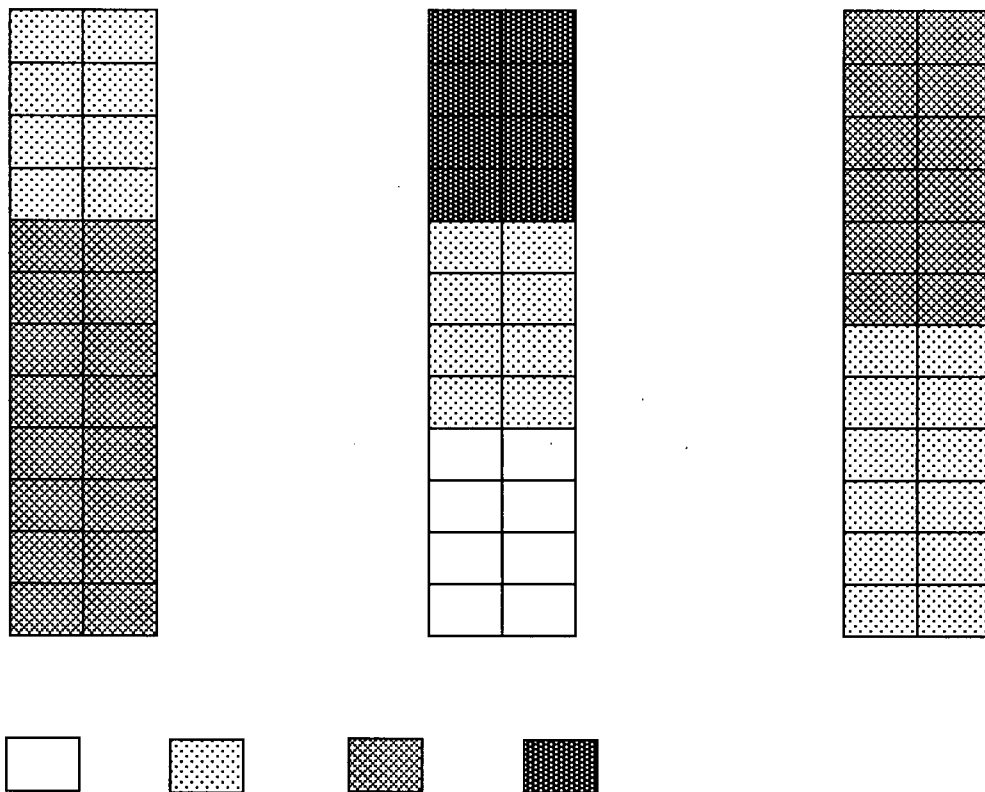
Figure 3.2. Cost structures for spatial scenarios, **a**, Scenarios 2, 3; **b**, Scenario 4; **c**, Scenarios 5-10.

Scenario 1 was a single substock within one statistical area, corresponding roughly to the single pool assumptions that underlie unit stock monitoring and assessment. Costs were uniform throughout the fishing area. However, note that, an origin in a single cell nursery area, and only moderate dispersal, created a strong density gradient, from high levels in the first row of cells, to a minimum in the twelfth row.

The structure of Scenario 2 fishery differed from Scenario 1 in that there was an annual migration to and from the spawning ground (Figure 3.1b). Additionally, there was a gradient in the cost of fishing, so that the cost of fishing away from the spawning ground was higher than normal (Figure 3.2a). The Scenario 3 fishery differed from Scenario 2 only in that there were 3

statistical areas. Scenario 4 added a strong gradient in fishing cost to Scenario 3, so that the grids near the spawning ground cells were twice as expensive to fish as those distant from them.

Scenario 5 represented a metapopulation structure with 3 distinct substocks, each centred in a statistical area (Figure 3.1.a). Management and stock spatial structure were thus approximately matched. Note that there was higher dispersal than previous scenarios (Table 3.3), and a gradient in fishing cost was such that it was cheaper to fish the weaker substocks, 2 and 3 (Figure 3.2.b). Scenario 6 was identical to Scenario 5 excepting dispersal was reduced (Table 3.3), and that there was only one statistical area Figure 3.1.a. Representing a benign fishery, in which there was reasonable mixing between the substocks and division into three statistical areas roughly corresponding to the substock, the cost gradient nevertheless also drove the Scenario 7 fishery to concentrate more on the weaker substocks. Although similar to Scenario 7, the Scenario 8 fishery had lower dispersal (Table 3.3), and was managed as single statistical area (Figure 3.1a).
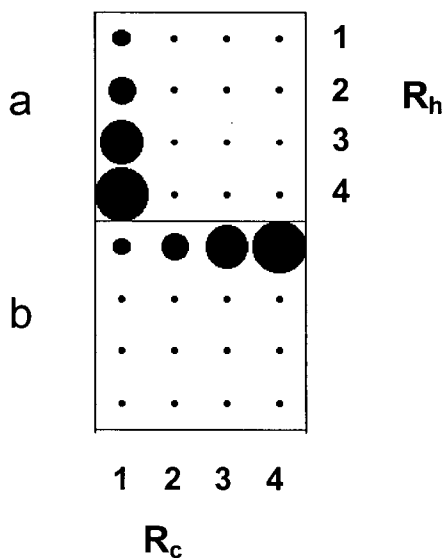
Figure 3.3. Method of presentation of outputs of Monte Carlo simulations for large numbers of comparisons. Each square in the figure contains 16 dots, the position of which indicates capacity R_c and growth in catchability, R_h. **a,** The series of increasingly large dots represents a series of increasing values of a performance measure as R_h increases. **b,** The series of increasingly large dots represents a series of increasing values of a performance measure as R_c increases.

Scenarios 9 and 10 represented the metapopulation structure described for Scenario 5, with an additional substock showing strong annual migration (Figure 3.1.b). Again, the cost of fishing ensured that fishing concentrated upon the weaker stocks (Figure 3.2). Scenario 9 was a fishery managed in three statistical areas, while in Scenario 10 was subject to control in just one statistical area (Figure 3.1.a).

3.2.3 Visualisation of general results

The output of Monte Carlo simulations for any fishery configuration was a set of mean and SD of performance measures. Given the wide number of comparisons attempted in this study – across spatial scenarios, monitoring methods, management controls, recruitment variation, capacities and growths in catchability – a pictorial method of presenting results was decided upon. For these presentations, it was necessary first to reduce the set of simulation outputs to include only those for integer values of capacities and growths in catchability, R_c and R_h. This meant that a combination of results for fishery configurations differing only in R_c and R_h could be presented as a 4 X 4 square, with R_c increasing horizontally and R_h increasing vertically (Figure 3.3). The size of symbols could then be used to signify values of the mean of the performance measure for that configuration. An example for two fishery configurations is presented as two rectangles in Figure 3.3.

3.3. Results

3.3.1 General results of fishery simulations

To provide broad comparison among factors in fishery performance, results of the simulations are summarized in Figure 3.4 to Figure 3.9, which present the results of the Monte Carlo simulations in terms of various performance measures. The dots in these figures represent the means of performance measures for sets of 100 simulations, as described above, with each square cell representing the range of R_c and R_h entered. Each spatial scenario is represented by a pair of columns, with the left hand column of the pair being those managed by the TAC control, and with those on the right being managed by the effort control. The horizontal divisions represent different monitoring methods (see key to Figure 3.4), with differing levels of precision as indicated.

In these figures, the size of the dots is proportional to the mean value for the performance measure, and the colour shade represents the standard deviation among the values in the set, all relative to the largest mean and standard deviation values for that performance measure. Large circles represent poor performance, and dark circles indicate high variability in that set; small, light-shaded circles represent good performance and low variability.
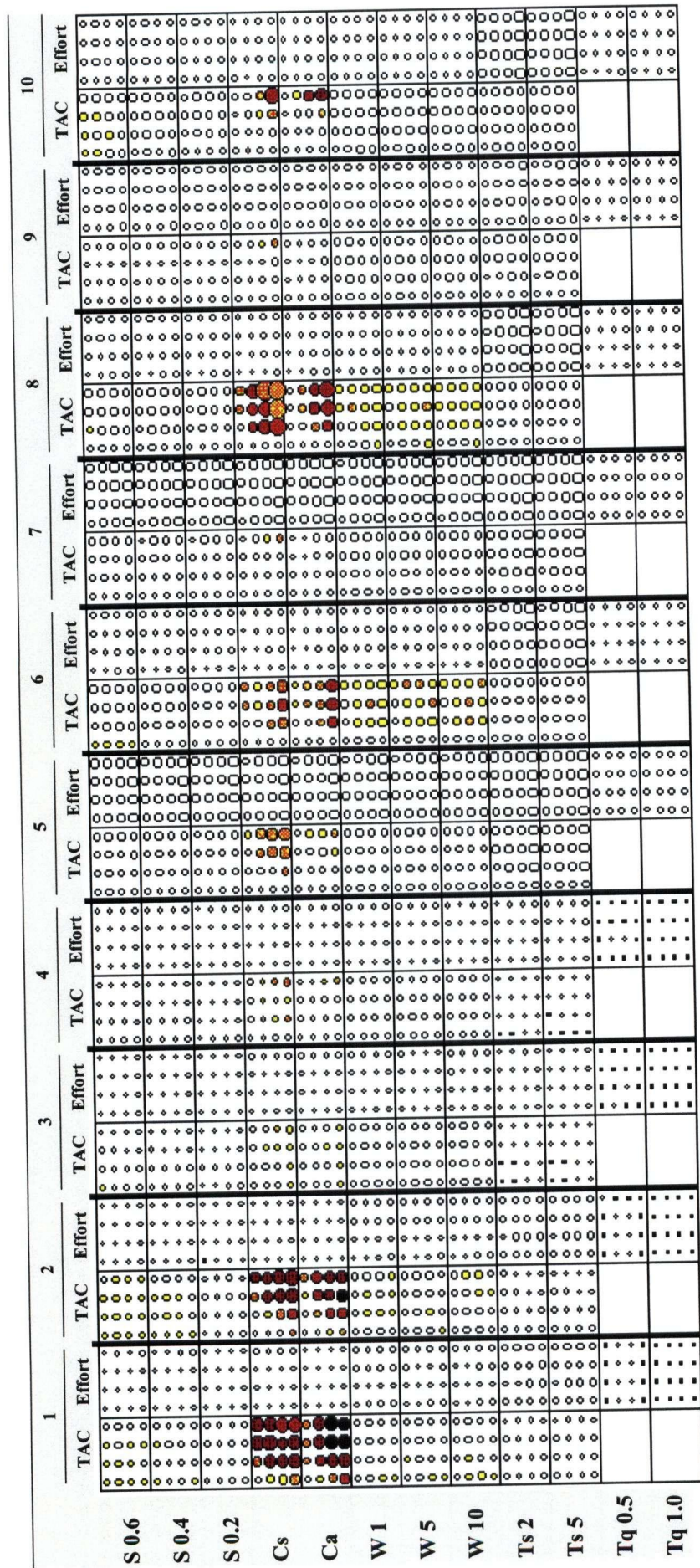
Figure 3.4. Relative performance of fishery configurations, by the L_{rel} measure. Recruitment variation $u_r = 0.2$.

SD: □, 0 -10%; ▨, 11 - 20%; ▨, 21 - 30%; ▨, 31 - 40%; ▨, 41 - 50%; ▨, 51 - 60%; ▨, 61 - 70%; ▨, 71 - 80%; ▨, 81 - 90%; ▨, 91 - 100%;

Topbar indicates spatial scenario, the sidebar indicates monitoring methods, with **S** = survey, with **CV** of 0.6, 0.4 or 0.2; **Cs** = spatial CPUE; **Ca** = aggregate **CPUE**; **W** = mean weight, for samples of 1000, 5000 or 10000 fish; **Ts** = tagging for stock assessment, with 2000 or 5000 tags released; **Tq** = tagging for catchability estimation, with 500 or 1000 tags released. For detailed description of figure, see text

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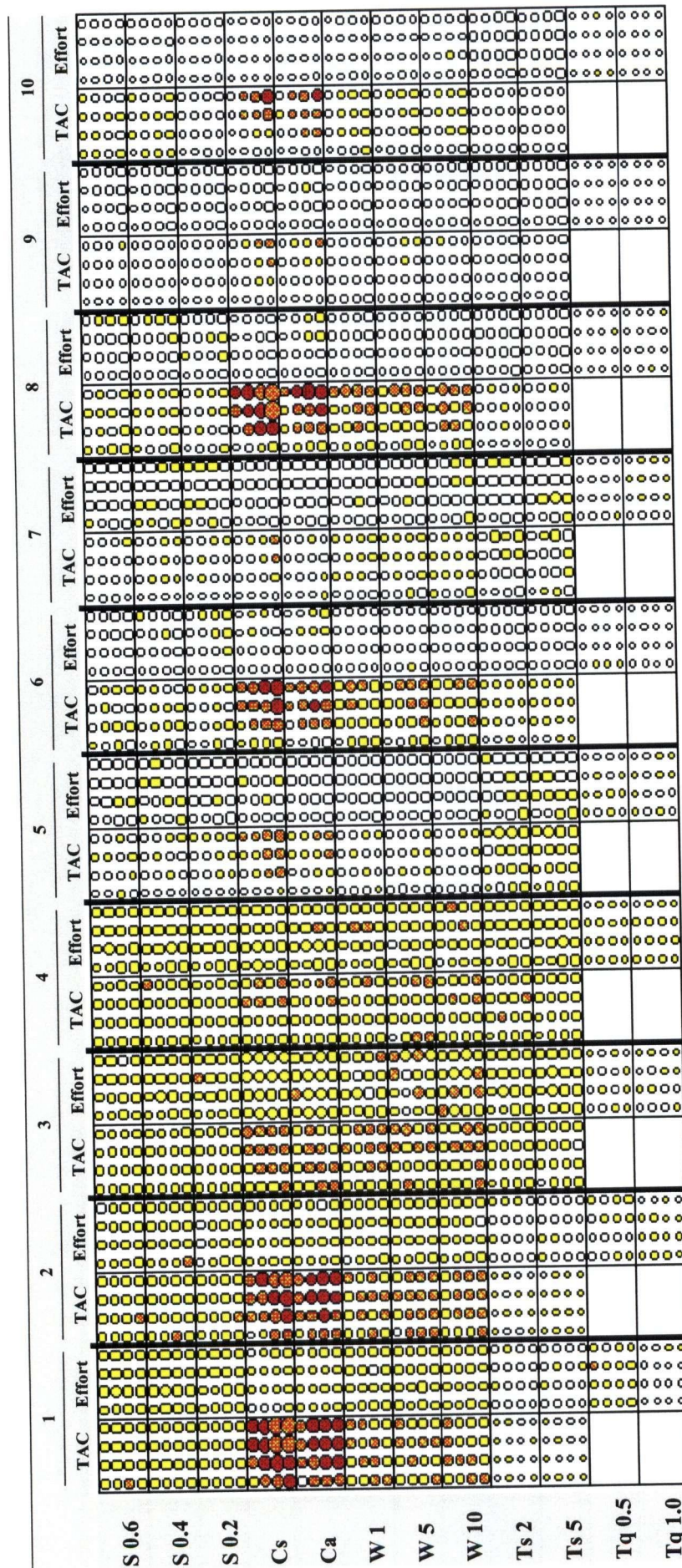
Figure 3.5. Relative performance of fishery configurations, by the L_{rel} measure. Recruitment variation $u_r = 0.6$. For detailed description of figure, see text ☐, 0 -10%; ▨, 11 - 20%; ▨, 21 - 30%; ▨, 31 - 40%; ▨, 41 - 50%; ▨, 51 - 60%; ▨, 61 - 70%; ▨, 71 - 80%; ▨, 81 - 90%; ■, 91 - 100%;

Figure labelling is as for Figure 3.4.

Figure 3.6. Mean number of years during which substock 2 was depleted. Recruitment variation $u_r = 0.2$. For detailed description of figure, see text

☐, 0 - 10%; ☐, 11 - 20%; ☐, 21 - 30%; ☐, 31 - 40%; ☐, 41 - 50%; ☐, 51 - 60%; ☐, 61 - 70%; ☐, 71 - 80%; ☐, 81 - 90%; ■, 91 - 100%;

Figure labelling is as for Figure 3.4.

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Figure 3.7. Mean number of years during which substock 2 was depleted. Recruitment variation $u_r = 0.6$. For detailed description of figure, see text

Legend: □, 0 -10%; ▨ (yellow), 11 - 20%; ▨ (orange), 21 - 30%; ■, 31 - 40%; ■, 41 - 50%; ■, 51 - 60%; ■, 61 - 70%; ■, 71 - 80%; ■, 81 - 90%; ■, 91 - 100%;

Figure labelling is as for Figure 3.4.

80
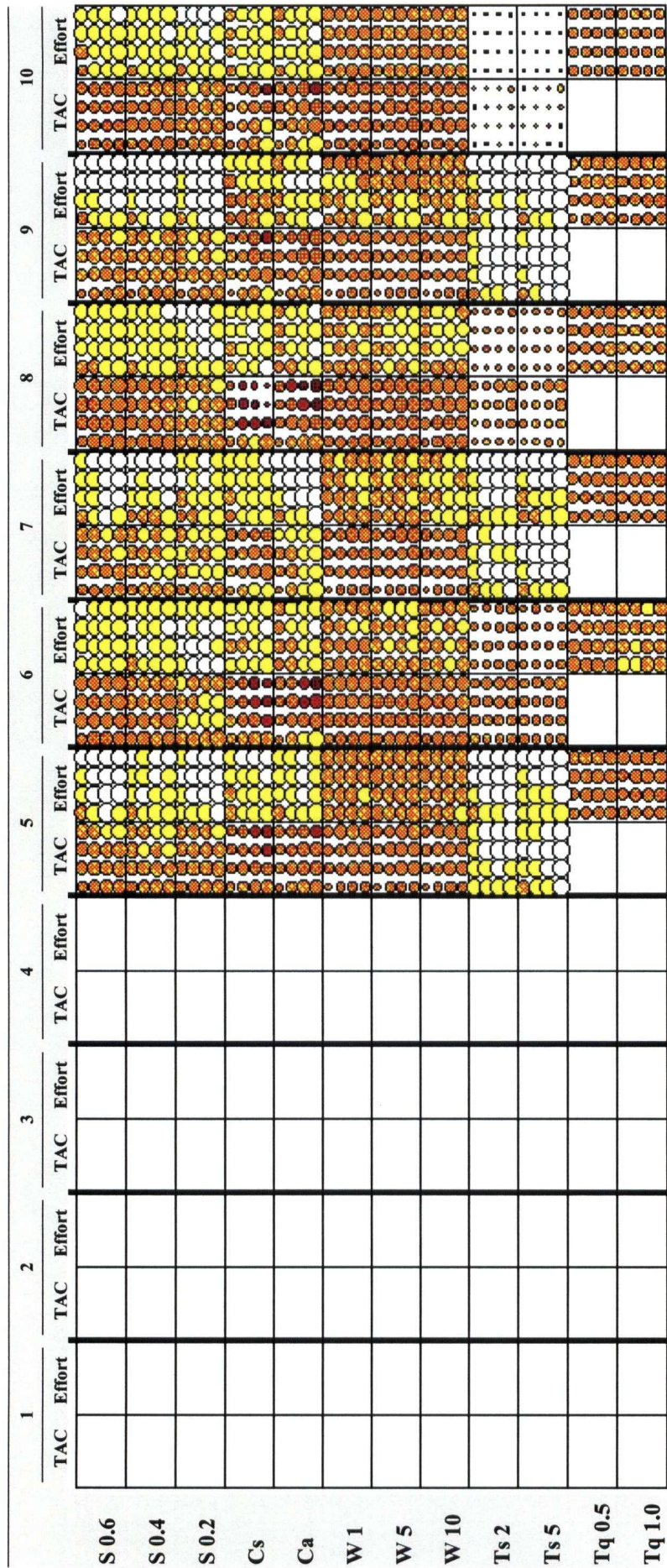
Figure 3.8. Mean number of years during which substock 3 was depleted. Recruitment variation $u_r = 0.6$. For detailed description of figure, see text

Figure labelling is as for Figure 3.4.

Legend: □, 0 -10%; ▨, 11 - 20%; ▨, 21 - 30%; ▨, 31 - 40%; ▨, 41 - 50%; ▨, 51 - 60%; ▨, 61 - 70%; ▨, 71 - 80%; ▨, 81 - 90%; ■, 91 - 100%;
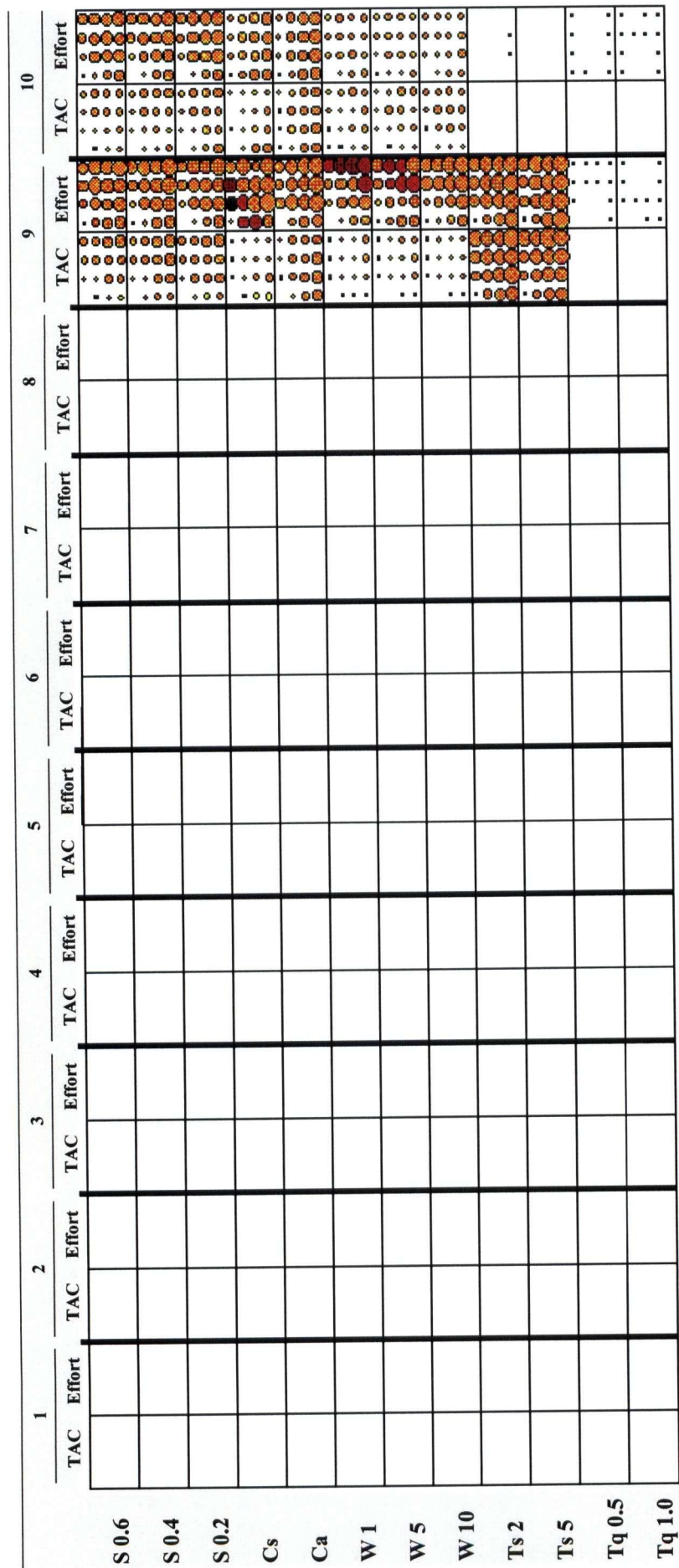
Figure 3.9. Mean number of years during which substock 4 was depleted. Recruitment variation ur = 0.6. For detailed description of figure, see text

☐, 0 - 10%; ▨, 11 - 20%; ▨, 21 - 30%; ▨, 31 - 40%; ▨, 41 - 50%; ▨, 51 - 60%; ▨, 61 - 70%; ▨, 71 - 80%; ▨, 81 - 90%; ▨, 91 - 100%;

Figure labelling is as for Figure 3.4.

Impact of different fishery configurations on loss functions

Both loss functions L_{abs} and L_{rel} were calculated for all fishery configurations, but showed minimal qualitative difference. L_{rel} was therefore chosen for summary presentation and any further analyses.

Performance deteriorated when recruitment variation increased, except in scenarios for which management control was implemented by direct monitoring and regulation of exploitation rate without reference to changes in total stock size. For the low recruitment variability ($u_r = 0.2$) configurations, the loss measure L_{rel} varied between 3.86, in Scenario 2, and 29.46 (Scenario 1), with an overall mean of 10.78. Larger loss values were generated in the corresponding high recruitment configurations, with L_{rel} ranging between 7.66 (Scenario 7) and 33.76, again in Scenario 1, and with an overall mean of 15.60, for $u_r = 0.6$ configurations.

Comparison of Figure 3.4 with Figure 3.5 also supports the general conclusion that the performance of most monitoring/ management control combinations deteriorated as the variation in recruitment was increased, especially among single stock fisheries (Scenarios 1 to 4), and is emphasised in Figure 3.10. In all scenarios, average performance was superior when mark-recapture was used to track catchability, with subsequent effort control (hereafter, "Tq" configurations) (Figure 3.4, Figure 3.5, Figure 3.10). For the single stock fisheries (Scenarios 1 to 4), L_{rel} ranged 4.67 to 5.09 for low recruitment variation but increased to 12.41 – 13.47 for $u_r = 0.6$. At $u_r = 0.2$, L_{rel} for multi-stock fisheries (Scenarios 5-10) was typically higher (L_{rel}: 4.81-10.77) than for the single stock fisheries but increased less with higher recruitment variation (L_{rel}: 8.02-12.37), so that at high recruitment variation, multi-stock fisheries generally performed better than single stocks (Figure 3.6).
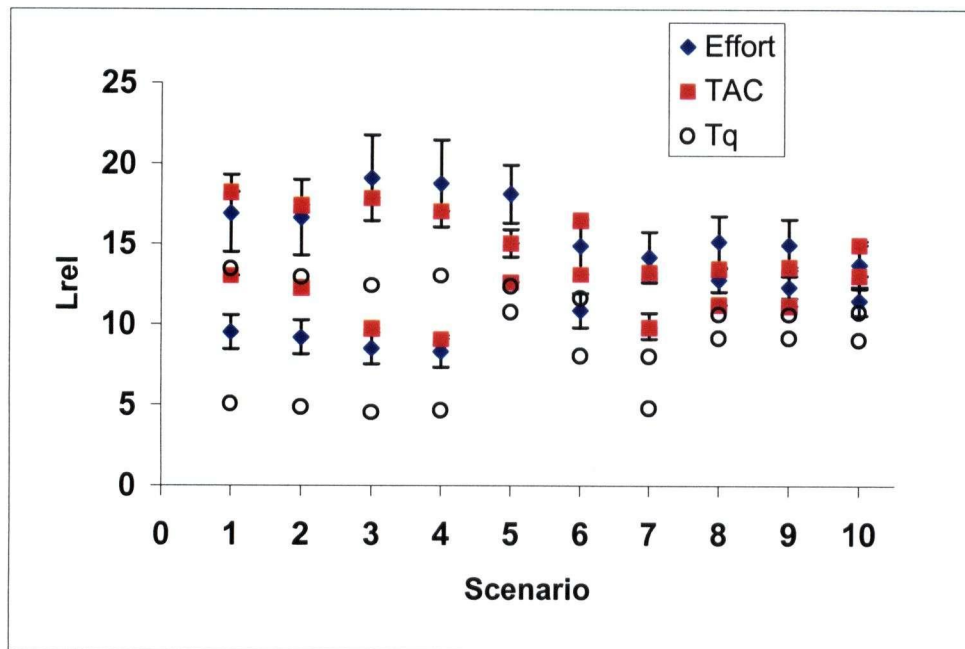
Figure 3.10. Mean of L_{rel} over spatial fishery scenarios, broken down by recruitment variation and general monitoring/management control. Effort and TAC signify fisheries monitored for abundance then subject to that control; Tq signifies fisheries monitored by mark-recapture for direct harvest rate and catchability estimation. The higher values of L_{rel} for any configuration are for $u_r = 0.6$. Error bars are means of SD values corresponding to the L_{rel} means. For clarity of presentation, small errors for Tq are not shown.

For fisheries where abundance was monitored, with subsequent stock assessment, there was no strong difference in performance between TAC-controlled and effort-controlled fisheries (Figure 3.10). For the single stock configurations (Scenarios 1-4), management generally performed better with effort control (L_{rel}: 8.31-9.50) than the TAC control (L_{rel}: 9.08-13.1), under low recruitment variation. However, under high recruitment variation, this was reversed, so that single stocks were on average better-managed with the TAC control (L_{rel} 13.24-18.24) than the effort control (16.65-19.11). In multiple stocks (scenarios 5-10) the deterioration in average performance as recruitment variation was increased was less marked; the relative performance of the monitoring controls was specific to the spatial scenario (Figure 3.4, Figure 3.5, Figure 3.10).

There was no clear trend in performance depending on whether management and spatial stock structure corresponded. Scenario 2, in which the single stock was managed in a single statistical area, corresponded to Scenario 3, managed as three distinct areas. For management using abundance indices, there was some relative deterioration in performance from Scenario 2 to Scenario 3 for high recruitment variation, but this trend was reversed for $u_r = 0.2$ and for the Tq configurations (Figure 3.10). Fisheries in Scenario 7, managed within three statistical areas, performed a little better than fisheries in Scenarios 6 and 8, which were similar in terms of stock spatial structure (3 substocks), but managed as single statistical areas. Average performance of fisheries in Scenarios 9 and 10 (4 substocks and, respectively, 3 and 1 statistical areas) was similar (Figure 3.10).

The effects of capacity and hyperstability on L_{rel}

The performance of the Tq configurations, averaged across scenarios, varied little with the input capacity, R_c, and growth in catchability, R_h (Figure 3.4 and Figure 3.5). For low recruitment variation ($u_r = 0.2$), the means of L_{rel} across scenarios varied between 6.85 (mean for $R_c = 1$ and $R_h = 1$) and 7.06 ($R_c = 3.5$ and $R_h = 3.5$). For $u_r = 0.6$, the variation was also trivial, between 11.51 ($R_c = 1.5$ and $R_h = 4.0$) and 11.85 ($R_c = 3.5$ and $R_h = 2.0$). This contrasted strongly with the other monitoring/ management control combinations, for which performance typically deteriorated as capacity and hyperstability increased.

For the effort control (excluding Tq), L_{rel} means across scenarios increased with R_c and R_h, between 8.83 and 11.29 for $u_r = 0.2$, and between 13.84 and 16.97, for $u_r = 0.6$. For the TAC control, L_{rel} also increased with R_c and R_h, between 9.72 and 13.12 ($u_r = 0.2$) and between 13.91 and 17.32 for high recruitment variation. However, as strongly indicated in Figure 3.4 and Figure 3.5, performance relative to capacity and hyperstability was very dependent upon the

monitoring method/ control combination and spatial scenario. I therefore examine the performance of the remaining monitoring/ control combinations in detail.

The mean performance of monitoring by the abundance index from survey is summarized in Table 3.4. Mean performance (L_{rel}) and its variation improved slightly with survey precision, but deteriorated substantially with increased recruitment variation, and was marginally poorer, on average, for the effort control (Table 3.4).

Increasing capacity, R_c and catchability growth, R_h increased the mean across scenarios of L_{rel} for the survey monitoring method with the increase slightly more pronounced for the effort control (Figure 3.11a, b). Thus, for example, the mean L_{rel} increased from 13.30 to 17.40 for the effort control (for survey with CV=0.6, u_r = 0.6) and for TAC, from 13.84 to 15.98, as R_c and R_h each increased between 1.0 and 4.0.

Table 3.4. Performance (mean L_{rel} values) for fishery configurations monitored with the survey abundance index, by control method, recruitment variation (u_r) and survey CV.

Control	u_r	Survey CV	Mean L_{rel}	Mean L_{rel} SD
Effort	0.2	0.2	9.74	0.89
Effort	0.2	0.4	10.04	0.95
Effort	0.2	0.6	10.16	1.02
Effort	0.6	0.2	16.51	2.07
Effort	0.6	0.4	16.55	2.11
Effort	0.6	0.6	16.55	2.06
TAC	0.2	0.2	9.71	1.14
TAC	0.2	0.4	10.52	1.29
TAC	0.2	0.6	11.24	1.53
TAC	0.6	0.2	15.11	2.22
TAC	0.6	0.4	15.47	2.25
TAC	0.6	0.6	15.77	2.26

The most striking feature of Figure 3.4 and Figure 3.5 is that the performance of fishery configurations with CPUE indices for monitoring abundance varied strongly between spatial scenarios, management controls and with capacity and the degree of hyperstability. In particular the fisheries with single statistical areas, Scenarios 1, 2, 6, 8 and 10, showed particularly strong growth in the loss function (= poor performance) with increasing capacity and hyperstability. As for other monitoring methods, L_{rel} for both CPUE methods increased substantially, by about 50-80%, with u_r (Table 3.5). Mean performance did not differ much between spatial and aggregate CPUE, within a control, especially considering the large variability in performance indicated by the mean SD values. However, effort control did perform on average better than the TAC control – thus the mean of L_{rel} across scenarios, for $u_r = 0.6$ for the effort control with SCPUE was 15.74, while the corresponding TAC value was 18.72 (Table 3.5).

Table 3.5. Performance (mean L_{rel} values) for fishery configurations monitored with the CPUE abundance indices, by control method, and recruitment variation (u_r).

Control	u_r	CPUE	Mean L_{rel}	Mean L_{rel} SD
Effort	0.2	Spatial	9.78	0.95
Effort	0.2	Aggregate	9.77	0.94
Effort	0.6	Spatial	15.74	2.03
Effort	0.6	Aggregate	15.94	2.02
TAC	0.2	Spatial	14.22	3.99
TAC	0.2	Aggregate	12.75	3.66
TAC	0.6	Spatial	18.72	4.83
TAC	0.6	Aggregate	17.32	4.57

The performance of CPUE as a monitoring method deteriorated severely with increasing capacity, R_c and catchability growth R_h (Figure 3.11c, d), although this was more marked in some scenarios than others (for example, comparing Scenarios 1 and 2 to 3 and 4, Figure 3.4 and Figure 3.5). The increase in L_{rel} was greater when in combination with the TAC control. For example, the mean L_{rel} for ACPUE and high recruitment variation increased from 13.69 to 16.72, as R_c and R_h each increased between 1.0 and 4.0 – comparable performance to the survey monitoring above – but the increase for the TAC control was from 14.28 to 21.89 (Figure 3.11c, d).

Mean weight used as an index of abundance performed comparably to the survey index. As for the other abundance indices, performance deteriorated when recruitment variation increased from u_r = 0.2 to u_r = 0.6. For example, the mean L_{rel} for the effort control, for sample size of 1000 fish, increased from 10.39 to 15.95, an increase of around 50%. The most notable aspect of Table 3.6 is that increasing sample size had minimal impact on mean performance.

Table 3.6. Performance (mean L_{rel} values) for fishery configurations monitored with the mean weight abundance index, by control method, recruitment variation (u_r) and the number of fish in the weight sample (Sample N).

Control	u_r	Sample N	Mean L_{rel}	Mean L_{rel} SD
Effort	0.2	1000	10.39	0.99
Effort	0.2	5000	10.24	0.97
Effort	0.2	10000	10.31	0.98
Effort	0.6	1000	15.95	1.97
Effort	0.6	5000	16.03	1.95
Effort	0.6	10000	16.15	2.00
TAC	0.2	1000	12.55	1.64
TAC	0.2	5000	12.47	1.62
TAC	0.2	10000	12.48	1.61
TAC	0.6	1000	16.39	2.67
TAC	0.6	5000	16.38	2.64
TAC	0.6	10000	16.49	2.68

As with the other methods for monitoring abundance, the performance of mean weight monitoring deteriorated somewhat with increasing capacity, R_c and catchability growth R_h (Figure 3.11e, f). The increase in L_{rel} was of similar magnitude as for the survey method, differing little between the controls. Thus the mean L_{rel} for mean weight monitoring, under high recruitment variation across scenarios increased from 13.99 to 16.38 for the effort control, as R_c and R_h each increased between 1.0 and 4.0, while the increase for the TAC control was from 14.68 to 16.4 (Figure 3.11e, f).

Mark-recapture to monitor abundance for stock assessment produced generally effective performance across scenarios (Figure 3.4 and Figure 3.5), comparable or superior, for some scenarios, to the survey and mean weight monitoring. However, it was not as effective in terms of L_{rel} as direct tracking of catchability by mark recapture (Figure 3.4 and Figure 3.5). The performance of the monitoring methods is summarized in Table 3.7. Mean of performance (L_{rel}) improved minimally with the release of more tags, but deteriorated substantially with increased recruitment variation. Performance was better, on average, for the TAC control (Table 3.7).

Table 3.7. Performance (mean L_{rel} values) for fishery configurations monitored with mark-recapture to estimate abundance (Ts), by control method, recruitment variation (u_r) and the number of tags released, $tags_r$.

Control	u_r	$tags_r$	Mean L_{rel}	Mean L_{rel} SD
Effort	0.2	2000	12.91	0.89
Effort	0.2	5000	12.97	0.93
Effort	0.6	2000	16.68	1.89
Effort	0.6	5000	16.69	1.92
TAC	0.2	2000	11.01	0.80
TAC	0.2	5000	11.01	0.80
TAC	0.6	2000	14.68	2.08
TAC	0.6	5000	14.69	2.11

This was also the case when L_{rel} was related to increasing capacity and catchability growth. The performance of mark-recapture monitoring coupled with the TAC control deteriorated, with the mean across scenarios of L_{rel} increasing from 12.63 to 15.52 as R_c and R_h each increased from 1.0 to 4.0. The increase for the effort control was 14.45 to 17.48 (Figure 3.11g, h).
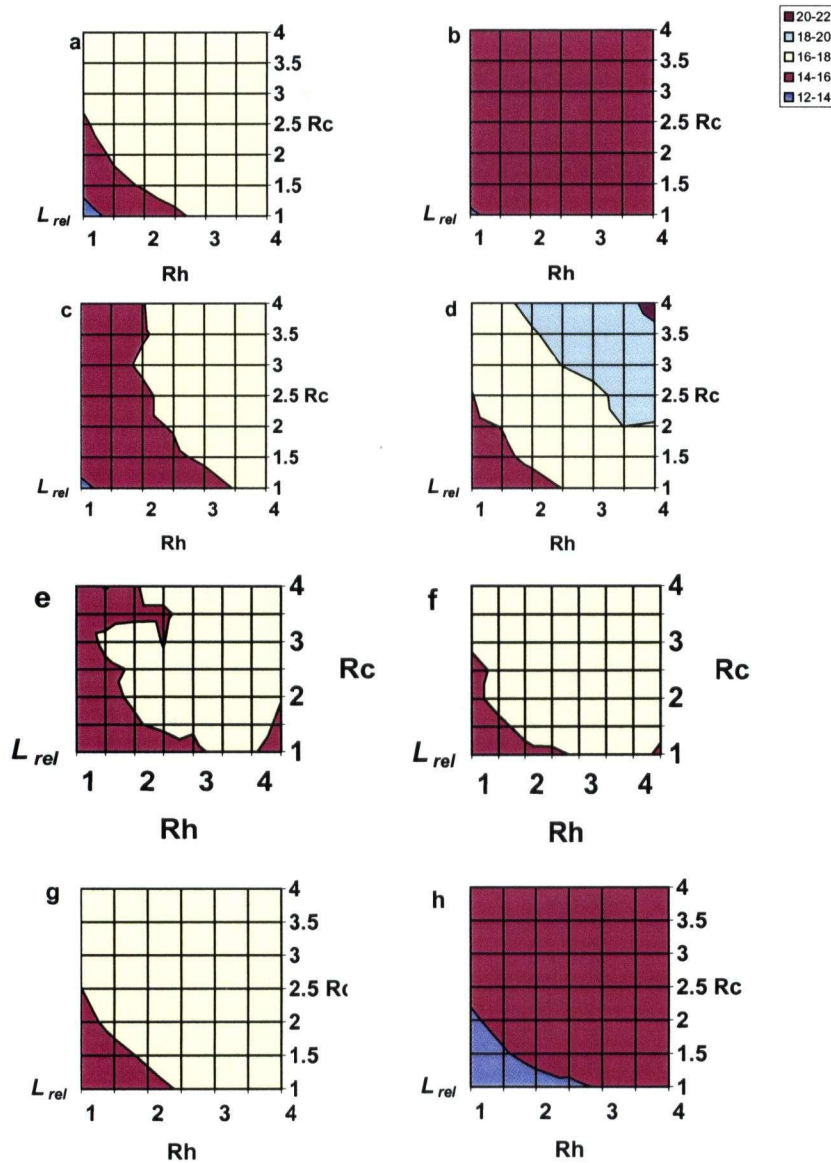
Figure 3.11. Relative loss L_{rel} versus capacity R_c and hyperstability R_h for various monitoring and control combinations, all for high recruitment variation **a**, survey, CV=0.6 Effort control; **b**, survey, CV=0.6, TAC; **c**, ACPUE, Effort; **d**, ACPUE, TAC; **e**, Mean weight, Effort; :**f,** Mean weight, TAC; **g**, Ts, Effort; and **h**, Ts, TAC.

Depletion: reduction of substocks below 20% of unfished biomass

Under this measure for the first substock, there was some contrast between high and low recruitment variation scenarios. The low variation configurations had a very low incidence of decline of substock 1 to less than 20 percent of unfished biomass (hereafter I term this "depletion"). For u_r = 0.2, the mean incidence across configurations was 0.27 years out of the 40 year time horizon (with mean of the SD being 0.53). For the higher recruitment variation configurations (u_r = 0.6), the mean across configurations was 6.05 years (with mean of SDs being 4.18). For the abundance-monitoring sets of configurations, for u_r = 0.6, the mean number of depletions was between 5.50, for mean weight monitoring and 7.78 for survey monitoring. Fisheries monitored and managed with the Tq monitoring/ effort control combination were rarely pushed below 20 percent of unfished biomass, with a mean of just 1.70 years in a depleted state.

For the second substock, with a relative reduction in recruitment productivity to 0.8, contrast between low and high recruitment variation scenarios was not marked (Figure 3.6 and Figure 3.7), means increasing only from 14.8 to 18.2 years (calculated across multi-substock configurations). As in the case of the first substock, the Tq monitoring was more effective at preventing depletion of the substock (mean of depletions for u_r = 0.6 was 11.07 years). Depletions under the remaining monitoring methods varied up to 22.60 for survey monitoring (over the high recruitment variation configurations). Variability was also high (the SD of this measure ranged from 5.98 for Tq to 7.18 for mean weight monitoring), and performance by this measure was largely dependent upon the spatial scenario. Results for the fisheries with Ts monitoring in Scenarios 6, 8 and 10 (fisheries managed as single statistical areas) contrasted with results for Scenarios 5,7, and 9 (multiple statistical areas) (Figure 3.7). Worthy of note

also in Figure 3.7 is the decline in incidence of deletions of substock 2 with increasing capacity and catchability growth in Scenarios 6 and 8 for the ACPUE/TAC combination. Under other monitoring measures, the main contrast was between management controls, with TAC control better than effort control, by this measure. CPUE and weight monitoring outperformed survey monitoring (Table 3.8).

Table 3.8. Mean number of years for which substocks were depleted, by monitoring method and recruitment variation.

u_r	Monitoring method	Mean #2	Mean SD #2	Mean #3	Mean SD #3	Mean #4	Mean SD #4
0.2	Survey	22.00	4.92	25.84	2.98	0.13	0.44
0.2	SCPUE	15.74	4.72	22.65	3.34	0.19	0.47
0.2	ACPUE	17.49	4.62	23.43	3.34	0.22	0.55
0.2	Mean Weight	11.82	5.30	21.90	4.33	0.11	0.33
0.2	Ts	12.52	2.90	19.01	1.43	0.33	0.63
0.2	Tq	2.76	1.62	18.74	3.65	0.00	0.00
0.6	Survey	22.60	6.08	27.29	4.39	7.29	5.19
0.6	SCPUE	17.94	6.67	23.41	5.67	6.78	4.97
0.6	ACPUE	19.37	6.37	24.58	5.50	7.43	4.91
0.6	Mean Weight	17.22	7.18	23.55	5.98	5.48	4.66
0.6	Ts	16.43	5.62	23.69	4.18	6.10	3.41
0.6	Tq	11.07	5.98	19.83	5.31	1.08	1.92

With a lower productivity (relative reduction in the slope of the S-RR was 0.6), the third substock suffered a generally higher incidence of depletion than substocks 1 and 2. This did not differ substantially between the low and high recruitment variation configurations, the depletion rate for the third substock across the multi-stock configurations for $u_r = 0.2$ was 22.4, the rate for $u_r = 0.6$ was 24.3 (Table 3.8). The pattern of rates of depletion was similar to that of the second substock. Again, Tq monitoring was the most effective in preventing depletion (with a rate of 19.83 for the high recruitment variation). As indicated in Table 3.8, the rate for other monitoring/management combinations was 27.29. Ts monitoring for single management area

fisheries was markedly more effective, again, than for multiple area fisheries; in multiple statistical area fisheries, depletion under Ts was relatively frequent, but was less variable than for other abundance monitoring scenarios (Figure 3.8). Again evident was the decline in incidence of depletions of substock with increasing capacity and catchability growth in Scenarios 6 and 8 for the ACPUE / TAC combination (Figure 3.8).

With a productivity the same as the first substock, the fourth substock was depleted much less frequently than substocks 2 and 3 – at a negligible rate for the low recruitment variation configurations (0.16), and a similar rate to substock 1, 5.98, for the high recruitment variation configurations.

Tq monitoring was particularly effective with a rate of just 1.08 for u_r = 0.6. Again for Ts, there was strong contrast between the performance of single area management, with few incidents of strong stock declines in the single management area scenario (Scenario 10), but high and variable incidence under multiple area management, in Scenario 9 (Figure 3.9). For the mean weight monitoring, there was a smaller difference between single and multiple area management (Figure 3.9). The effort control for survey and CPUE monitoring method performed more poorly under this criterion than did TAC control (Figure 3.9).

3.3.2 The combined effects of fish stock concentration, estimation error and implementation error.

In this section I examine in detail some aspects of the simulated fisheries and the model outputs. Considering that the most basic fishery of the simulated set, Scenario 1, conforms most closely to the archetypical "pool" fishery – a single stock, managed as a "unit" stock - it is of particular interest to examine why the performance of this fishery was so poor under the CPUE/ TAC

control combinations. Performance of fisheries monitored by different methods varied widely (Table 3.9; mean L_{rel} for $u_r = 0.2$ was smaller but showed a similar pattern among the monitoring/control combinations). The value of mean L_{rel} increased from 11.78 for the Ts/TAC combination, to 27.96, for the SCPUE/TAC combination.

High catches ($>>Catch_{opt}$) were frequently observed early in the time series of the simulations with CPUE monitoring/ TAC control combinations, particularly with input values of high R_c (high capacity) and R_h (hyperstable). For high capacity scenarios, assessment errors early in the time series meant that substantial reduction in the stock could occur even during the development period. This was exacerbated by implementation error (ie effort taking greater

Table 3.9. Mean of L_{rel} by monitoring method and management control, for the Scenario 1 fishery, for high recruitment variation ($u_r = 0.6$).

Control	Monitoring method	Mean Lrel	Mean of L_{rel} SD
Effort	Survey	18.51	2.63
Effort	SCPUE	15.99	2.29
Effort	ACPUE	16.45	2.46
Effort	Mean weight	17.73	2.57
Effort	Ts	14.04	1.85
Effort	Tq	13.47	2.33
TAC	Survey	18.21	2.87
TAC	SCPUE	27.96	8.63
TAC	ACPUE	23.40	8.46
TAC	Mean weight	17.62	3.35
TAC	Ts	11.78	2.06

catches or imposing higher mortality rates than predicted), as well as any subsequent assessment errors. This was compounded in those cases where R_h was also high, causing growth in catchability as the biomass was reduced.

The stock in this fishery was highly concentrated, exemplified in Figure 3.12a-d, in a randomly chosen example simulation. Here the distribution at the end of the minimal first year's fishing (Figure 3.12a) reflects the unfished distribution, concentrated around the top of the map. Catches in the first few years of the fishery varied widely (Figure 3.13), substantially exceeding the optimum in 3 of the first seven years. There was minimal reduction in vulnerable biomass by the end of the second year, with Figure 3.12b reflecting a lower level of fishing than optimal, followed by massive overfishing in the third year. This resulted in almost complete removal of the vulnerable biomass at the end of the year (Figure 3.12c). This fishery was thus already near collapse before the development period was completed. Some vulnerable biomass (produced
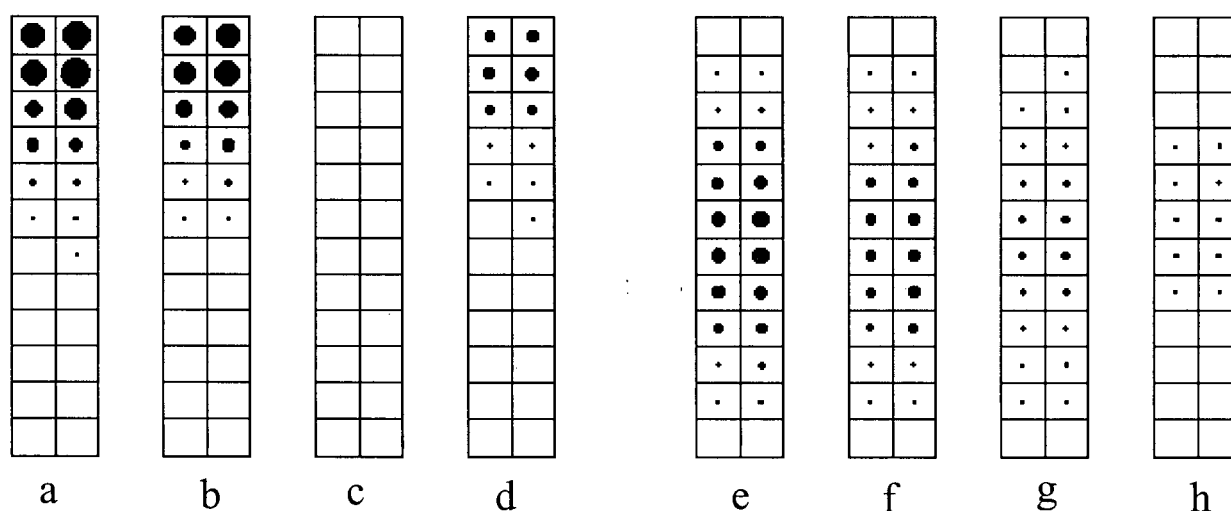
Figure 3.12. The spatial distribution of vulnerable biomass in two example fishery simulations, Scenario 1 (a-d) and Scenario 3 (e-f), at the end of the twelfth month, of years 1 (a, e), 2 (b, f) 3 (c, g) and 6 (d, h). Each fishery was monitored by aggregate CPUE and managed with a TAC control. Capacity Rc = 4; Catchability growth Rh = 4. The size of circles is proportional to the maximum vulnerable biomass among the cells in both fisheries (405.4 t in cell (1,2), a). Blank map cells indicate a very low biomass.
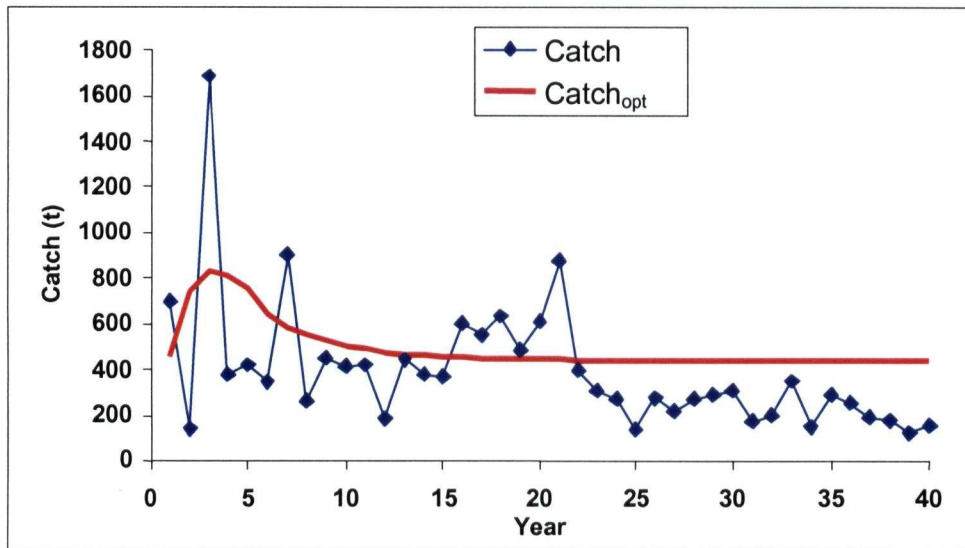
Figure 3.13. Catch and optimal catch series for an example fishery simulation. The fishery was monitored by aggregate CPUE and managed with a TAC control. Capacity R_c = 4; Growth in catchability R_h = 4.
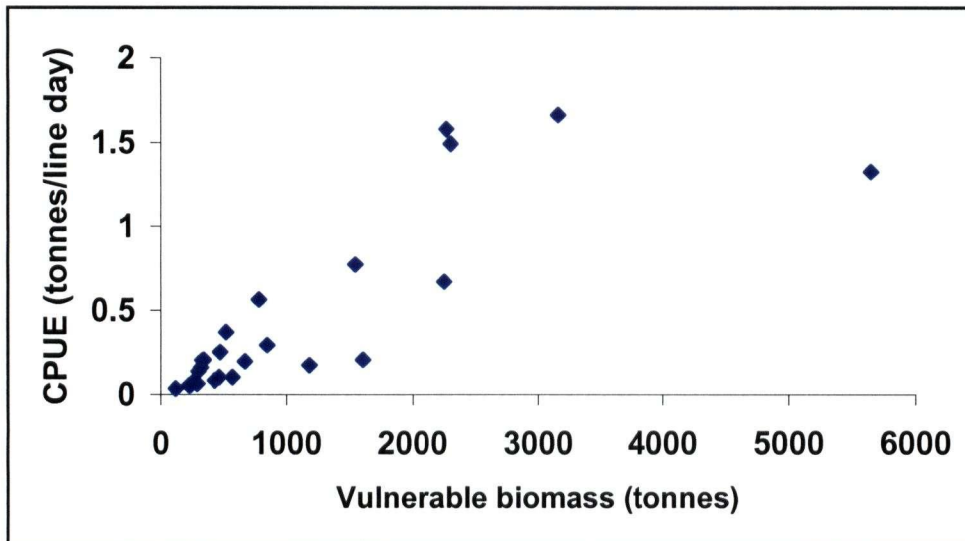
Figure 3.14. CPUE vs vulnerable biomass in an example fishery simulation. The fishery was monitored by aggregate CPUE and managed with a TAC control. Capacity R_c = 4; hyperstability R_h = 4.

through continued reproduction of pre-recruits and lagged recruitment with reduced fishing) was nevertheless evident at the end of the sixth year (Figure 3.12d). Over fishing relative to the remaining vulnerable biomass continued throughout much of the remainder of the fishery (Figure 3.13).

Centred around the middle of the map, an example Scenario 3 fishery (Figure 3.12e-h) was more diffuse and was fished down more gradually. Aggregate CPUE provided poor information for estimation. This is first illustrated by Figure 3.14, which shows the hyperstable relationship between ACPUE and vulnerable biomass over the course of the fishery. The catch rate response to biomass is uninformative, noisy and strongly non-linear. In contrast, the Tq monitoring approach had high predictive power, with predicted catchability deviating from the real value by around 9% on average (Figure 3.15). An example time series demonstrates the devastating pathology of estimation error combined with implementation error for a highly concentrated fishery. A sequence of marginal posterior distributions for B_0 estimated for a Scenario 1 fishery with ACPUE monitoring (Figure 3.16a) indicates slow convergence. Data for the first years of this example fishery were not informative and assessments were consequently prone to error, and hence, so were the control instruments (TAC or Q). As the time series evolves, a strong bias is also indicated, further increasing the probability of overfishing. This contrasts strongly with the assessment of the survey abundance index, even for the high CV (=0.6) example presented here (Figure 3.16b), in which early years' data were informative and with minimal bias.
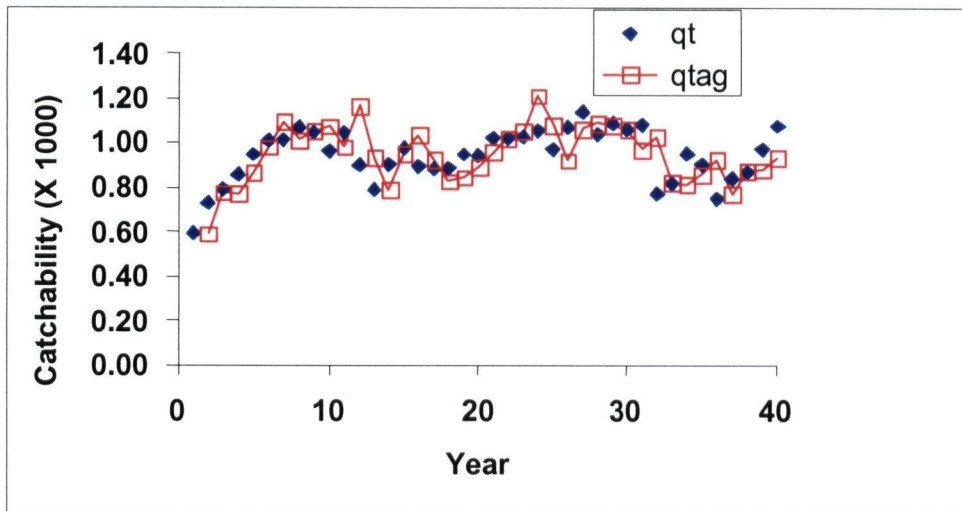
Figure 3.15. Actual (qt) and predicted (qtag) time series for an example fishery, under spatial Scenario 1, with mark-recapture monitoring (1000 tags released), and effort control, with high recruitment variation ($u_r = 0.6$) and high capacity and hyperstability ($R_c = 4$; $R_h = 4$).
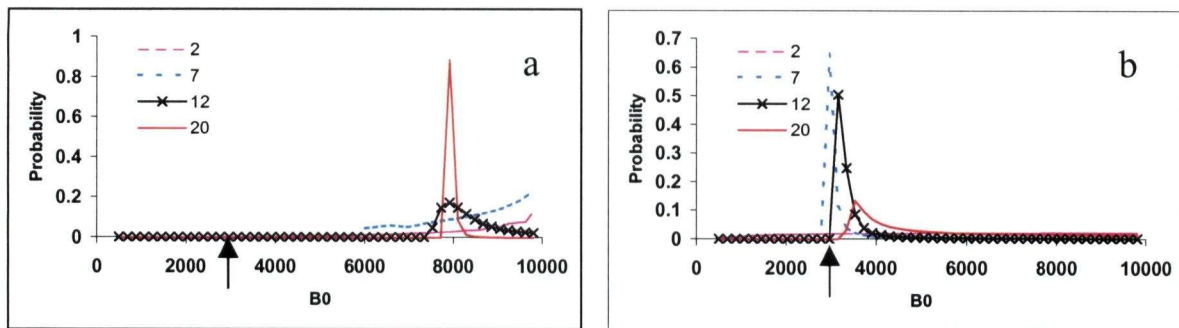
Figure 3.16. Typical sequences of marginal posterior distributions for B0 (unfished biomass of recruited fish) based on **a**, ACPUE monitoring, and **b**, survey abundance index, for a scenario 1 fishery with high recruitment variation ($u_r = 0.6$) and high capacity and hyperstability ($R_c = 4$; $R_h = 4$). Survey CV = 0.6. The actual value of B0 is close to 3000 t (arrowed).
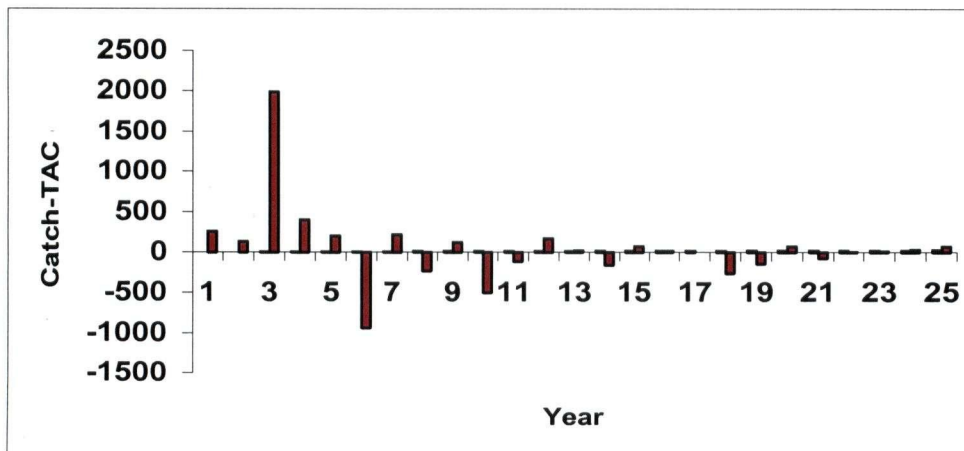
Figure 3.17. Error in TAC implementation, for an example fishery, under spatial Scenario 1, ACPUE monitoring and TAC combination, with high recruitment variation ($u_r = 0.6$) and high capacity and hyperstability ($R_c = 4$; $R_h = 4$).

For monitoring methods other than CPUE, the TAC and effort controls were similar in performance for Scenario 1 (Table 3.9) and for the set of simulated fisheries in general (as indicated above). This suggests that the TAC control was more prone to failure than the effort control when the information on stock status was biased upwards.

In the highly aggregated fisheries with high capacity, biomass could be removed rapidly. Thus, over the first five years of the example fishery, there was a series of positive implementation errors (ie catch exceeding the target TAC; Figure 3.17); the sixth year (Figure 3.17) shows a negative error because the TAC was substantially larger than the remaining vulnerable biomass (Figure 3.18).

Despite a large recruitment pulse in year 5, the vulnerable biomass was reduced to around 30% of the unfished value by the sixth year of the fishery, with erosion of biomass and high F throughout the remainder of the series shown (Figure 3.18). Performance for the CPUE/TAC combination in the single stock fisheries with high capacity and hyperstability, as summarized in Figure 3.4 and Figure 3.5, was poor whether recruitment variation was low or high. This suggests that, for the concentrated fisheries, uninformative or biased monitoring causing error in assessment in the development phase with any implementation error could rapidly result in the removal of much of the vulnerable biomass.

The objective of both management controls used in this study was to constrain the annual fishing rate F to around F_{opt} (approx. 0.844 for this fishery as specified, when applied to the vulnerable biomass).

For fisheries that differ only in whether they are TAC- or effort-controlled, then comparison of mean F over the development period (5 years) is indicative of relative implementation

performance of the two controls. For each of the controls (TAC and effort) 100 simulations were conducted of Scenario 1 fisheries, with ACPUE monitoring, high recruitment variation (u_r=0.6) at each of a range of input capacities (R_c) and catchability growth (R_h), over five years. R_c and R_h input each ranged 1.0 – 4.0, in increments of 0.5.

The mean F values increased with both R_c and R_h, generating values ranging between 0.33-1.29 for the TAC control and 0.43-0.80 for the effort control (Figure 3.19a, b). These values compared to the mean F_{opt} for the five year developmental period, 0.51 (for years 1 to 4 of each fishery $F_{opt}(t) = F_{opt}\ t\ /\ 5$). Mean F values exceeded this optimum for TAC-managed fisheries when R_c values were greater than 2.5 and R_h values were greater than 1.5 (Figure 3.19a). Mean F values exceeded this optimum in effort controlled fisheries when R_c was greater than 3.0 and R_h was greater than 1.0; however, mean F increased substantially more with R_c and R_h for the TAC–control fisheries, than for the effort-control fisheries (Figure 3.19a, b). The standard deviation for these mean F values was usually greater for the TAC fisheries, ranging from 0.099-0.410, than the effort-control fisheries (SD 0.095-0.199). The variability increased with both R_c and R_h.

The consequence of this relative implementation error with the TAC control was substantially poorer performance over the five year developmental period.

Multiple statistical areas and diffuse distribution tended to reduce the concentration of effort evident in the Scenario 1 fisheries. To illustrate this phenomenon, I ran simulations of each spatial scenario with the ACPUE/ TAC combination, the monitoring/control set most likely to generate concentrated fishing. Capacity and growth in catchability were set high ($R_c = R_h$= 4), as was recruitment variation (u_r = 0.6).
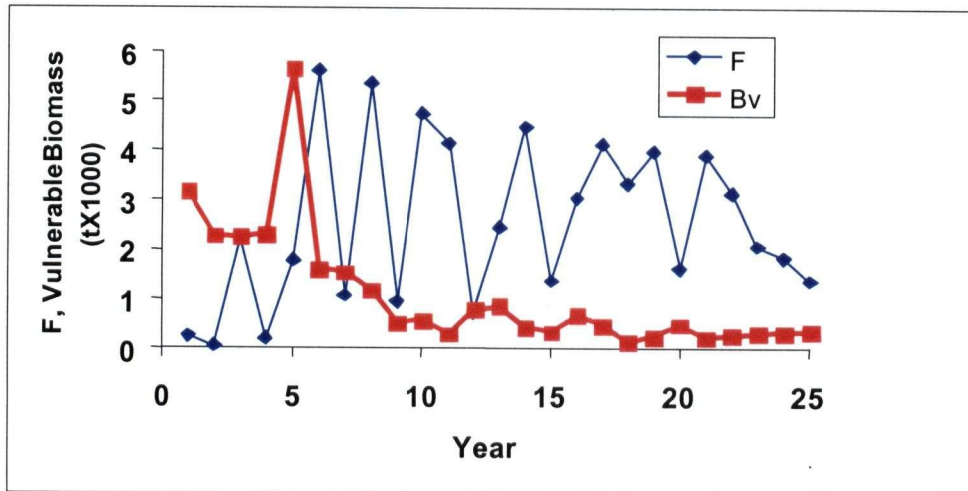
Figure 3.18. Time series of vulnerable biomass (*Bv*) and fishing rate, *F*, for an example fishery, under spatial Scenario 1, ACPUE monitoring and TAC combination, with high recruitment variation (u_r = 0.6) and high capacity and hyperstability (R_c = 4; R_h = 4).
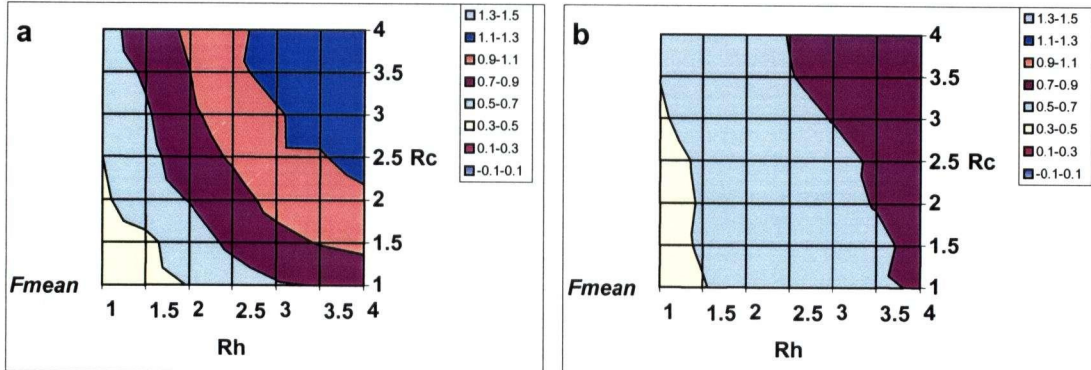
Figure 3.19. Comparison of performance measure values between TAC and effort controlled fisheries for the first 5 years of fishing. All were Scenario 1 fisheries with CPUE monitoring. F_{mean} is the mean of the annual fishing rate *F* across 100 repeat simulations for **a**, TAC control fisheries and **b**, effort controlled fisheries. R_c and R_h are the inputs that determine capacity and hyperstability (see text).

During ten simulations of each scenario, I recorded the number of cells from which most (75%) of the catch was taken in the first five years of each fishery simulation (Table 3.10). In the Scenario 1 fishery, typically 6 cells produced the bulk of the catch over the first 5 years. However, in the otherwise structurally similar Scenario 2, fish movement and a cost gradient (so that the most populous cells were more expensive to fish), tended to reduce the concentration of fishing, with 9-11 cells producing 75% of the fish. This structural subtlety had a small effect on fishery performance, so that performance in this fishery was better than in Scenario 1 (see Figure 3.4 and Figure 3.5). Otherwise structurally identical to Scenario 2, Scenario 3 produced

75% of catches from a similar number of cells (mean 9.9, Table 3.10) but overall performance for this fishery was much better than for Scenarios 1 and 2.

In the case of multiple area fisheries, assessment and the value of the regulatory control (reg_k, the proportion of the available effort predicted under the control) were independently calculated for each statistical area. This effectively limited the effort that could be focused on a single cell or group of cells within one area. The effort constraint of three statistical areas and the extra cost to fishing the highest-abundance cells for Scenario 4 diffused effort, so that a mean of 13.9 cells produced 75% of the catch (Table 3.10). In terms of the performance measures, the protection afforded by limiting the concentration of effort is evidenced in the comparison of Scenarios 5 (multiple management areas) and 6 (single), even though fishing in this multi-area

Table 3.10. Mean number of spatial cells producing 75% of the catch in the first 5 years of each fishery time series. The statistics were generated by 10 simulations of each spatial scenario. All fisheries were ACPUE monitored, TAC controlled with high recruitment variation ($u_r = 0.6$) and high capacity and catchability growth ($R_c = 4$; $R_h = 4$).

Spatial scenario	mean	SD
1	6.0	0.14
2	9.6	0.78
3	9.9	0.80
4	13.9	0.89
5	8.1	0.78
6	10.9	2.62
7	14.0	1.83
8	10.6	1.94
9	13.9	1.69
10	14.4	1.43

fishery was more concentrated on average than in the fishery with a single administration (Table 3.10). Multiple management areas diffused fishing in Scenario 7, relative to Scenario 8 (single) but did not have this effect for Scenarios 9 and 10. These scenarios included both sedentary and migratory stock components; here the number of cells producing 75% of the catch in each scenario was very similar.

Nevertheless, even in a relatively diffuse fishery, a cost gradient can have the effect of concentrating effort on the locations that are the most valuable to fish. This is not necessarily where the abundance is higher. This can result in the consequence of serial depletion, as described in Chapter 2 - stocks at the lower cost locations will be fished down to low levels before fishing switches to other locations. This is illustrated with Figure 3.20, from the twelfth year of a Scenario 6 fishery with high recruitment variation monitored and managed with the SCPUE/ effort combination, and with low capacity (R_c =1) and hyperstability (R_h) inputs.
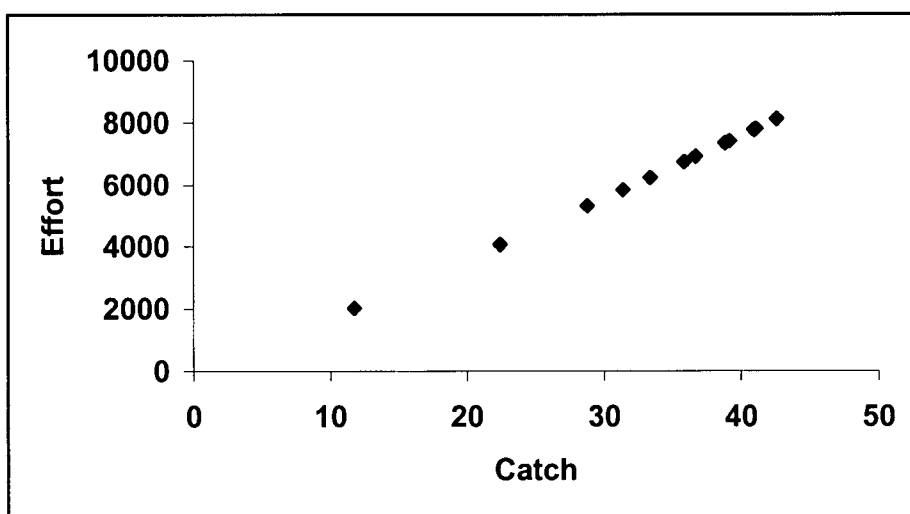
Figure 3.20. A series of in-season catches from an example fishery showing the characteristic flattened curve of serial depletion.

3.4. Discussion

Fisheries are complex systems, in which a myriad of factors acting at different time and spatial scales all interact to produce the series of catches and other information that we observe. The model system and example fisheries I have examined here are of course, highly simplified. In many respects these fishery systems were deliberately quite benign. The assessment model and the population reference model were closely matched in structure and specification. Relative to the vulnerability schedule, the early maturation and reproduction of the fished species would confer substantial resiliency to these fisheries, as would an unchanging vulnerability at age schedule. Target F values were reduced to reflect a mix of productivities. Although including sampling error, the monitoring methods did not include other errors and biases with which real fisheries contend. Management controls were probably more responsive than might ever be implemented in real fisheries. My intention was to examine the effects of fishery system spatial dynamics, the combination of fish stock and fishery spatial dynamics, on fishery performance.

I have shown that the underlying spatial dynamics profoundly affect the behaviour of a fishery. These spatial dynamics strongly interact with monitoring and control systems.

For the fisheries I examined here, performance was typically good when capacity and hyperstability were relatively low – where the inputs determined that overfishing was unlikely. As these inputs were increased, performance became increasingly more dependent upon control and the quality of information from monitoring. Poor information (from CPUE) and relatively poor control (TAC) led to very poor performance in these situations, especially in concentrated fisheries. Conversely, given good monitoring information, such as in the Tq and Ts configurations, both control systems gave good results.

Spatial dynamics and scaling

The protocol used here to distribute fishing effort was based on the assumption that, in attempting to maximize their rate of return, fishers will fish all areas down so that their rates of return (value of catch rates minus costs, or some other measure of value) are equalized (Gordon 1953). An application of Holling's disc equation (Holling 1959; Hilborn and Walters 1987) was then used to predict the distribution of effort on the basis of local values. The disc equation provides a realistic representation of the non-linear relationship between catches, with the mixture of search and fishing down processes that commercial fishing entails. As Hilborn and Walters (1987) note, the approach can be readily adapted to incorporate a wide range of factors that might contribute to the "value" of a particular fishing site. The analytic expression of effort as a function of catch (or biomass or catch rate) allows for the efficient modelling of effort distribution, reducing the computation required relative to the iterative approach developed by Hilborn and Walters (1987). This should make the approach more accessible for other simulations.

In the real world, it must be rare for the spatial attributes of a fishery's management structure to correlate perfectly with those of the fished stocks – boundaries would rarely define unit stocks more than roughly and more often might include many stocks or substocks, or share them with adjacent administrations. Interpretation of fishery behaviour will often be made at annual scales, and management unit scales. For many of the scenarios examined here, important events such as the rapid depletion of individual cells were at fine temporal and spatial scales.

The interplay of spatial stock structure and spatial management arrangements in the simulations emphasises the need to match the scales of stock and fishery dynamics with the scale of fishery management. In the case of the single substock fisheries - close to the "unit stock" fisheries on

which most fisheries science is built - the realistic distribution of fishing effort in the simulations led to the rapid fishing down of concentrations of fish; similar effects were seen in multi-substock fisheries managed in one administration. These were effects occurring at scales smaller than the fishery was managed, so that all capacity available to the fishery could be deployed at the scale of a few spatial cells. Having more statistical areas was effective in reducing this effect, in that it simply limited the effort that could be focussed on any one location, making the scale of management finer. Multiple stock fisheries also afforded examples of unexpected interaction between stock abundances and costs and the distribution of effort.

The effects of hyperstability were pervasive. It affected monitoring (below) and compounded the effect of capacity, as abundance decreased (ie catchability increased with abundance reduction). More subtly, it interacted with the distribution of fishing effort to erode concentrations of substocks through serial depletion. It is highly unlikely that any mix of substocks in a real fishery would have equal productivities so that any mix will usually result in the depletion of the less-productive stocks (Hilborn 1985). Serial depletion, likely to result wherever there is a spatial value gradient (Gordon 1953; Hilborn and Kennedy 1992), will be exacerbated when there is a difference in productivity between stocks. Consequently, where there was a cost gradient or where the substock was less productive, the substock was frequently overfished. In the multi-substock scenarios here, the mean F_{opt} corresponded closely to the F_{opt} for the second substock. Reasonably high levels of depletion nevertheless resulted, basically due to the spatial cost gradient. This was more exaggerated for the less productive third substock. Nevertheless, even in the simple fisheries of this study, there were complex interactions between cost gradients, the effort distribution function, management response and

the degree of concentration of fish so that in some instances it was the more productive stocks that were the most depleted.

Even with the policy to limit the fishery control targets at fractions of the "optimum" during the first five years, a "constrained development", the combination of assessment errors and implementation error in the high capacity and hyperstable fisheries simulated here often meant that much of the vulnerable stock had been removed even during this period. This analysis reinforces the well-worn advice for managers to limit the capacity of a fishery at the outset (eg Hilborn and Walters 1992), and to hedge by containing development until biological and operational knowledge allow appropriate scaling (Perry *et al.* 1999). However, I emphasise here that it is not total capacity, but the capacity that can be focussed at a fine scale that is important. This is very apparent in this study in the comparison of fisheries managed by single or multiple administrations. In Australia's Northern Prawn Fishery, for example, confounding of effects at different time and spatial scales has been apparent. Based on a mosaic of nine commercial species in several apparent stocks across Australia's northern coastline (the details are not yet known), the fishery has undergone an extended and expensive process to remove capacity. This has been with the intention of removing effort, in an attempt to rectify probable recruitment overfishing of the tiger prawn (*Penaeus esculteus* and *P. semisulcatus*) component of the fishery. However the removal of effort *per se* has been only partially effective, because fishing is directed firstly on the concentrations of tiger prawns that occur at the beginning of each season – mid-season closures to ensure that spawning biomass is high are likely to be much more effective at guaranteeing maximised recruitment (Dichmont *et al.* 2001).

The fine scale dynamics of a fishery have revealed themselves to be of major importance, and effort must be managed to contain effects at fine spatial and time scales. Thus fine scale spatial

and temporal processes need to be recognized explicitly (Die and Watson 1992; Perry *et al.*

1999), so that effects such as the concentration of effort on stock concentrations can be

acknowledged, and managed for. At the same time, it is important to remember that it is the

detection and fishing of concentrations of fish that makes most fishing economical, so that the

control of effort may require careful use of other controls to ensure both the realization of

opportunity and the protection of stocks. Tactics for management of stocks at fine spatial and

in-season scales might include, for example, spatial or seasonal closures to optimize biomass of

new recruits or egg production, or simply to contain F.

It is unlikely that the details of fine scale dynamics of a fishery can be predicted before it really

begins, although experience from other fisheries may indicate some attributes such as

aggregation and spawning migrations, with anecdotal information perhaps indicating locations

and timing. Ideally, development would be sufficiently controlled to enable knowledge of the

fishery to accumulate and management to be adaptive. If not, the alternative result for fisheries

in which the stock is highly concentrated may be extended depletion (eg eastern gemfish, Punt

and Smith 1999; seamount fisheries, Myers and Worm 2003). There are recommended

procedures for determining appropriate capacity and building a management basis for a fishery

(eg Clark *et al.* 1985; Perry *et al.* 1999). Application of the "proven production potential"

approach (Pearse and Walters 1992; Walters and Pearse 1996) is one protocol that could be

adopted. Capacity also has a tendency to grow (eg as "fishing power", Robbins *et al.* 1998), or

to be redeployed rapidly. There should be continuous endeavour to contain the capacity of a

fishery, and to prevent its concentrated deployment. Fine scale spatial dynamics of fish stocks

and responses to them by fishing fleets should be considered in the design and implementation

of other management controls, such as spatial and temporal closures, at all scales from small

seasonal area closures (eg Die and Watson 1992) to the design of large-scale MPAs (Walters *et*

al. 1999). Additionally, complex interactions are not restricted to fine spatial scales - the observations here on fine scale processes also apply to a wider view – similar processes occur at several scales. Phenomena such as serial depletion occur across fisheries and across species (eg the serial depletion of Alaskan crustacean resources, Orensanz *et al.* 1998), and need to be guarded against.

Management controls

The management controls presented here were simple tactical rules for applying a harvest rate strategy, with the intention being to ensure that the basic operating principles of the controls were exposed. Compared to real fisheries, they were highly idealized – effort and catch were included in calculation of controls without error, and for the TAC, the predicted catch was made as if with perfect knowledge of the spatial distribution of abundance and of the catchability relationship. Providing that information on status of the fishery was good, both control methods generally produced good performance in terms of the loss function L_{rel}. Nevertheless, the management performance for both controls deteriorated with assessment error, the TAC control being problematic when assessment was badly in error. The effort control was more responsive to stock abundance than the TAC approach, so that when information was poor, the implementation error was greater with the TAC control. This is a well established result (eg Hilborn 1979). A TAC will usually be reached regardless of the accuracy of the stock abundance prediction, unless the prediction is so optimistic that it is close to exceeding the stock size; catches from the effort control will have a damped response to a higher or lower abundance than predicted. In the hyperstable and high capacity fishery configurations monitored with CPUE, positive bias in the biomass estimate led to particularly poor performance. The performance of the TAC control is critically dependent on the accuracy of

the stock assessments, and thereby the expense of obtaining sufficiently accurate monitoring information (Walters 1996; Walters 1998b; Walters and Bonfil 1999). One response for managers to this problem is to develop TAC or effort targets more conservatively than simple choice of the maximum likelihood estimate from stock assessments (Walters 1996, 1998b), which reduce targets by inclusion of uncertainty in the stock size estimate. This approach has the advantage that, as the precision of stock size estimates improve, the targets may increase. This can be used as an incentive for participants in the fishery to undertake information gathering activities to improve precision of the estimates (Perry *et al.* 1999), perhaps by contributing to either extensive surveys at the beginning of the fishery (to narrow the range of potential estimates of unfished biomass) or for on-going participation in information gathering (Walters 1998b). Effort control performance should similarly improve with estimation performance. Nevertheless, it should also be acknowledged here that effort as used in these simulations was standardized effort. The definition and standardization of effort remain problematic (eg Robbins *et al.* 1998; Punt *et al.* 2000).

Monitoring methods and assessment

I used simple simulations of standard monitoring techniques. All were implemented at levels of precision that provided reasonably effective monitoring, to examine their basic feasibility and robustness relative to the various spatial structures. Two methodologies were atypical in performance. Firstly, the Tq approach simulated annual mark-recapture experiments to establish a harvest rate, used this to update an estimate of catchability (using a Kalman filter), then applied the effort control. By most measures and in most configurations, relative to the other monitoring / management sets, the performance of Tq was superior. This has several implications.

It may be more effective to make the simpler set of assumptions required in the annual update of the catchability estimate – for this approach this is the assessment of the state of the fishery – than those required for the more complex process of population assessment, which entails assumptions of model structure, and several inputs for specification. In these simulations, the assessment model and the population reference model were very closely matched. In real assessments, the fishery scientist has the additional concerns about model choice, and the extent to which data models and assessment models mimic reality, and their ability to detect stock declines (National Research Council 1998).

If the harvest strategy adopted for a fishery is to maintain constant F_{opt}, then the real focus of assessment is obviously F (Walters and Martell 2002), estimated each year from the tag recapture rate. Biomass need not be estimated for an effort control; the only real requirement to estimate abundance is for calculation of a TAC or catch quotas (and then, biomass may be estimated as the catch divided by the harvest rate estimated from tagging, or from the catchability prediction). The precision of abundance indices is rarely adequate to ensure safe use of TAC approaches (Walters 1998; Walters and Bonfil 1999), and as Walters and Martell (2002) contend, this more direct approach might avoid the costs and pitfalls of traditional management approaches, whose primary focus is abundance estimation.

I included the Ts mark-recapture approach (ie full population assessment using the tag information as an abundance estimate) for comparison, although in an attempt to achieve comparable performance to the Tq approach, these required more tags released. The effort control configurations, in particular, demonstrate the relative cost in performance that the full stock assessment imposes (Figure 3.4, Figure 3.5) compared to simply tracking catchability (even with the better information from more tags). An assessor might require the full stock

assessment to calculate q from mark-recapture in analyses where it was used in conjunction with other time series information on abundance (eg surveys). Walters and Martell (2002) have shown how informative small tagging programs can be for improving the performance of assessment otherwise based on abundance indices.

Regular tagging programs used for monitoring fishing mortality and catchability could have additional uses. They provide the basic requirements also for movement studies (but requiring logbook systems that produce spatially-referenced catch and effort data), providing for additional knowledge of the fine scale process of the fishery. Given appropriately designed programs, natural mortality rates might also be estimated (eg Hampton 2000). In fact Hilborn *et al.* (1990) contend that most of the useful information in fisheries has come from tagging studies.

The Tq approach required some strong simplifying assumptions, so it should be emphasised with similar strength that its performance might deteriorate markedly if those assumptions were not met, and performance may not be so superior to other monitoring methods if it were possible to ensure that all were equally "idealized". Nevertheless, the simulations here showed the potential of management with a method that is truly informative of the impact of fishing on a stock, a direct measurement of the probability of a fish being caught. The Tq approach suggests that relatively few tagged animals are necessary for effective monitoring of q and F, but there is a set of stringent assumptions that must be met. The extent to which these are not met degrades the value of a mark-recapture approach. These assumptions are that tag-shedding and tag-induced mortality are well quantified, that the tagging is representative over the exploitable population, and that the reporting rate is known exactly. If unaccounted for, these errors can lead to bad bias in the estimates. All these assumptions might be met in particular

circumstances by extra experimentation and control. The difficulty (which often equates to cost) of meeting them is probably a major reason that tagging has not been used even more extensively. Many species have been simply too difficult to tag with the economy required for a monitoring method.

I propose that genetic mark-recapture based on DNA-fingerprinting techniques and remote tissue collection has large potential for use as a monitoring tool, overcoming some of the limitations in conventional tagging listed above. The approach also has the potential to incorporate additional information from catch and release activities of a recreational fishing sector (Buckworth and Martell 2003). I further examine the feasibility of genetic tagging as applied to NT *Scomberomorus commerson* in Chapters 5 and 6.

Both levels of the CPUE method (spatial and aggregated) performed badly in combination with the TAC control, in some spatial scenarios. This was due firstly to failure of the assumption of a linear relationship between CPUE and abundance – a well known phenomenon (eg Hilborn and Walters 1992) – which led to biased abundance estimates, in combination with implementation error inherent in TAC control. A modification of the CPUE monitoring, to reduce bias could have been to model the relationship between the CPUE and abundance as a power relationship

$$Y = qB_t^b \qquad\qquad \text{Eq 3.1}$$

where b is estimated as part of the assessment process, or is known from independent information (Cooke and Beddington 1984; Walter 1996). This might have improved the general performance of this monitoring method, by reducing the bias in abundance estimates. However introduction of an extra parameter would have meant an extra requirement on assessments that

were already information limited, especially during the first few years. An alternative approach would be to make a "safe" assumption of hyperstability, setting b at some value ≤ 0.5. This precautionary approach would penalize fisheries that were not hyperstable.

The mean weight monitoring performed well, but as for the Tq approach there were implicit assumptions that the only error was sampling error - there was no measurement error, nor non-independence (spatial or temporal correlation among samples). For long-lived species that are slow to mature and recruit at higher ages than considered here, mean weight is likely to be poorly responsive to abundance change. The assessment model also did not accommodate changing vulnerability at age, so that additional error and bias could be extensive. Nevertheless the use of mean weight data, at least as auxiliary information (Hilborn and Walters 1992), could merit further investigation. For many fisheries that have central points of landing or processing, mean weight data might be readily obtained. If fisheries are for large, relatively few fish, it may be feasible to estimate weight from landed weights and catch number information collected as part of logbook data systems.

3.5. Concluding remarks to Chapter 3

I begin this set of remarks with a reminder that in many ways the simulations here represent relatively benign systems: real fisheries are almost certainly worse-behaved than those I have depicted. The analysis of this chapter produces several clear messages for managers of spatially complex fisheries. The first message reiterates a familiar warning that fisheries must be contained early in their development, or risk prolonged depletion. The concentration of effort on concentrations of fished stocks can cause rapid erosion of stock biomass; serial depletion can erode substocks and biomass more subtly.

This prompts the second message, which is that fishery scientists and managers should seek to ensure that quality information on the spatial behaviour of a fishery is captured.

A third message is that quality information on system status is necessary to ensure good performance. This might be in the form of accurate indices of abundance but the suggestion here is that monitoring F and tracking q might be a more effective approach. The choice between monitoring approaches is typically a trade off between perceived benefits, some average over a range of possible management results, and costs. The choice of a poorly informative monitoring measure (eg based on CPUE) in combination with a TAC is not a safe choice. Development of new techniques and protocols similar to the Tq approach described here may require investment that really pays off in terms of fishery performance.

4. ASSESSMENT OF THE FISHERY FOR NARROW-BARRED SPANISH MACKEREL, *SCOMBEROMORUS COMMERSON* IN THE NORTHERN TERRITORY, AUSTRALIA

4.1. Introduction

The narrow-barred Spanish mackerel, *Scomberomorus commerson* (Lacepède 1800-1803), is a large, fast-swimming pelagic predator, found throughout tropical and sub-tropical neritic waters of the Indo-West Pacific (Collette and Nauen 1983). Usually taken in depths < 100 m (Munro 1943; McPherson 1988), it is often associated with reefs and islands, and is targeted in commercial, artisanal and sport fisheries throughout its range. Recent work has emphasised the species' excellent table, nutritional and storage qualities (Hay *et al.* 1996; Slattery 1998). It is also renowned as a light gamefish.

Annual global landings of the species are substantial. They increased steadily during the late 1990s from less than 170,000 tonnes in 1995 to nearly 205,000 tonnes in 2000 (FAO Fisheries Production Statistics). The Australian commercial fisheries make only a modest contribution (< 1%) to this catch, their annual landings being just 1200-1700 tonnes (whole weight) over 1999-2002 (ABARE 2003). Recreational and indigenous landings are not precisely monitored, but add significantly to the commercial catch: a recent survey estimated the total catch of *Scomberomorus* spp. catches in Australian waters by recreational fishers at 471,671 individuals (Lyle *et al.* 2003a), and by indigenous fishers at 4,222 (Coleman *et al.* 2003). This recreational catch includes *S. semifasciatus, S. queenslandicus, S. munroi*, and possibly *S. multiradiatus*, as well as *S. commerson*. Without further information on species and size composition, the

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magnitude of the non-commercial catch can only be approximated, but combined with the commercial catch, they imply the Northern Australia total landings of *S. commerson* to be in the range 2000-3000 tonnes.

The species is taken in a managed Northern Territory (NT) fishery, and NT managers should confront the questions with which I began this thesis. In this chapter I ask "what is the impact of fishing on the Northern Territory stocks of narrow-barred Spanish mackerel?" Below, I examine the problems facing management in the NT Spanish mackerel fishery. I review the biology of the species, particularly with respect to information that might affect assessment and the choice of monitoring and management control sets for fisheries in northern Australia. I develop an equilibrium model for the NT fishery, and also provide a further assessment based on catch, effort, weight and age structure information. I conclude that without new monitoring information that better measures fishing mortality rates, the status of the fishery and production potential for the NT fishery remain grossly uncertain.

4.2. Biology of Spanish mackerel

4.2.1 Life History

Spawned in oceanic conditions on reef slopes and edges, the eggs of *S. commerson*, with a large oil droplet, are presumably pelagic (Munro 1942; Mackie *et al.* 2003). The spatial distribution and dynamics of the larvae are poorly known. However, there is some evidence of separation of *S. commerson* larvae from their congenerics. In Great Barrier Reef (GBR) shelf waters, larvae of the species were found only in the oceanic conditions of the lagoon. They were spatially separated from those of *S. semifasciatus*, which were found in coastal bays and the inner lagoon,

and *S. queenslandicus*, which were located in coastal bays as well as across the lagoon (Jenkins *et al.* 1985).

Spawning is seasonal (Munro 1942; Devaraj 1993; McPherson 1993; Mackie *et al.* 2003) and appears to be increasingly protracted with the higher water temperatures of the tropics (McPherson 1993; Buckworth and Clarke 2001; Mackie *et al.* 2003). Most fisheries for the species are based on pre-spawning feeding aggregations. In NT waters, most female fish in the catch during the July–December fishing season are close to spawning or have recently spawned (Buckworth and Clarke 2001). Spanish mackerel are batch spawners, and size at 50% maturity for females is attained at a size of around 80 cm (FL) and age of 2 years (Devaraj 1993; McPherson 1993; Mackie *et al.* 2003). The timing of spawning coincides with high water temperatures and maximal food availability and maximal growth conditions for the larvae (Jenkins *et al.* 1985).

Juveniles are found in coastal-estuarine habitat (McPherson 1981, 1988). Lack of *S. commerson* larvae in sampling of near-shore habitats, but presence offshore in the GBR lagoon, indicates directed movement inshore by young juveniles (Jenkins *et al.* 1985).

Although the earliest larval stages consume principally larvaceans (Jenkins *et al.* 1984), *S. commerson* of all later life history stages are largely piscivorous. Later larvae, with exaggerated jaw and mouth development and a well-differentiated stomach, prey on relatively large fish larvae, selecting larger prey as they grow - apparently following the large prey-fast growth strategy typical of scombroids. Schooling also begins at end of the larval stage (Jenkins *et al.* 1984). Small juveniles consume the larvae and juveniles of small fishes and crustaceans (Kumaran 1962; Rao 1962). Similarly, the diet of older juveniles and adults is chiefly smaller fish and squid (McPherson 1987a; Devaraj 1993).

Several studies have described age and growth of *S. commerson* (the more detailed studies include Devaraj 1982; Dudley *et al.* 1992; McPherson 1992; Govender 1993; Buckworth 1998a; Mackie *et al.* 2003). Mackie *et al.* (2003) provide an extensive review of growth. Spanish mackerel may achieve a length of > 2.0 m, and grow rapidly so that 50% of this length is achieved by faster-growing fish within 2 years (Dudley *et al.* 1992; Mackie *et al.* 2003).

Australian studies of the species have indicated that there is a strong sexual dimorphism in growth, with females being larger at age than males (McPherson 1992; Buckworth 1998a; Mackie *et al.* 2003). Longevity is only moderate: the oldest specimens reported include 16 years in the Australian east coast fishery (McPherson 1992), 12 years in the NT fishery (Buckworth 1998a) and 20 and 22 years (Mackie *et al.* 2003) from northwest Western Australia. In Oman they are suggested to attain a maximum age of 10 or more years (Dudley *et al.* 1992).

Size at age appears to be highly variable, although some of this variability may be due to ageing error arising from difficulties with otolith interpretation, due to otolith morphology and multiple banding (McPherson 1992; Govender 1993; Buckworth 1998a; Mackie *et al.* 2003).

Estimates of natural mortality for Spanish mackerel also vary substantially between studies, locations and the methodologies used to produce those estimates. They are typically in the range $M = 0.3$ year^{-1} to 0.8 year^{-1} for *S. commerson* and for *Scomberomorus* in general (see tabulations by Welch *et al.* 2002 and Mackie *et al.* 2003). Relying on a range of empirical approaches, estimates from Western Australian waters are relatively low, 0.16 to 0.20 year^{-1} in warm temperate waters, but are 0.32 year^{-1} for females and 0.37 year^{-1} for males in northern tropical waters (Kimberley region). For Queensland waters, a maximum likelihood estimate of $M = 0.40$ year^{-1} was derived from an age-structured dynamic model fitted to catch per effort and age structure data (Hoyle 2002).

4.2.2 Stock structure

Scomberomorus commerson are normally regarded as being highly migratory, but there also appear to be resident substocks (Collette and Nauen 1983). Migratory behaviour has been inferred from seasonality of fisheries (eg McPherson 1988; Buckworth and Clarke 2001, northern Australia; Dudley *et al.* 1992, Oman; Govender 1993, southern Africa), the results of the only substantial tagging study undertaken with the species (Australia east coast, McPherson 1987b), and studies of the ecologically similar *S. cavalla* (eg Fable 1990, Johnson *et al.* 1994). In contrast, localized phenomena such as external parasite scarring and observations of local depletions (G. McPherson, Queensland Department of Primary Industries, unpublished data), and the very localized incidence of ciguatera, imply that many fish are effectively resident.

Allozyme studies (Shaklee *et al.* 1990) indicate that in the Australian region, *S. commerson* has three broad genetic stocks: an east coast stock, a single stock ranging from southern Papua New Guinea across northern Australia and down the west coast of the continent, and a stock in the Torres Strait area (shared by Australia, Papua New Guinea and Irian Jaya) with strongest affinity to the northern Australia stock. Use of mitochondrial DNA (mtDNA) (Ovenden *et al.* 2003) has supported these analyses, although the use of mtDNA enabled identification of a clade that defined the affinity of Torres Strait fish with the east coast stock and a small number of individuals who may have been from stocks from the north of the New Guinea region.

Observations of movement from tagging and seasonal fisheries, and genetic studies, both suggest that Spanish mackerel are highly mobile and well-mixed over substantial ranges. However, recent studies of otolith chemistry (Newman *et al.* 2003) and parasite incidence (Lester *et al.* 2001; Moore *et al.* 2003) contradict the panmixia hypothesis. The otolith and parasite studies indicate that adults from samples taken throughout northern Australia form

distinct functional assemblages that mostly do not mix even at distances as small as a few

hundred km. Resolving the apparent disparity between these two sets of studies – genetics

compared with otolith chemistry/ parasite incidence – requires consideration of the time scales

to which the sets of information refer. The genetic studies are generally informative about

mixing that occurs over millenia, so that the major divisions in northern Australia probably

reflect isolation of the east coast and northern/western populations of *S. commerson* during low

sea level periods of the last Ice Age (Ovenden *et al.* 2003). Relative genetic homogeneity of

stocks could be maintained by very low levels of mixing, by mechanisms such as limited larval

and juvenile dispersal, or by the movement of small numbers of individuals between the adult

functional groups (as inferred from parasites by Lester *et al.* 2001, and mtDNA by Ovenden *et al.* 2003).

Spanish mackerel population structure in northern Australia may be described as a mosaic of

substocks, perhaps as a metapopulation, that effectively do not mix as adult fish, separated by a

few hundred km or less. Under this hypothesis, seasonality in catches would represent seasonal

changes in availability as fish concentrate in pre-spawning feeding aggregations; the migration

of east coast fish described by the mark-recapture work of McPherson (1987, 1988) may then

represent a response to seasonal extension of habitat, the extent and incidence of migration

reflecting the latitudinal distance between spawning and seasonal feeding grounds. Whether

new recruits join migrating or non-migrating groups, "contingents", could be based purely on

learned behaviour, as suggested for Hudson River striped bass, *Morone saxatilis* (Secor 1999)

and Atlantic herring, *Clupea harengus* (McQuinn 1997).

4.3. Description and history of NT fishery

Located principally around the islands and shoals near the principal ports of Darwin and Nhulunbuy (Figure 4.1), the Northern Territory (NT) Spanish mackerel fishery is mostly a hook and line fishery. The NT annual commercial catch in recent decades has been as little as 54 tonnes (1986), and as much as 385 tonnes in 2001 (Figure 4.3; logbook data, Fisheries Group, NT Department of Business Industry and Resource Development; hereafter, NT Fisheries).

There are no time series of recreational Spanish mackerel catch data for Australia as whole. However, in the Northern Territory, estimated 2001 recreational catches (including those from guided fishing) of combined *Scomberomorus* species were 21300 individuals (precision of this estimate was not available at the time of writing). This included an estimate of 5751 (27%) or 40.3 tonnes of *S. commerson* (Lyle *et al.* 2003a) and assuming similar species composition, 2.7 tonnes were also landed by indigenous fishers (Coleman *et al.* 2003; these authors assume a mean weight of 7kg for *S. commerson*). Recreational catch was similar in 1995, with the estimated catch for all *Scomberomorus* species being 25,099 individuals (± 1,807 s.e.) in 2001 (Coleman 1998). Recreational fishers in general harvested most (97%) of the Spanish mackerel catch (Coleman 1998). In contrast to general recreational fishing trends, catches of the species by recreational fishers during guided fishing tours in the Northern Territory have increased more than twofold over the same period, from 1165 to 2423 fish with more than 60% released in most years (NT Fisheries logbook data). Indigenous catches in 2001 were relatively small, at 4 tonnes (Coleman *et al.* 2003).
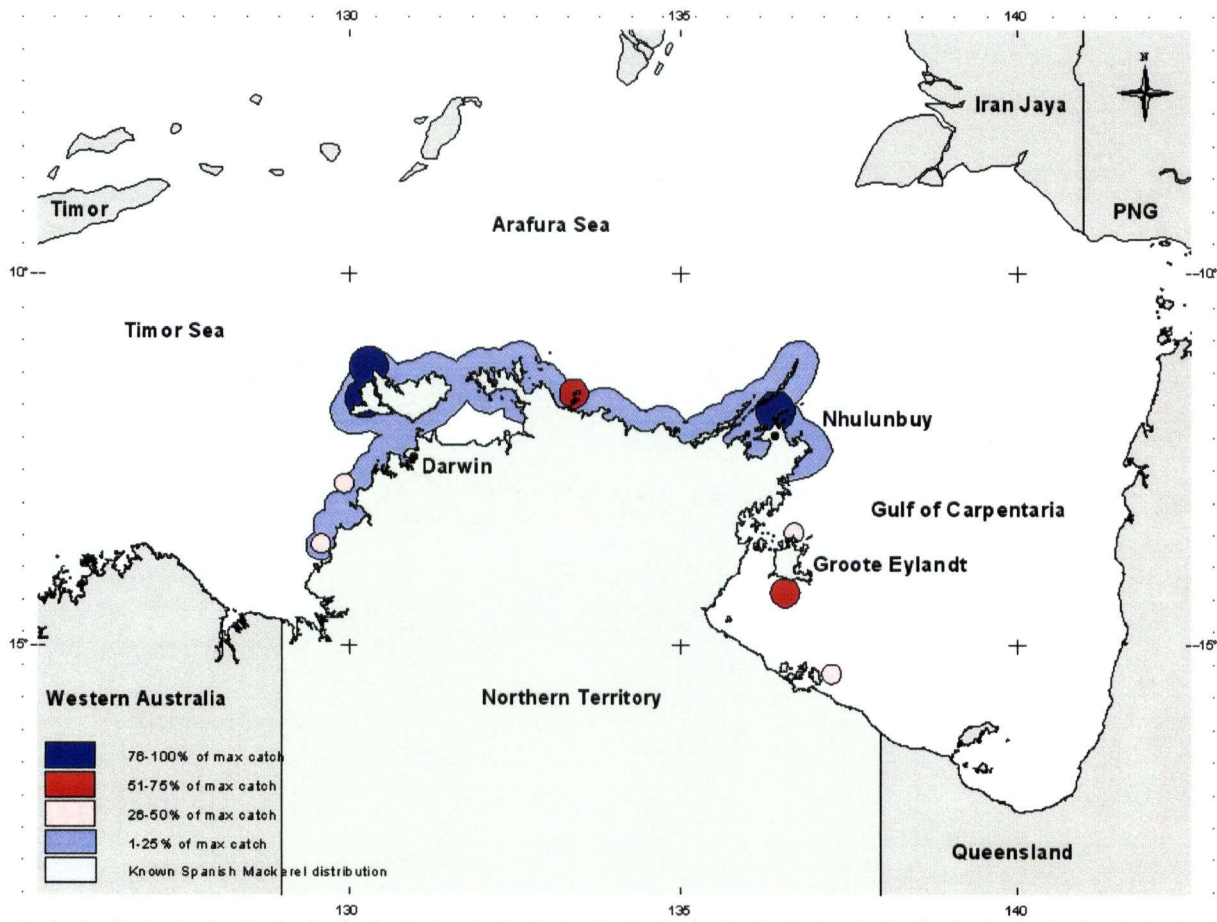
Figure 4.1. The relative importance of different locations to the NT Spanish mackerel troll fishery, based on relative catches within one degree grid squares during 1995-1999. The size of the symbols relates the catch in any grid to the maximum catch grid. Only grids where five or more licensees reported catches are shown (after Buckworth and Clarke 2001).

In all the Australian fisheries, management is chiefly by input controls (limited entry and gear restrictions) for commercial fisheries, additionally with minimum size limits in the Queensland and Western Australian jurisdictions. Output controls apply in recreational fisheries (possession limits and/ or size limits). Historically, the biggest impact on NT Spanish mackerel populations may have been a Taiwanese distant water gillnet fleet, that fished northern Australian and Indonesian waters from 1974 to 1986 (Figure 4.2). Before 1978 the Taiwanese distant water fleet was able to fish throughout Australian waters, outside 12 nautical miles from the coast. They were excluded from the Gulf of Carpentaria from August 1978 but licensed with the declaration of the Australian Fishing Zone (AFZ) at the end 1979, to take a total annual quota of

7000 tonnes (of all species retained, Stevens and Davenport 1991). Further exclusions were adjacent to Melville Island and the Arnhem Land coast (Figure 4.2). The principal targets were shark species, but tunas and *Scomberomorus* spp. were also significant in the catch, and taken throughout the permitted fishing area. Operations became uneconomic and ceased in late 1986, when concerns over cetacean bycatch led to restriction of the amount of net that could be set per vessel, to 2500 m.

Although there were reports of annual catches of 1000 tonnes or more of Spanish mackerel by the Taiwanese fleet in the late 1970s (Rohan and Church 1979), mackerels were typically less than 10% of total catches in most months, and logbooks and observers reported catches of 400-500 tonnes of all *Scomberomorus* through the early 1980s (Stevens and Davenport 1991). Nevertheless, there were persistent observations of misreporting of catch species composition, under-reporting of catches, and various ploys to circumvent quota limits (Millington 1981). Multiple vessels even displayed identical names and licence numbers (Milton Miller, NT Mackerel Troll Fishermen's Association, Darwin, *pers. comm.*).
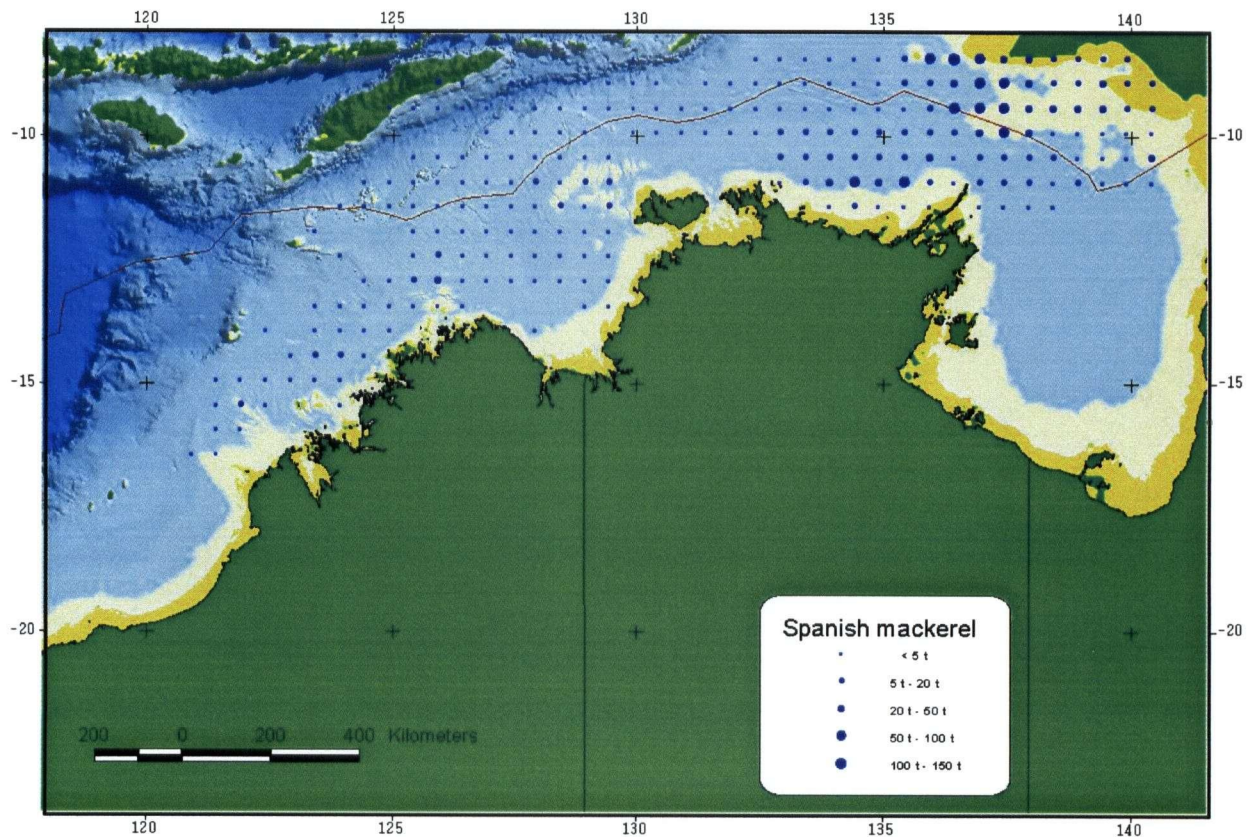
Figure 4.2. Distribution of Spanish mackerel catches from the Taiwanese pelagic gillnet fishery in the northern Australian region, 1980-1986, as indicated by logbook data 1981-1987. Symbols represent total catches by ½ degree squares. The seaward line indicates the boundary of the Australian Fishing Zone. Map provided by Mr K. McLoughlin, Bureau of Rural Sciences, Canberra.

Reductions in catch per unit effort and mean size of *S. commerson* in the Taiwanese catch over the early 1980s were signals that the NT stock of Spanish mackerel may have been overfished (Stevens and Davenport 1991). In response to these signals, regulation of Spanish mackerel as a target fishery in the NT began in 1990. Until this time, Spanish mackerel was among many species taken in the domestic fishery under a general "offshore" licence. With little information for alternative assessments, the apparently stable (approximate equilibrium) recent catch of the Taiwanese fleet (450 tonnes pa) was taken as indicative of the sustainable yield for the fishery. Choosing this value as the management target catch, a conservative management regime was developed to ensure protection of the resource.

A primary management action was limitation of the number of commercial licences in the fishery, reducing the number of potential participants from several hundred to 28. A management plan (1993) encoded several additional measures to contain fishing effort, such as gear restrictions and a scheme for the removal of licences (Buckworth and Clarke 2001). By July 2003, 19 operators remained in the fishery.

A primary purpose of research and monitoring in the past decade has been to examine the accuracy of the 450 tonnes management target catch, but the major conclusion of repeated assessments over time has been that the status of the fishery remains very uncertain (eg Walters and Buckworth 1997). Revision of the management plan is likely to include a total catch of 450 tonnes from all sectors as a limit reference point, with review of the management of the fishery triggered if that value is approached. A key question in such a review is likely to be whether the fishery can sustain substantially larger catches, or whether greater management control will be necessary to ensure that overfishing does not occur.

Landings by the Northern Territory domestic troll fishery for Spanish mackerel were minor before rising rapidly during the 1980s and 1990s. The troll fishery achieved an estimated annual catch of close to 400 t (whole weight) in 2001 (Figure 4.3). Chilled, whole Spanish mackerel may fetch prices exceeding Aust. $10 kg^{-1} (www.sydneyfishmarket.com.au), potentially valuing the 2001 catch at around Aust. $4 million. However, operational constraints such as distance from the NT's few isolated ports, small local markets, transport costs and high ambient temperatures mean that most of the catch is sold frozen in fillet or trunk form, equating to whole fish prices closer to Aust. $5 kg^{-1}. Given the low level of participation and unused capacity (see below), effort and catches from the fishery could increase substantially in response to incentives from market or operational innovation. The conservative nature of the

management regime for the NT Spanish mackerel fishery was recognized by recent accreditation for export of product under the Environment Protection and Biodiversity Conservation Act (Commonwealth of Australia 1999), which is similar to Marine Stewardship Council accreditation. The fishery was the first Australian finfish fishery to receive this accreditation. Further details of management controls of the fishery are provided by SMACFAC (Spanish Mackerel Fishery Management Advisory Committee) (2000) and Buckworth and Clarke (2001).

4.4. Assessment

The objective of assessment has been to establish the status of the NT Spanish mackerel fishery – whether catches can be sustained at the current fishing mortality rate, or even increased. In this section, the catch and effort histories of the fishery are summarized, and an equilibrium model is used to examine basic aspects of the fishery's dynamics. A dynamic age-structured model is then employed to examine the impact of fishing in more detail.

4.4.1 Catch and effort history of the NT fishery

The NT Fisheries logbook system stores records of catches and effort in the troll fishery and other commercial fisheries since 1983, and includes Fishing Tour Operations (FTO: guided fishing) since 1994. Catch and effort data are submitted monthly by fishers. Catch data are available as weight (kg) of processed form (ie whole, fillet, trunk etc). These data were converted to whole fish weight using standard conversion factors (Mackie *et al.* 2003). These conversion factors represent a substantial correction from those previously used in summary catch estimation and subsequent analyses, so that catch series presented here are around 15% less for any year than previously published (eg by Buckworth and Clarke 2001).

Total catches (Figure 4.3) show two peaks in the 1979 - 2001 period, the first a period when annual catches exceeded 400 tonnes, centring on a peak of 442.3 tonnes in 1984 that was principally due to the Taiwanese fishery, and a second peak of 429.7 tonnes in 2001. The main catches after the Taiwanese fishery ceased were from the troll fishery. Troll catches increased steadily through the 1980s and 1990s, varying from a mere 25.9 tonnes in 1986, to 373.1 tonnes in 2001 (Figure 4.3).

Other commercial fisheries, principally a gillnet fishery that targets shark and grey mackerel (*S. semifasciatus*), took over 59 tonnes in 1985, but usually 10-20 tonnes through the 1990s and to 2001 (Figure 4.3). The decrease reflects both increased specialisation in those fisheries, and the action of management constraints. The recreational catch in 1995 was estimated as 47.4 tonnes (Coleman 1998) converting a catch of 25099 fish, assuming *S. commerson* were 27% of the catch, at a mean weight of 7 kg, as per Lyle *et al.* (2003b). In 2001, including indigenous catch, the recreational take was 44.3 tonnes (Lyle *et al.* 2003b; Coleman *et al.* 2003; Figure 4.3).
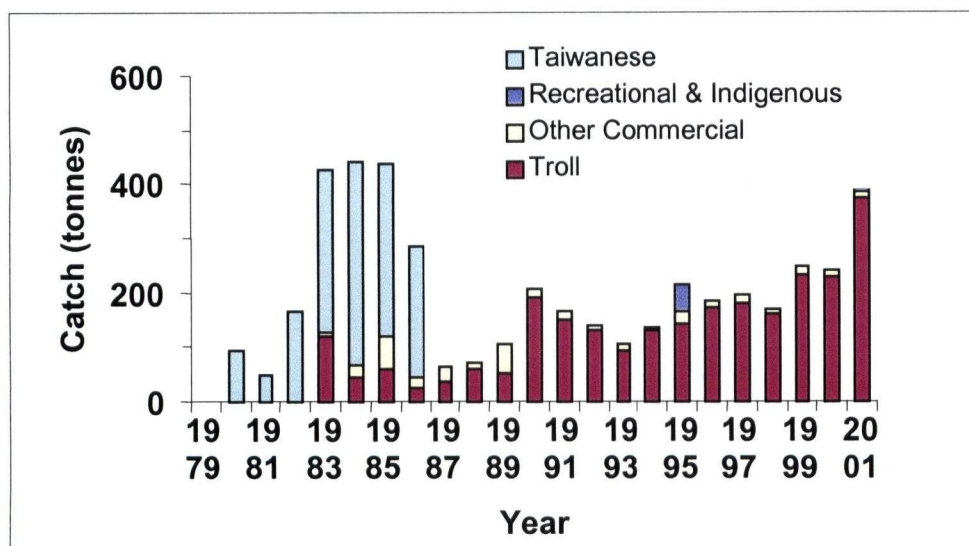
Figure 4.3. Estimated catches of Spanish mackerel from Northern Territory fisheries, as indicated by logbook and survey data. Taiwanese data are from 1981-1987, NT troll fishery and other commercial fishery data are from the NT Fisheries logbook system and recreational and indigenous catches are from Coleman (1998), for 1995, and Coleman *et al.*(2003), for 2001.

Logbook data are available for the Taiwanese sector of the fishery for 1980 to 1986. Relatively low catches recorded as Spanish mackerel by the licensed Taiwanese driftnetters in 1980 and 1981 increased to a peak of 376.6 tonnes in 1984 (Figure 4.3). Spanish mackerel were the principal species taken in this sector (Table 4.1) although the species groups in which data were recorded do not correspond directly to species taken in Australian waters. Presumably, "Spanish mackerel" were chiefly *S. commerson*, with the other groups comprising *S. semifasciatus, S. queenslandicus,* and *S. munroi* and perhaps *S. multiradiatus, Acanthocybium solandri* and *Grammatorcynus* spp. Total catches of all mackerels by the Taiwanese fishery ranged from 86.4 to 818.0 tonnes (Table 4.1).

Operators in the troll fishery are permitted to use a mother boat, used for accommodation, processing and storage, and often for fishing, and up to two dories (small boats <8m). Mother boats typically tow 4-6 lines, dories 2 or 3 each. Terminal tackle is usually gangs of hooks, baited with squid or garfish, or lures. Fishing is a mix of test fishing known fishing locations in

Table 4.1. Annual catches of *Scomberomorus* mackerels, 1980-1986, by species group indicated in Taiwanese fishery logbooks. Data summarized from 1981-1987.

Year	Spotted Spanish mackerel	Spanish mackerel	Japanese mackerel	Total weight
1980	211.2	92.9	13.6	317.8
1981	22.6	47.1	16.7	86.4
1982	104.8	166.3	57.1	328.2
1983	83.1	298.9	72.9	454.8
1984	116.3	376.6	90.0	582.9
1985	48.7	318.5	53.6	420.9
1986	453.9	239.7	124.4	818.0

sequence, and remaining at locations while catch rates remain adequate (ie better than anticipated elsewhere). Exploration consists of test fishing sites where fish have been observed leaping from the water (a habit of *S. commerson*), or where experience about bathymetric features such as reefs and shoals marked on charts suggest fishing for Spanish mackerel might be successful.

The basic effort unit provided by the NT Fisheries logbook system is the number of fishing days (days of operation of a fishing unit), referenced to one-degree grid squares. The number of lines used has been recorded throughout the data series, so that linedays can also be used as an effort unit. Troll fishery effort (Figure 4.4) increased steadily through the 1980s, from a low point of 122 boatdays in 1985. A sharp increase in 1990 to the highest level of the series, of 1887 boatdays, reflected the imminent introduction of management control of the fishery, with the issue of licences for the fishery dependent upon catch history. Effort declined again through the early 1990s, to another low point of 553 boatdays in 1995, then increased through the remainder of the 1990s and increased sharply again to the 1155 boatdays of 2001.
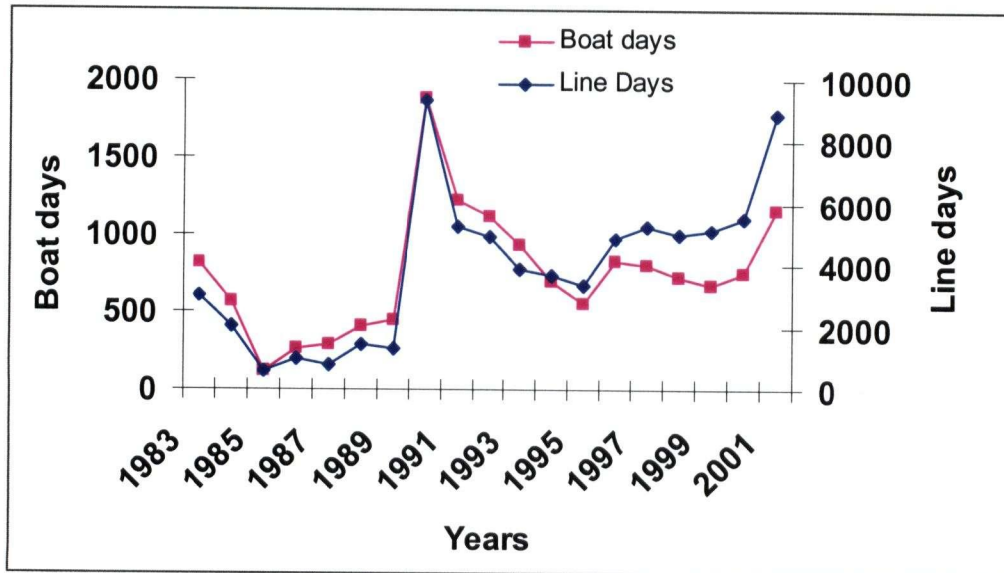
Figure 4.4. Total annual effort for the Northern Territory Spanish mackerel troll fishery. (Source: NT Fisheries logbook data system.)
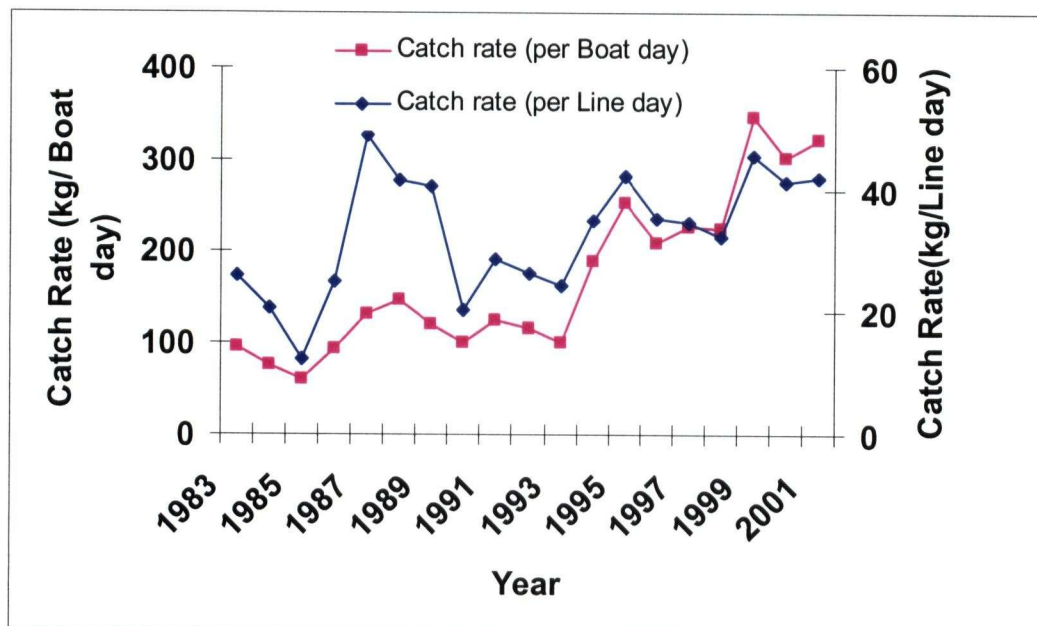
Figure 4.5. Gross catch rates for the Northern Territory Spanish mackerel troll fishery. (Source: NT Fisheries logbook data system.)

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The trend for linedays as a measure of effort was very similar (Figure 4.4). However, an increasing trend in the number of lines used through the late 1990s is apparent in the steeper increase in line days fished over that period. Thus in terms of boat days, 2001 effort (1155 boat days) was only 61% of the 1990 effort peak (1887 boat days), but effort measured as line days in 2001 (8884 line days) was 95% of the 1990 peak.

Gross annual catch rates, annual catches divided by total annual effort, have generally increased through the history of the fishery (Figure 4.5). With annual effort measured as boat days, catch rates declined from 95.5 kg boatday^{-1} in 1983 to 59.4 kg boatday^{-1} in 1985, but then generally increased to peak at 346.4 kg boatday^{-1} in 1999. Although following a similar, generally increasing trend (Figure 4.5), with a low point of 12.3 kg lineday^{-1} in 1985 and a high point, in 1999, of 45.8 kg lineday^{-1}, catches per lineday also showed a strong rise during the mid 1980s, with a peak of 49.0 in 1987 kg lineday^{-1}.

The number of licensees fishing showed a similar pattern as the effort time series, up until 1990. The number of participants fell from 22 in 1983 to just 7 in 1985, then increased steadily through the late 1980s to peak strongly in 1990, with 36 participants. The number of participants has declined steadily since, reflecting in part the reduction in the number of licences in the fishery (the reduction between 1991 and 2001 was from 28 to 19).

Even given the low number of licences entitled to take part in the NT Spanish mackerel troll fishery, most of the catch has been taken by a handful of operators through most of the course of the fishery (Figure 4.6). During the mid-1980s period of low effort, 70% or more of the catch was taken by just 1 or 2 operators. Peak participation was in 1990 and 1991, when 70% of the catch was taken by 11 licensees. Participation since has been such that between 3 and 6 licensees take 70% or more of the catch (Figure 4.6).
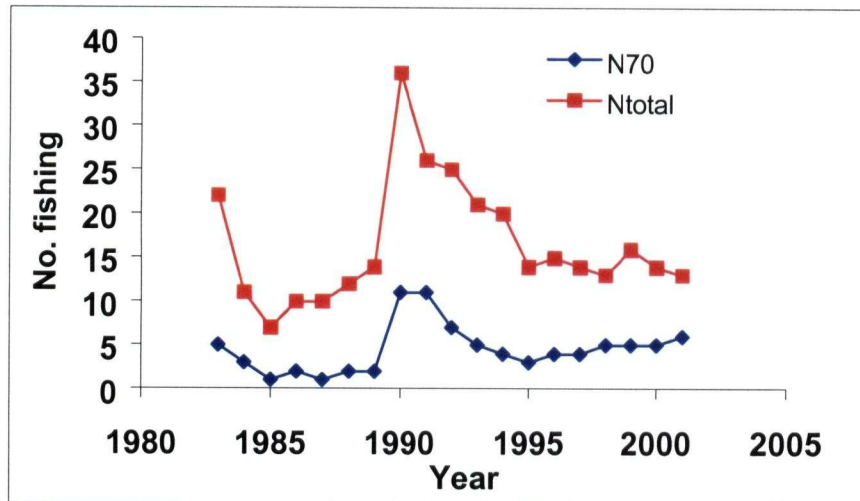
Figure 4.6. Number of participating licences and proportion taking the bulk of the catch, in the NT Spanish mackerel fishery, 12983-2001. Ntotal = number of licensees who fished during the year; N70 = number of licensees that took 70 % of the total catch each year. Source: NT Fisheries logbook data

4.4.2 An equilibrium model for the NT Spanish mackerel fishery

Using a spreadsheet implementation of the methodology described in Chapter 2 (section 2.2.4), an equilibrium model was used to investigate the effects of changing relative vulnerability at age and fishing mortality rate on equilibrium yield and egg production from the fishery. The model was also used to provide inputs for the dynamic age-structured model used below.

Methods

Schedules of weight, w_a, and fecundity, f_a, at age a were derived for fifteen age classes (Table 4.2). Weights at age were estimated as the mean of weights for males and females predicted at each age using the von Bertalanffy parameters and length-weight relationship for east coast Spanish mackerel (McPherson 1992), as in Chapter 2. Although relating to the NT fishery for *S. commerson,* the von Bertalanffy parameters estimated by Buckworth (1998a) were not regarded as suitable, since they were derived from a sample set with high variability in size

at age and too few small animals to adequately define the von Bertalanffy k parameter. Fecundity and maturity at age were predicted from whole weight using the relationships provided by Mackie *et al.* (2003). Note that the values used in Chapter 2 are scaled values of the product of the fecundity and maturity values given here.

The input natural mortality rate, $M = 0.39$ year^{-1}, was estimated from the annual survival rate (0.68) used in Chapter 2, which was derived by assuming that, in an unfished population, 1% of animals would survive to an age of 12 years. The maximum age observed in the NT fishery is 11 years (Buckworth 1998a). This is analogous to the Hoenig's (1983) procedure for total mortality rate estimation. Survivorship at age was calculated as in Section 2.2.4, using this M value and input annual harvest rate U values, the fraction of the vulnerable population removed by fishing,

$$U = Catch / \sum_{a=1}^{a_{max}} N_a v_a w_a .$$

Relative vulnerability at age a, v_a was modelled as

$$v_a = a^m / (a_h^m + a^m)$$

with a_h being the age at 50% relative vulnerability, and m a concentration parameter. Initial values for the equilibrium model of these parameters, $a_h = 4.434$ and $m = 3.838$ were chosen to fit the NT fishery age composition information presented by Buckworth (1998a), using the age structured model (Section 4.4.2 below).

Table 4.2. Schedules of mean weight, fecundity and maturity at age used in the equilibrium and age-structured models of the NT Spanish mackerel fishery.

Age (years)	Weight (kg)	Fecundity (eggs)	Maturity
1	2.188875	45963.66	0.1
2	3.962661	104511.2	0.5
3	5.956252	183701	0.8
4	7.991939	275943	0.9
5	9.946853	373547.5	0.9
6	11.74722	470261.3	0.9
7	13.35657	561708.1	0.9
8	14.76403	645250.4	0.9
9	15.97486	719624.2	0.9
10	17.00357	784542	0.9
11	17.86911	840348.1	0.9
12	18.59188	887753.1	0.9
13	19.19184	927646.1	0.9
14	19.68753	960969.3	0.9
15	20.09553	988641	0.9

Results

The equilibrium analysis suggests resilience to heavy fishing of the Spanish mackerel fishery. The highest yields per recruit (>2.5 kg) were predicted with annual harvest rates $U > 0.45$, when the age of 50% vulnerability was greater than 1 year, and relatively high yields (>2.0) per recruit for $U > 0.35$ and a_h values 3 to 5 years (Figure 4.7). However, this apparent resilience could be misleading due to recruitment impacts of high exploitation rates.

The unfished equilibrium number of eggs per recruit (EPR) was approximately 0.5 million. For a fished stock, EPR was relatively high (>0.4 million eggs) for only very low harvest rates when a_h was also low (Figure 4.7) but as a_h increased, the harvest rates at which high EPR was maintained were high. Thus, at $a_h = 7$ and $U = 0.25$, EPR = 0.4 million eggs. For $a_h > 4$, EPR

was >0.2 million. Combinations of high U and low age at 50% vulnerability ($a_h < 2.5$) led to EPR that was less than 20% of the unfished value (Figure 4.7).

Mean weight of fish in the vulnerable population at unfished equilibrium was calculated as 10.28 kg. Mean weight was not sensitive to U, within any narrow range of a_h (Figure 4.8). However, mean weight was sensitive to a_h, so that a reduction in the age at 50% vulnerability by 1-1.5 years would cause a reduction in mean weight in the catch of about 2 kg (Figure 4.8).
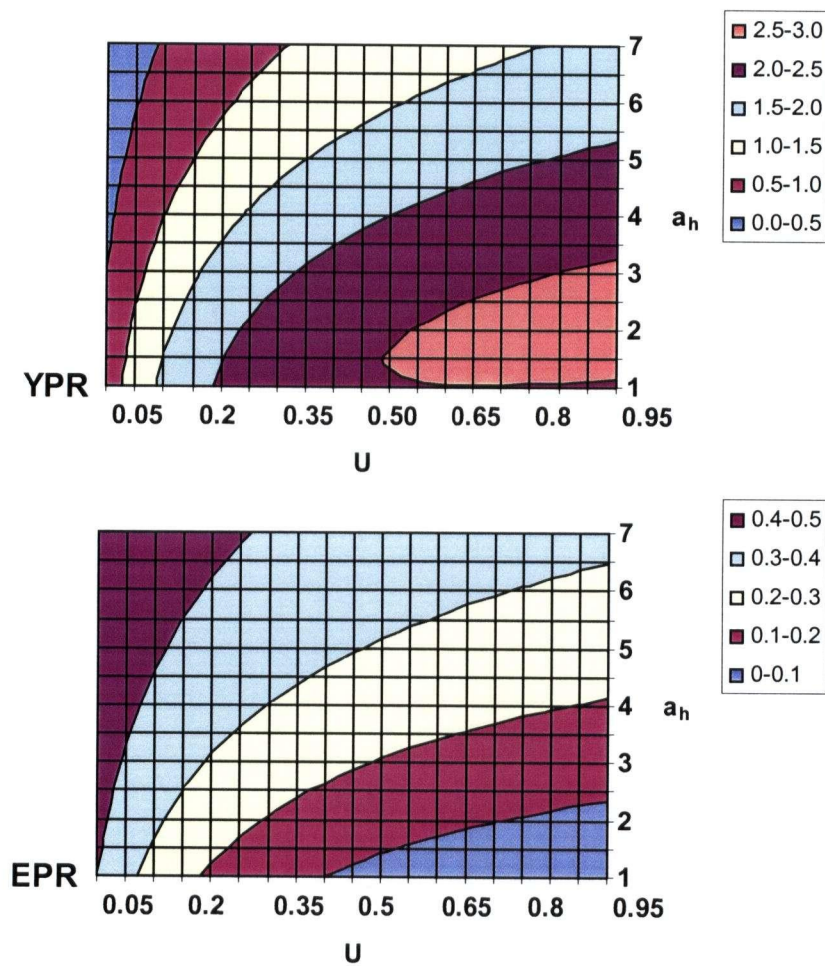
Figure 4.7. Yield per recruit (**YPR**) and Eggs (millions) per recruit (**EPR**) as functions of harvest rate U and the age at 50% vulnerability, a_h.
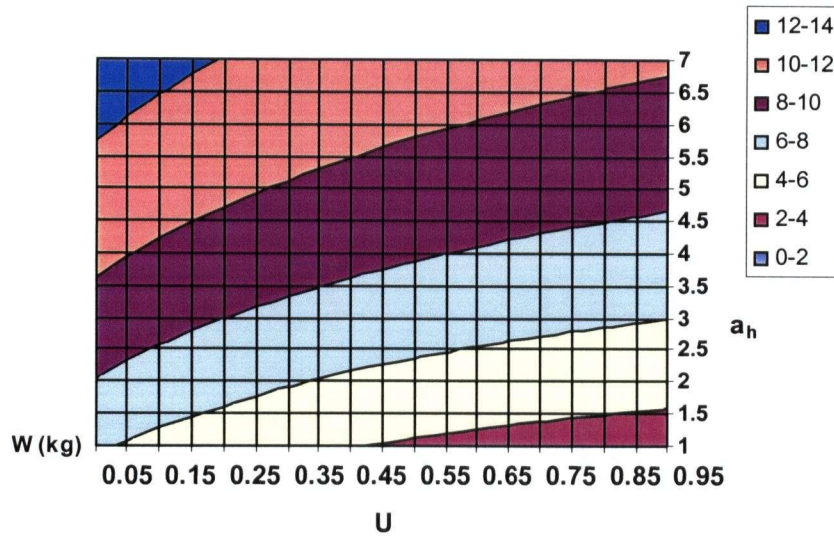
Figure 4.8. Mean weight in catch predicted from equilibrium analysis, as a function of harvest rate(U) and age at 50% vulnerability, a_h.
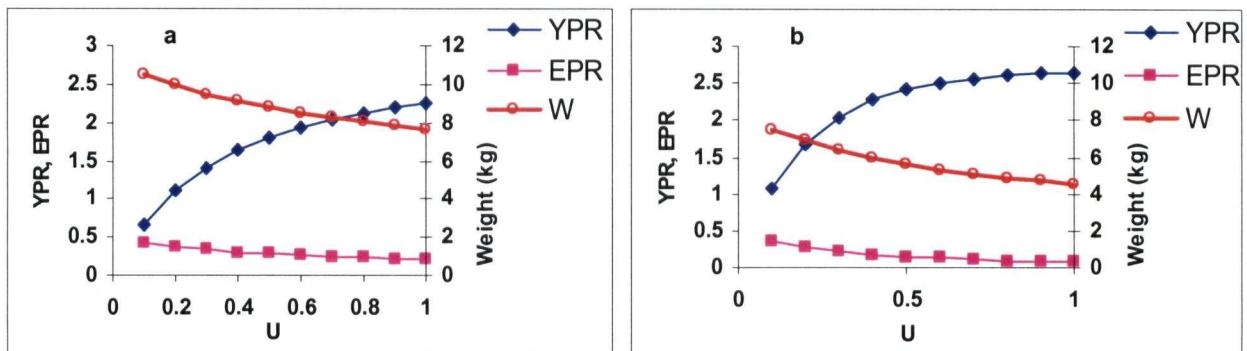
Figure 4.9. Comparison of Yield per recruit (YPR), Eggs per recruit (EPR) and mean weight, **a**. a_h = 4.434; **b**, a_h = 2.000.

Reduction of the age at 50% vulnerability to a_h = 2.000 years (Figure 4.9b) had the effect of increasing the initial slope of the yield curve, so that YPR at $U = 0.1$ was 1.1 kg, 2.4 kg at $U = 0.5$, and 2.7 kg at $U = 1.0$. Over the range of U from 0.1 to 1.0, EPR declined steadily from 0.50-0.06 million, a substantially greater reduction than for the higher a_h value. The equilibrium mean weight for a_h = 2.000 years declined steadily from 7.5 kg at a harvest rate of

0.1, to 4.5 kg for $U = 1.0$, substantially lower than for the corresponding values at $a_h = 4.434$ years (Figure 4.9b).

4.4.3 An age-structured dynamic model of the NT Spanish mackerel fishery[3]

Population model

The age-structured model was a single stock, single area model, with an annual time step, so that this model was a simplified version of the reference population model of Chapter 2, and implemented as a spreadsheet. The dynamics of older fish were described by

$$N_{a+1,t+1} = N_{a,t}.\exp(-v_a F_t - M), \text{ for } a < 15.$$

The 15 year class was treated as a plus class so that

$$N_{15,t+1} = N_{14,t}.\exp(-v_{14}F_t - M) + N_{15,t}.\exp(-v_{15}F_t - M).$$

Tabulated schedules of vulnerability, mean weight, fecundity and maturity at age were as assumed for the equilibrium model. The fishing mortality rate was calculated $F_t = -\ln(1-U_t)$, with $U_t = Catch_t / \sum_{a=1}^{a_{max}} N_a v_a w_a$. The initial recruits, R_0 and subsequent annual recruits to the population, $N_{1,t}$, were calculated using the Beverton-Holt S-RR, parameterised with leading parameters K_r and B_0 (recruitment curve slope measured as compensation ratio, and unfished biomass) as in Chapter 2 (Eq 2.3 and Eq 2.4).

As in Chapter 2, it was assumed that fishery began at an unfished equilibrium, (in 1970 in this case).

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[3] Previous versions of this model have been used to conduct assessments of the Spanish mackerel fisheries of NT (Walters and Buckworth 1997), and Queensland (Hoyle 2002; Welch *et al.* 2002; Hoyle 2003) Spanish mackerel fisheries.

Catch Data Series

For the domestic commercial fishery it was assumed that the catch in 1963 was 10 tonnes, and that catches grew by 10% each year until 1982 when the more precise logbook information described earlier in this chapter first became available. It was assumed that the recreational catch was 10% of the commercial catch until 1980 but that the proportion of the commercial catch grew by an additional 1% of the commercial catch until 1995. Thus the recreational catch was estimated as 11% of the commercial catch in 1981, 12% in 1982, and so on, to 28% of the commercial catch in 1994. The survey estimate of recreational catch in 1995 represented 31% of the commercial catch.

Recreational fishery catch estimates for the years 1996 to 2000 were made by interpolating between the survey estimates of 1995 and 2001. Indigenous catches were assumed to have been the same fraction (6.1%) of recreational catches throughout.

Two separate series of catch information for the Taiwanese sector of the fishery were alternately used in assessments. The first, hereafter "Taiwanese A" was based on the declared Spanish mackerel catches from logbooks (Table 4.1), but creating a catch time series for 1974 to 1986 by assuming that between 1974 and 1979, the Taiwanese sector annually took the same catch as in 1980. For the second series, "Taiwanese B", I included the declared total mackerel catch (Table 4.1), assuming this was more representative of the Spanish mackerel catch by the Taiwanese fleet, again extending the series back to 1974 using the 1980 value.

Abundance and stock composition indices

Three sets of information were employed as abundance and composition indices in the assessment: age composition of the troll fishery catch for 1991 to 1993 was derived from

Buckworth (1998a); catch (kg) and effort (expressed both as boatdays and linedays) from the NT Fisheries logbook system were used to compose catch per unit effort (CPUE) series; and, mean weight data were available from a subset of the NT Fisheries logbook data for 1994 to 2001.

To overcome small sample sizes, the age composition data for 1991-1993 were amalgamated to form one set of proportions of each age in the catch. Fish ranged in age between 1 and 11 years, but those aged 3 to 6 years were the bulk (76%) of the catch (Figure 4.10).

Rather than the gross annual catch rates (Figure 4.5), I calculated "spatial" CPUE sets, with effort measured as either boat days or line days, as abundance index time series. As described in Chapter 2, these were developed by dividing the sum of catches in each grid for each month (where that grid was fished), by the total effort in that grid for that month. The annual abundance index was then the mean over the year for all months and grids fished in the year.
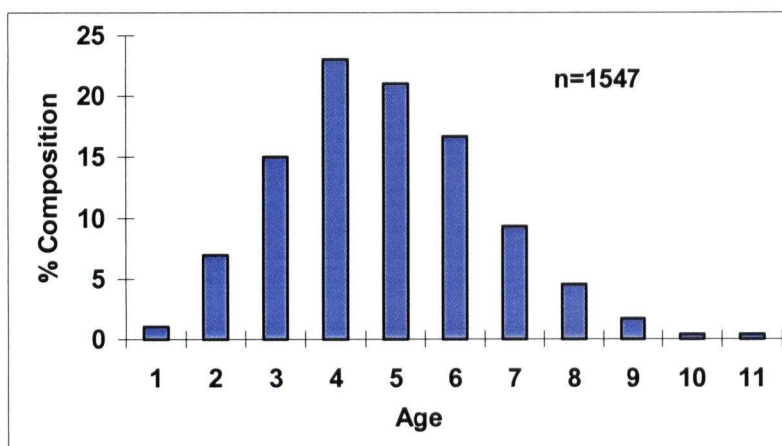
Figure 4.10. Age composition of the NT Spanish mackerel troll fishery catch, 1991 to 1993. Data from Buckworth (1998a).

CPUE with effort measured as boatdays (CPUEB) showed a strong positive trend through the time series, increasing from less than 100 kg boatday^{-1}, to around 300 kg boatday^{-1}, by the end of the 1990s (Figure 4.11). CPUE with effort measured as linedays (CPUEL) also showed an upward trend, but a period of high values was again very apparent for 1986 to 1989 (Figure 4.11). Both series strongly reflected the gross catch rate data (Figure 4.5).

Voluntary logs, in which fishers record activities per fishing session, have also been in operation in the troll fishery since 1994. In addition to the total weight of product (eg weight of fillets or trunks as above), these indicate the number of fish taken in a fishing session. Mean catch weights were estimated for each session recorded in the logbook data set, then the mean of these was calculated for each year from 1994 to 2001. These data accounted for 28% to 73% of the landings for the troll fishery (Table 4.3). Estimated mean weight of fish in the catch (Table 4.3) has ranged between 6.9 kg in 1999 to 8.0 kg in 1995.
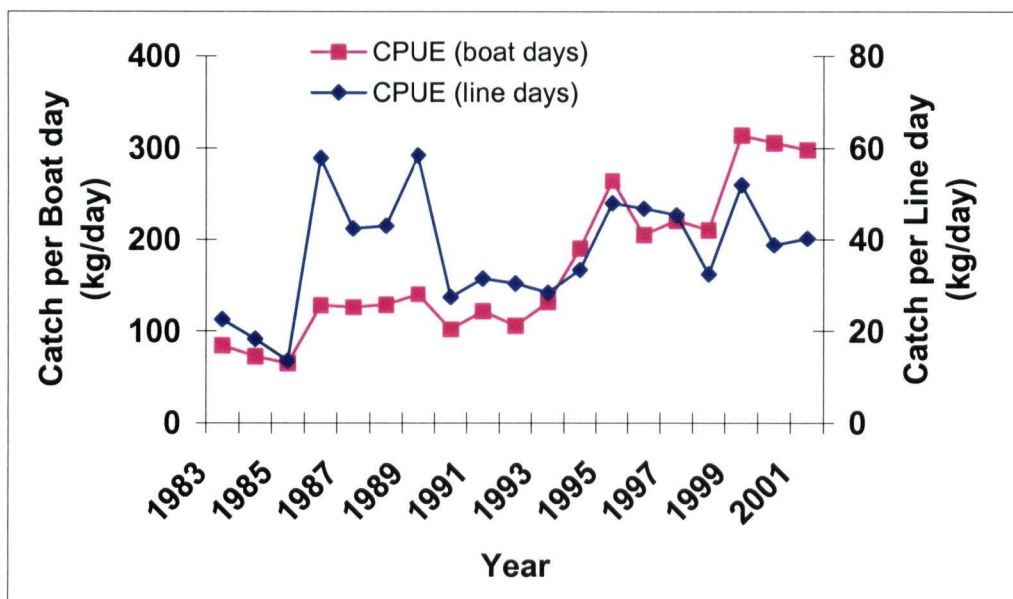
Figure 4.11. Spatial catch per unit effort abundance index time series for the NT Spanish mackerel troll fishery, 1983-2001.

Table 4.3. Mean weights calculated from catch numbers and weights provided in NT fisheries logbook (voluntary) data records, 1994 –2001.

Year	Catch Weighed (tonnes)	Mean weight (kg)	s.d.	No. of sessions	Total Catch (tonnes)	% of catch sampled
1994	39.9	7.4	0.15	261	129.9	30.7
1995	38.9	8.0	0.14	234	140.7	27.6
1996	40.4	7.7	0.16	281	172.2	23.5
1997	83.0	7.7	0.17	305	181.2	45.8
1998	79.2	7.7	0.25	375	162.3	48.8
1999	112.7	6.9	0.12	468	233.5	48.3
2000	164.7	7.2	0.16	874	227.2	72.5
2001	196.3	7.1	0.10	920	373.1	52.6

Likelihoods

The leading variable in the assessment was the unfished biomass, B_0. A uniform prior over the range 1000 to 10000 tonnes was applied throughout, although some parameter searches were also undertaken for the wider range 1000 to 50000 tonnes. For each abundance and composition index series, the Solver routine of Excel was used to maximize a likelihood by varying B_0. For the CPUE indices and mean weight indices, likelihood kernels and posterior probabilities given those series were calculated using the methodologies of Walters and Ludwig (1994) and Walters (1998b), as described in Chapter 2. For CPUE-based assessments, the power model described in Chapter 3 (Eq 3.1) (Walters 1998b) was also employed.

This necessitated that the calculation

$$z_t = \ln(Y_t) - b\ln(B_t) \qquad \text{Eq 4.1}$$

was substituted for $z_t = \ln(Y_{k,t} / B_{k,h,t})$ (Eq 2.40) in the likelihood estimation (Walters 1998b). I set $b = 0.5$.

For the age composition data, B_0 was estimated so as to maximize a multinomial log-likelihood kernel

$$LL(B_0 \mid A) = \sum_{a=1}^{a_{max}} \left[x_a \ln(p_a) \right] \quad \text{Eq 4.2}$$

A represents the set of age composition observations, x_a is the number of fish of age a in the sample, p_a is the expected proportion of fish age class a under the model for that value of B_0, and a_{max} is the maximum age class in the population.

Relative vulnerability at age parameters

An initial requirement was the choice of suitable values for the parameters a_h and m for the relative vulnerability at age model, for all assessments. This was made by allowing these parameters to vary for a range of feasible B_0 values, first choosing the values of a_h and m which maximized the likelihood (Eq 4.2) for the age composition data, for each of these B_0 values. The a_h and m finally adopted were those which corresponded to the maximum of these log-likelihood values.

For the catch series including Taiwanese catches specified as Spanish mackerel, and over a series of B_0 values ranging over 1300 – 3900 tonnes, the maximum likelihood values were $a_h = 4.434$ and $m = 3.838$, which occurred at $B_0 = 1300$ tonnes. Conditional maximum likelihood estimates of a_h declined and of m increased as B_0 increased; however; these changes in a_h and m were small, with the corresponding values at $B_0 = 3900$ tonnes being $a_h = 3.370$ and $m = 4.350$.

Assessment based on age composition data

Alternative assessments were carried out for catch histories differing in the assumed magnitude of the Taiwanese catch. The first catch history series (Appendix Table 4.1) included the Taiwanese A catch series. The second included the Taiwanese B series. For analysis of the Taiwanese A catch series, it was necessary to constrain B_0 to a minimum of 1300 tonnes, to ensure that catches did not exceed vulnerable biomass in any year. Given this constraint, the most likely B_0 value given the age data was the minimum feasible, 1300 tonnes, with the likelihood of higher B_0 values declining steeply (Figure 4.12). For B_0 at 1300 tonnes, there was a very close agreement between predicted and observed age compositions (Figure 4.13), but catch rates did not track vulnerable biomass levels, particularly for the latter half of the time series, suggesting inconsistency between the age composition and catch rate information (Figure 4.16).

The highest probability B_0 given the spatial CPUE abundance index based on effort as either boat days or line days was again at the minimum feasible, 1300 tonnes (Figure 4.14). However, for the series based on line days, comparable relative probability values were also generated by a large range of much higher B_0 values (Figure 4.14b), indicating a lack of fit by the model to this series.
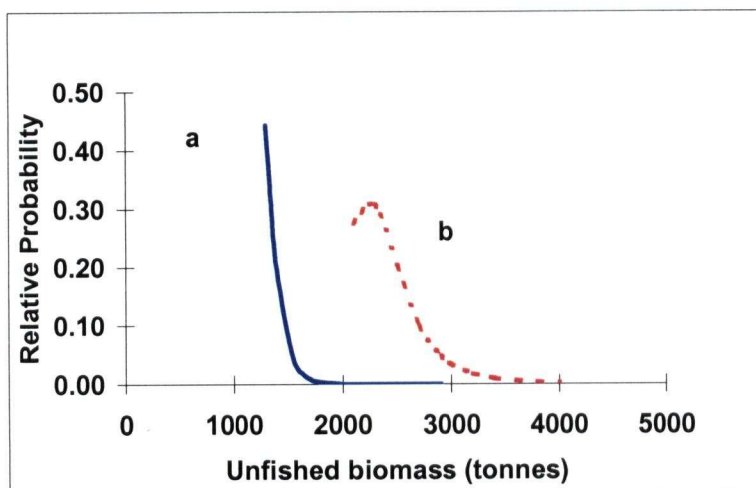
Figure 4.12. Relative marginal probabilities for unfished biomass (B_0) given the age composition data, **a**, includes the catch series Taiwanese A; **b**, includes the catch series Taiwanese B.
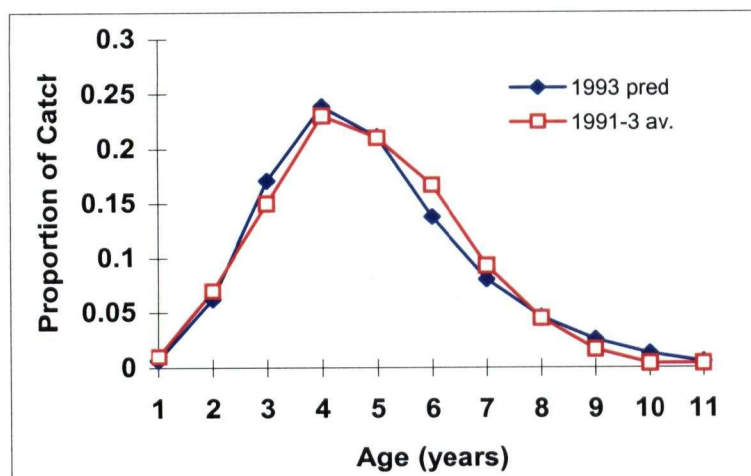
Figure 4.13. Age composition of the NT Spanish mackerel troll fishery catch for 1991-1993: observed proportions and those predicted under the model.
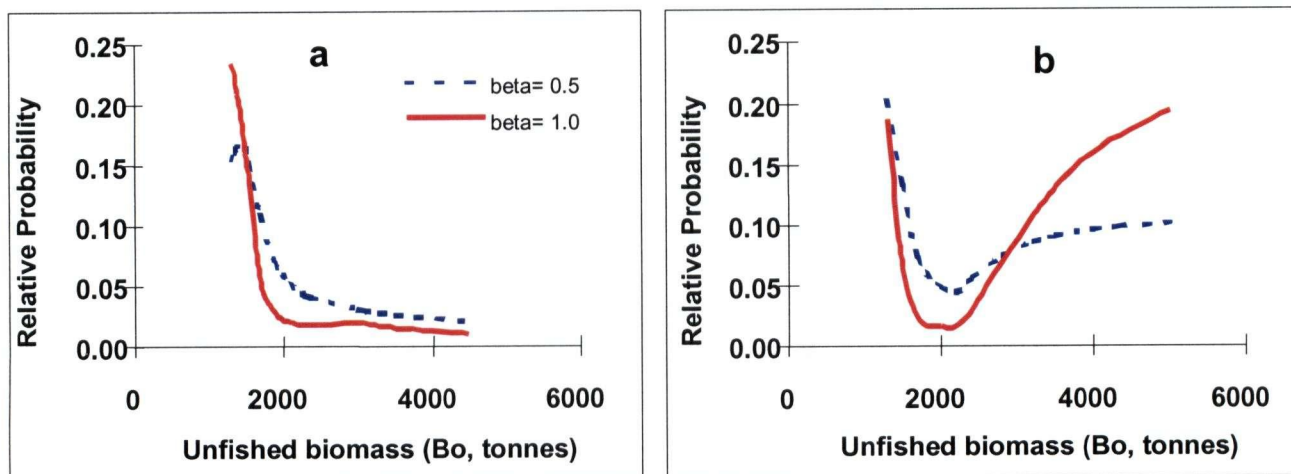
Figure 4.14. Relative probability of B_0 values for **a**, spatial CPUE based on effort as boatdays; **b**, spatial CPUE based on effort as linedays.

Mean weight as a measure of abundance also suggested that the unfished biomass of the fishery was small, the most probable value for B_0 being 1396 tonnes. The fit to the index series was, nevertheless, relatively poor with much larger values of B_0 having significantly high probability (Figure 4.15). Mean weight values in the index series (Table 4.3), with a mean of 7.46 kg, were substantially less than the predicted weight series for 1994 to 2001, with a mean of 9.27kg. The latter was in agreement with mean weight of the 1991-93 age sample, which predicted from the sample numbers at age applied to the weight at age schedule of Table 4.2 was 9.40 kg (SE=0.080). Resetting a_h to values of between 3.0 and 4 improved the agreement of the data series with the model predictions of mean weight, and had the effect of shifting the relative probability profile to the right: at $a_h = 3.2$, the maximum likelihood estimate of B_0 was 1701 tonnes, but the profile was similar (Figure 4.15).

Given an unfished vulnerable biomass of 1300 tonnes, the combined sectors reduced the vulnerable biomass of NT Spanish mackerel to around 30% of B_0 in the mid 1980s. The fish stock rebounded from this low point with the departure of the Taiwanese fleet, but growing levels of fishing by the combined Australian sectors were such that by 2001, the vulnerable
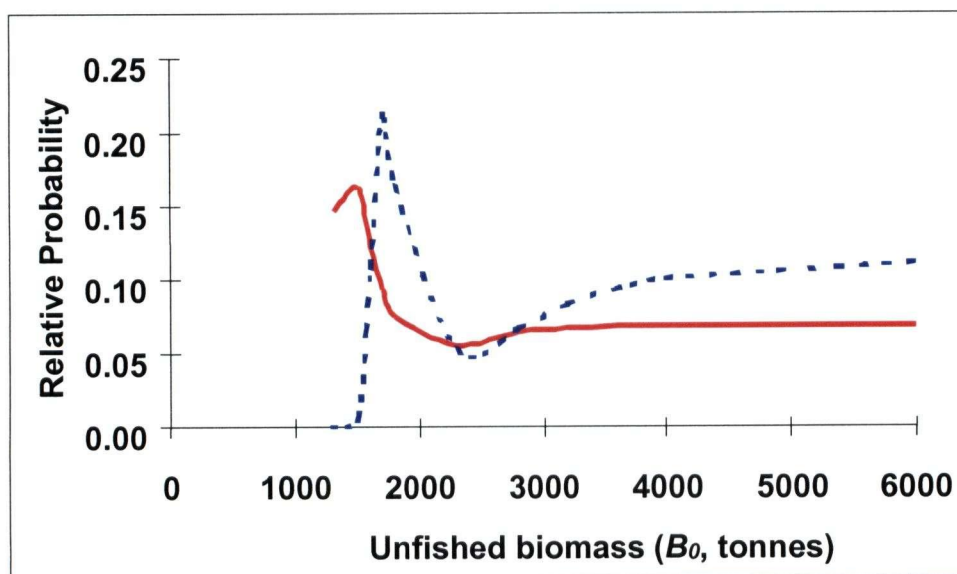
Figure 4.15. Relative probability of B_0 values for mean weight abundance index series, 1994-2001. Profiles are for a_h = 4.434 (solid line) and a_h = 3.2 (dashed line).

biomass was again reduced to around 30% of the unfished level. At each of these points in the fishery's history, the annual harvest rate is estimated to have exceeded 0.8. Projecting forward, if the 2001 harvest rate, $U = 0.84$, ($F > 1.2$) is maintained, then 2001 catches are not sustainable but will decline steadily to around 280 tonnes through 2002-2010, declining further during the subsequent decade. A reduction in effort to so that $U \approx 0.5$, thereby reducing the 2002 catch to around 200 tonnes, would generate a slowly increasing population and catch, with a maximum sustainable catch of around 250 tonnes. The effect of higher harvest rates would be initially higher catches, rapidly dwindling to around 250 tonnes annually (Figure 4.17).
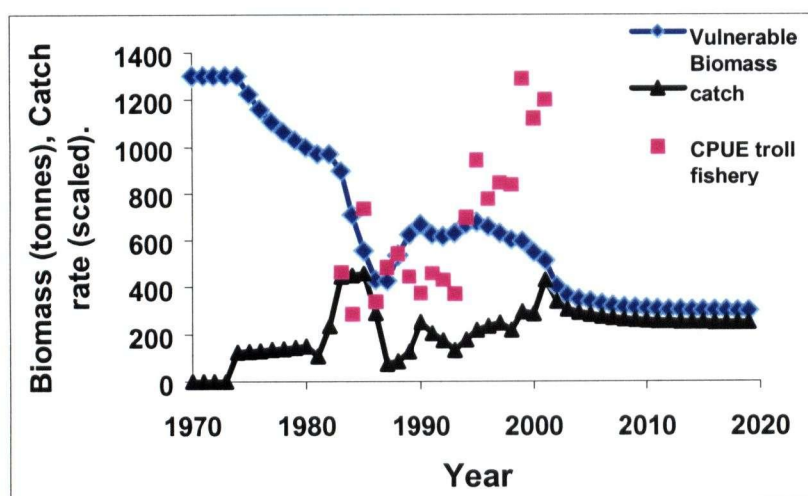
Figure 4.16. Vulnerable biomass trajectory estimated from age-structured model fitted to age composition data, (scaled) catch rate time series, and catch history for Spanish mackerel in NT waters, including the Taiwanese A series. Projected catch and biomass assuming $U = 0.84$.
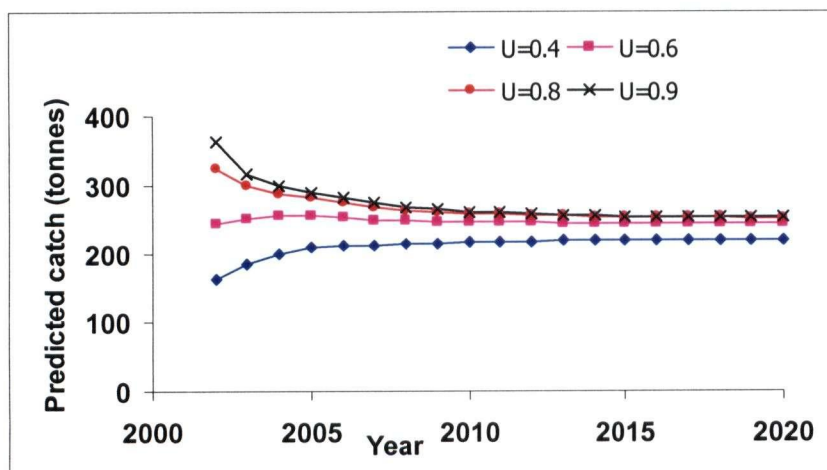
Figure 4.17. Predicted 2002-2020 catches for the NT Spanish mackerel fishery under different annual harvest rate (U) levels, assuming a B_0 of 1300 tonnes.

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The immediate effect on the modelled fishery of specifying a series of larger catches for the Taiwanese sector (using the catch history including the Taiwanese B series) was that the unfished vulnerable biomass was necessarily a minimum of 2300 tonnes, to avoid catches greater than the vulnerable biomass at any time. Again the most probable value of B_0 was the minimum feasible, 2300 tonnes in this case (Figure 4.12). This value resulted in similar close agreement to the age composition data as the previous catch history scenario (plots were virtually identical to Figure 4.13 and so are not shown), but also resulted in abundance trends that did not track catch rates closely.

Under this scenario, the impact of the Taiwanese fishery was again reduction of the biomass to about 30% of unfished levels; the population grew again after their departure, until the mid-1990s. The 2001 U was estimated as 0.29 ($F = 0.34$), which, if maintained, would produce continued annual catches of around 350 tonnes over the next 2 decades. Increasing harvest rates would initially increase catches but these would rapidly dwindle to a sustainable level of between 400 and 500 tonnes, as indicated in Figure 4.18.
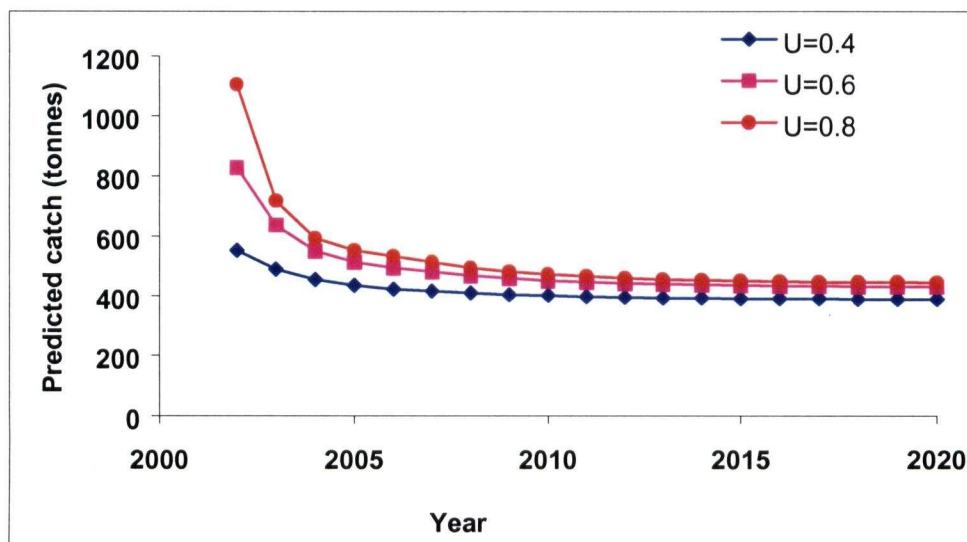
Figure 4.18. Predicted 2002-2020 for the NT Spanish mackerel fishery under different annual harvest rate (U) levels, assuming B_0 = 2300 tonnes.

4.5. Discussion

The central policy problem addressed by this assessment is whether the current impact of fishing on the NT Spanish mackerel stock is sustainable. Given the deliberately "conservative" start to its management, whether this fishery can be allowed to grow (whether the limit reference point for annual catches of 450 tonnes per year might be revised upward) is a continual query. The most direct interpretation of the results of this assessment of NT Spanish mackerel fishery is that it consists of a fairly small stock that was fished down by the combined Taiwanese and Australian sectors in the late 1970s and early 1980s but it recovered substantially, to again be fished at very high levels by 2001. When a catch history that contained the catches declared as Spanish mackerel in the Taiwanese sector log books was used to drive the model, the same conclusion resulted from three different indices of abundance and composition: that the fish stock was small, and that though high levels of fishing mortality could be maintained, catches taken in this most conservative scenario would be sustainable at around 250 tonnes in the long term.

Use of the model with a less conservative catch history, based on the assumption that the declared Taiwanese catch of all mackerels was more indicative of their Spanish mackerel catch than the amount they specifically declared, provided similar fit to the age composition data, and indicated that the fish stock is nevertheless relatively small. Under this scenario current harvest rates are moderate, but if maintained would result in a series of declining catches as the vulnerable biomass was fished down, to a long term sustainable annual catch of around 350 tonnes annually. Larger harvest rates of $U = 0.6$ ($F > 0.9$) or more, initially producing large catches of more than 800 tonnes annually, would yield no more than 450 tonnes annually in the long term. The limit reference point currently in place matches this scenario.

The fishery status indicated by these assessments is moderately to heavily fished. Harvest rates at the level reached in 2001 are not likely to lead to the collapse of the fishery, but catches may decline to a lower sustainable level, depending on the initial and current size of the fish stock. Previous assessments of the fishery, that did not include the Taiwanese catch history (Walters and Buckworth 1997) noted poor fit of the model to age composition data, including an apparent lack of older animals in the age composition.

The inclusion of the Taiwanese catch series improved the fit of the model to the age composition information, even for the smaller series of catches actually declared as Spanish mackerel (noted in a further assessment in 2000 by Walters and Buckworth, unpublished). Nevertheless, the catch series here are necessarily inaccurate – assumptions were made to provide catch series for the period of the 1970s when, it is known, that significant catches of Spanish mackerel were taken by the Taiwanese and domestic fishing fleets. I have assumed that those catches were similar or smaller than the years of the 1980s for which there are available data.

The size of the Taiwanese sector catches are a continual source of conjecture, as it is recognized that if they were substantially larger, this could lead to revision of the "size" of the fishery, as measured by the limit reference point, upwards. It was also necessary to make assumptions about recreational and indigenous catches, and I have ignored the loss of fish to illegal activities, which at times are rumoured to be significant. However, without further information, the actual magnitude of all these unaccounted catches remains conjectural. The problem of 'sizing' the fishery (Perry *et al.* 1999) remains.

There are many additional uncertainties, some of which have been ignored in the assessment models by specifying fixed values of parameters, or relating variables determininistically (e.g.

recruiment was deterministically related to spawning biomass), and thus lending apparently high precision to the estimates of B_0. Overdispersion in the age composition data (variability is under-estimated) may also have lent apparently greater precision to the assessment based on the age composition data. There were also apparent conflicts within the data that indicate either sampling issues or model mis-specification. Thus, the relative vulnerability at age model was parameterised to fit the set of age composition observations for 1991-1993. When the assessment model was used with this parameterisation, mean weight data did not match predicted weights well. This indicates that either the relative vulnerability at age schedules have changed over time (age composition data were for 1991-1993, the mean weight series was for 1994-2001), and/ or that the age structure data were not representative of the age composition of the vulnerable population. Both of these possibilities are reasonable. It is also possible that the mean weight time series is not representative of the catch. There was a substantial difference between the mean weight estimated from the age composition sample representing 1991-1993 (9.40 kg) and the mean weight estimated from log book information in 1994 (7.4 kg). Although the mean weight time series annually represented 28% to 72% of the catch, and so might have substantial credibility, the small size of the fishery means that fishing practices of individuals could also bias these data.

The relative vulnerability at age schedule is a product of both the fishing gear applied, and the behaviour of the fishers. Minor changes in fishing operations such as changes in standard rigs could change the average schedule, as could, in a small fishery, the entry of a new operator who preferred to catch smaller fish, or work safer inshore grounds. The schedule could change very rapidly if, for example, a lucrative niche market were to develop for a particular size range of fish. Additionally the model assumed that the relative vulnerability at age schedules applied to all sectors of the fishery.

Age composition in catches does appear to vary spatially, so that both sampling error and fisher spacing behaviour could be in effect. Further collection and analysis of age composition data and mean size information would be valuable to address some of the questions raised in this analysis, and to monitor and rapid changes in age/size composition and in relative vulnerability at age. Analysis should examine the potential spatial and temporal sampling problems.

It is of interest that the assessment using the series of CPUE in boatdays fit the model, while the CPUE linedays series did not. Linedays would intuitively appear to better reflect fishing effort; however, this may not be so if effort is a mix of search processes occurring at different spatial scales. Locating concentrations of fish, "being in the right place at the right time", might reflect boatdays of searching. CPUE based on boatdays would thus reflect abundance. CPUE based on linedays may measure more effectively the rate of capture of fish once concentrations have been located. Another way to express this is as a difference in two operational spatial scales. The larger scale is a fishery wide scale, applying to the search for grounds that have concentrations of fish (boatdays effort measure this), and the finer scale relates to the location and capture of fish on that ground (measured by effort as linedays). This is an area of the fishery's dynamics that could be examined further.

Different choices of either of the input parameters, the natural mortality rate, M, and the increase in recruitment success at low population sizes, K_r, could substantially change the predictions of the assessments. Although the values used for these parameters were considered as reasonable for *S. commerson* in the NT fishery, the parameters were nevertheless very uncertain. M was empirically derived here, based on the oldest age observation in the age composition data. Another empirical approach, the Pauly (1980) method, with the von Bertalanffy parameters used here and applied to mean temperatures of $26°$ C to $28°$ C, generates

similar values for natural mortality $(0.40 \text{ year}^{-1} < M < 0.42 \text{ year}^{-1})$. Similar values for M were estimated for Kimberley region (Mackie *et al.* 2003) and east coast (Hoyle 2002) *S. commerson*. However, alternative growth parameter values would generate a corresponding alternative set of M predictions using Pauly's method. For example, applying the parameter values for a von Bertalanffy function with $t_0 = 0$, from the NT age composition data (Buckworth 1998a), indicates $M > 0.8 \text{ year}^{-1}$. Natural mortality values for *S. commerson* and other *Scomberomorus* in the literature range from $M = 0.16 \text{ year}^{-1}$ (for Pilbara, Western Australia males, Mackie *et al.* 2003) to $M = 1.23 \text{ year}^{-1}$ (Philippines, Pauly *et al.* 1984) but are usually in the range 0.3 year^{-1} to 0.8 year^{-1}. A substantially lower real M value would imply a larger unfished population biomass but a lower sustainable catch for the NT fishery. It would also be necessary to explain the rate of disappearance of older fish in the catch as some combination of a larger historical catch, and reduced vulnerability of older fish, due to their movement away from the fishery. A higher M would imply a lower biomass but higher sustainable catch. Clearly, the assumed value of M is another major source of uncertainty in this assessment.

In assessments by Hoyle (2002) of the east coast fishery for the species, the model was also used to estimate K_r. From that assessment, K_r was also poorly defined but the maximum probability value was near the high end of the prior distribution, which was bounded above at $K_r = 7$. The value I employed in this assessment, to represent a moderately productive fish species, was taken in the mid-range of values inferred from the analyses of Myers *et al.* (1999). Lower values than that used here would imply a correspondingly less productive population, less resilient to fishing, while higher values would imply one that is correspondingly more productive and resilient. However, I emphasise that any resilience indicated by either the equilibrium or dynamic models may be illusory under heavy fishing pressure, as they do not

account for fine scale population structure effects (see below), or depensation arising from ecological interactions, such as cultivation effects (Walters and Kitchell 2001).

The only documented case of the collapse of a *Scomberomorus commerson* fishery of which I am aware occurred in Omani waters, where catches exceeded 27000 tonnes in 1988 but dwindled to as low as 3265 tonnes in 1993 (Siddeek and Al-Hosni 1998). With several thousand participants using predominantly driftnets, a substantial proportion of the catch in this fishery was fish less than two years of age (Dudley *et al.* 1992). This combination of factors, very high participation and effort levels, combined with the capture of very young fish, is unlikely to occur in NT waters.

A major structural shortcoming of the model was that it could not account for spatial stock structure and dynamics. As well as the fact that it ignores borders and potential exchanges of fish from the NT not only with neighbouring Western Australia, Queensland, but also Indonesia and East Timor, a strong lesson from Chapter 3 was that spatial dynamics on fine scales can have substantial impact on the course of a fishery, and on the value of different monitoring and management controls.

Fine scale dynamics are important. In the case of Spanish mackerel, the potential fine-scale spatial structuring that has been revealed by the recent otolith isotope chemistry (Newman *et al.* 2003) and parasite incidence analyses (Lester *et al.* 2001; Moore *et al.* 2003), coupled with the strong cost gradients that must exist in a fishery that stretches across the most sparsely-inhabited coast in Australia, creates concerns about the potential for serial depletion problems and generation of very high fishing mortality rates on local scales, as the species is fished in aggregations. There is potential for effort to be concentrated on these aggregations, with unpredictable effect. If *S. commerson* populations are indeed structured as metapopulations,

then interchange between demes of recruits and adults will ultimately determine the effects of concentrated fishing. The fishery may be less resilient than the equilibrium analyses indicated.

The NT Spanish mackerel troll fishery is currently managed by input controls, limited by a total output, a total allowable catch (TAC) in effect. A limit that specifies catch is currently necessary for this fishery, as there are no means by which F in the fishery could be measured against an F_{opt} target.

In the most conservative assessments in this chapter, the current TAC would exceed catch values that are sustainable. For populations hypothesised in those assessments, effort at 2001 levels should generate reduced catches in response to reduced vulnerable biomass. This may not be apparent if there is a change in the relative vulnerability at age schedule, as a consequence of fishers targeting smaller and younger fish to maintain catch rates.

There is capacity in the fishery to drive substantial increases in effort, through greater participation and increasing the number of days and hours fished per operator; a threefold increase is certainly feasible, and there is always the potential for cryptic increases in effort that are due to improved technical efficiency, particularly in the search process; fishers in the NT Spanish mackerel troll fishery do not currently use sonar, for example. This has the greatest danger if the fishery were to change to output controls and, as in Chapter 3, catches were maintained through uncontrolled effort increase, to the point of fishery collapse.

The assessment results were dependent on the magnitude of the catch history, which is largely uncertain. The fishery may nevertheless be resilient to heavy fishing pressure. But there are warnings: possibly reflecting a metapopulation structure, *S. commerson* spatial dynamics could create the problems described for some populations in Chapter 3, and I emphasise that the

population reference model in that chapter was parameterised for Spanish mackerel populations in varying spatial configurations. While it is possible that better data could be sought on historical catch values, addressing one source of uncertainty in the current assessment, a much more effective and ongoing solution to the problem of ascertaining the status of this fishery would be to use a method of directly monitoring fishing mortality rates. Application of such a method could then allow management to move away from the potentially dangerous TAC limit (Chapter 3; Walters and Martell 2002).

In Chapter 3, I demonstrated that a mark-recapture program could be very effective in monitoring spatially complex fisheries, even those in which spatial structure was poorly known and did not match administrative boundaries. Such a program would also provide insight into the fine-scale dynamics of the fishery, creating potential for use of direct harvest rate controls. For a metapopulation structure, innovative approaches such as rotating spatial closures may even be future management options.

For Spanish mackerel conventional tagging programs, it is thought that tag loss (tag shedding and mortality) due to the capture and tagging process, is high. The only published accounts of tagging experiments with *S. commerson* produced lower than expected recapture rates ($\approx 2\%$ or less for experiments by McPherson (1992), and around 1% in the experiments by Mackie *et al.* (2003). In the following chapters I examine the feasibility of monitoring the NT Spanish mackerel fishery using a mark-recapture approach based on genetic tagging.

4.6. Appendix - Catch time series for assessment of the NT Spanish mackerel fishery

Appendix Table 4.1. Catch time series used for assessment of the NT Spanish mackerel fishery

Year	Troll Fishery	Other Commercial Fisheries	Recreational Fishery	Indigenous Fishery	Taiwanese Fishery - Spanish mackerel	Total	Total[1]
1974	28.5		2.9	0.2	92.9	124.5	349.4
1975	31.4		3.1	0.2	92.9	127.7	352.5
1976	34.5		3.5	0.2	92.9	131.1	356.0
1977	38.0		3.8	0.2	92.9	135.0	359.8
1978	41.8		4.2	0.3	92.9	139.2	364.0
1979	45.9		4.6	0.3	92.9	143.8	368.6
1980	50.5		5.1	0.3	92.9	148.9	373.7
1981	55.6		6.1	0.4	47.1	109.2	148.5
1982	61.2		7.3	0.4	166.3	235.3	397.1
1983	121.5	6.6	16.7	1.0	298.9	444.6	600.5
1984	43.3	22.4	9.2	0.6	376.6	452.1	658.4
1985	61.2	59.3	18.1	1.1	318.5	458.2	560.6
1986	24.9	19.3	7.1	0.4	239.7	291.4	869.8
1987	37.5	26.1	10.8	0.7		75.1	75.1
1988	60.1	11.1	12.8	0.8		84.7	84.7
1989	53.0	53.2	20.2	1.2		127.5	127.5
1990	190.1	17.7	41.6	2.5		252.0	252.0
1991	151.1	14.7	36.5	2.2		204.5	204.5
1992	129.4	9.5	33.3	2.0		174.3	174.3
1993	93.3	11.2	27.2	1.7		133.4	133.4
1994	129.9	6.9	38.3	2.3		177.4	177.4
1995	140.7	23.8	51.4	3.1		219.1	219.1
1996	172.2	10.5	50.2	3.1		236.0	236.0
1997	181.2	15.2	49.0	3.0		248.5	248.5
1998	162.3	6.3	47.8	2.9		219.4	219.4
1999	233.5	14.0	46.7	2.8		297.0	297.0
2000	227.2	12.1	45.5	2.8		287.5	287.5
2001	373.1	12.3	44.3	2.7		432.4	432.4

[1]Including the series of all mackerel catches declared by the Taiwanese fishery logbooks.

5. DESCRIPTION AND DEVELOPMENT OF A SIMPLE DEVICE FOR REMOTE TISSUE COLLECTION[4]

5.1. Introduction

Advances in molecular genetic technology used in population biology for identification of individuals, i.e. for "tagging" samples of individuals, mean that there are now many potential applications that require collection of small amounts of tissue from animals sampled from populations of interest. Such applications of molecular genetics range from mark-recapture studies in terrestrial mammals (Taberlet and Waits 1998; Taberlet and Luikart 1999), through analyses of patterns of relatedness and bioaccumulation in marine mammals (eg Fossi *et al.* 1997), to studies to ascertain stock structures in fish (eg Blandon *et al.* 2001; Broughton *et al.* 2002). Genetic mark recapture has recently seen preliminary use in marine populations, providing information on, for example, stocking success in sciaenid fishes (Ward *et al.* 1995), movement of whales (Palsboll *et al.* 1997), tag loss in lemon sharks (Feldhem *et al.* 2002) and growth in holothurians (Uthicke and Benzie 2002).

Frequently, the collection of tissue in fish population studies is facilitated by sampling from commercial catches. However, there are many cases where capture is not desirable. This may be for economic reasons (eg. catching sufficient animals is costly), or where behavioural effects or mortality caused by capture are regarded as inappropriate. The latter is typically the case for species with a high conservation importance where, for terrestrial species, the minimal tissue

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[4] The devices described here are subject to an International Provisional Patent held by Aquatic Remote Biopsy P/L, Australia. No University of British Columbia resources were used in their development.

requirements of molecular genetics have enabled non-invasive approaches to be developed for

genetic mark-recapture (Taberlet and Waits 1998; Taberlet and Luikart 1999; Mills *et al.* 2000),

and for population structure in marine mammals (eg Todd *et al.* 1997; Parsons *et al.* 1999).

Although such approaches are not available for most aquatic species, relatively benign *in situ*

biopsy has become a standard in marine mammal studies (eg Barrett-Lennard *et al.* 1996).

I emphasised in Chapter 3 the potential value of mark-recapture programs for directly

monitoring fishing mortality rates, *F*. However, for many species, it is difficult to meet the set

of stringent requirements that this would entail. There are innovative approaches that might

overcome some of theses problems: one attempt at developing *in situ* tagging protocols is a

"robot tagger" installed in a trawl net (Schrope 2000). However, such a device would not be

suitable for *Scomberomorus commerson* and other fish species that are not readily trawled

undamaged, or are behaviourally unsuitable for the approach. I suggest that a methodology

suitable for *S. commerson* and similar species would be a mark-recapture protocol entailing *in*

situ tissue collection ("tagging"), using a relatively benign technique, followed by sampling of

tissue from the landed catch (providing "recaptures"), and then comparison of individual

identities from the two sets of samples using molecular DNA techniques.

Post-marking mortality arising from conventional tagging processes may be substantial, is

difficult to quantify, and is often confounded with behavioural change and tag loss (eg Taylor *et*

al. 2001). Such mortality leads to underestimates of exploitation rate, since the estimator of this

rate is simply the number of tags recovered in the harvest divided by the number of fish tagged.

These problems can usually be overcome with sufficient funding and large samples sizes, but

this limits application of routine mark-recapture in small fisheries, or fisheries that are just not

very valuable. In contrast, a genetics-based approach is amenable to *in situ* methods of

collection that have negligible effects on a target animal (as very little tissue is needed for micro-satellite DNA sequencing). There is no tag loss problem, overcoming the requirements of Chapter 3 that tag shedding and tag loss are well-quantified. Total catch is reasonably well-known for most fisheries, so that the problem of estimating a reporting rate is replaced by a more tractable sampling problem. The final requirement, that tagging is representative over the fished population, remains as a project design problem.

For the protocol that I have described for genetic mark-recapture monitoring of F to be feasible, two major technical requirements must be met. The first is that genetic techniques are developed which not only identify individual fish of the target species to a sufficient level of precision, but are also inexpensive enough to allow the screening of a large number of individuals each year. The extensive use of molecular DNA techniques in fish population studies (eg for *Scomberomorus* spp., Gold *et al.* 1997; Gold *et al.* 2002; Broughton *et al.* 2002) and an ever-growing range of other applications means that there is a large body of applicable genetic information and a rapidly-evolving and ever-more accessible technology. For *S. commerson*, a study to develop economical identification and comparison of bulk samples using microsatellite DNA has already been initiated Overden (2003).

The second requirement for my genetic mark-recapture protocol is a methodology for collection of tissue *in situ*. I describe below the design concepts of a tool for the tissue collection, and a series of experiments that examine the relative effect of different design features on the success rate when the tool was deployed to collect tissue samples from *S. commerson*. The aims of these experiments were to develop a device that would predictably provide samples of tissue for at least 40% of the individual fish from which sampling is attempted, and to provide a series of basic observations upon which a more extensive program of technical development could be built.

5.2. Methods

5.2.1 Basic device

Attached to a lure body, the tissue sample collection tool is formed from a piece of copper tubing bent into the shape of a hook (Figure 5.1a,b). A sharpened stainless steel tip is affixed to the hooked end. The tip (Figure 5.1c) is provided with a pressure-relief hole (Figure 5.1d). One or two dental broaches (small sharp files) are positioned within the tip (Figure 5.1e,f). The other end is bent and soldered to form an eye (Figure 5.1g), for attachment to the lure body with a screw.
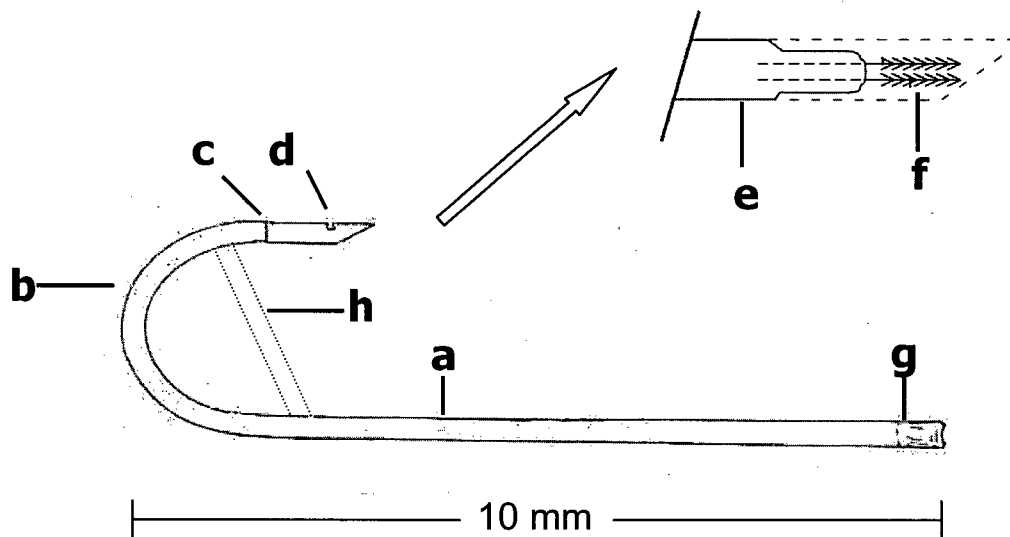
Figure 5.1. Basic configuration of device for *in situ* tissue collection (Design 1). The device consists of annealed copper tubing (**a**) bent to form a hook (**b**), with a stainless steel tubing tip (**c**) pressed or screwed to the hooked end; a small hole (**d**) is a pressure relief hole. The arrowed detail of hook end with tip removed (**e**) indicates the way that the dental broaches (**f**) are fixed into the tube body. The end of the device distal to the hook is formed into an eye (**g**) for attachment to a line or lure body. Position of brace is indicated by dotted lines across gape of hook (**h**).

When mounted on a lure, the tool works in much the same manner as the hook typically employed to catch a fish: when a fish attacks the lure, the force of the attack and the forward

force from the line on the lure bring the hook into contact with the side of the fish's jaw. The sharpened steel tip penetrates or scrapes the skin sufficiently to drive a tissue sample into the bore of the tip portion. After initial contact with the fish, continued force from the line and actions of the fish bend the flexible copper tubing backward so, as there are no external barbs, the device is disengaged and the fish freed. The broaches within the tip retain a small sample of tissue. The deformation of the hook also inactivates the device, preventing sampling from a second fish.

The basic test tool was manufactured from #11 gauge annealed copper tubing (inner diameter 2.24 mm, outer diameter 3.54 mm), usually sold as refrigeration tubing. In Experiment 3 below, some tools (Design 6) used annealed copper wire (diameter 2.5 mm). The other end was bent to a hook (J) shape. The tip was formed from a 15 mm length of #12 gauge stainless steel tubing (biopsy needle blades, 12 g X 101 mm, Portland Surgical Products, Australia), sharpened at an angle of around 60°. Either 1 or 2 coarse dental broaches (Antaeos Nervadeln, usually size 470; fine broaches were size 450) were inserted into the end of the hooked tube, and the tip crimped on (most designs; Figure 5.1e,f). In Experiment 1 (below) some devices were braced by soldering a piece of the copper tubing across the hook gape (Figure 5.1h). For devices made with a wire body, tips were tapped and screwed onto the threaded shaft of the hook. The broaches were firstly mounted in a piece of plastic tubing, then with epoxy cement and held in place in the screwed-on tip (see details in Figure 5.2).
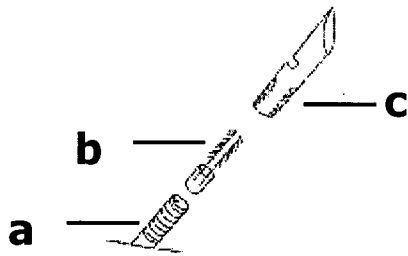
Figure 5.2. Detail of screw method of attaching tips and broaches for wire-shafted (Design 6) hooks. The end of the wire hook body is tapped (**a**). The dental broaches are firstly mounted in a piece of plastic tubing (**b**) then held tightly in place by the threaded tip (**c**) and epoxy cement.

The single hook design (Design 1) above was varied to employ multiple hooks, with the underlying assumption that a fish striking a lure with several hooks is more likely to come in contact with a tip or tips, increasing the chance of a successful harvest of tissue.

Lure bodies used in the experiments described here were # 6 (6 inch) Smith's jigs, which are frequently used in the commercial fishery for *Scomberomorus commerson*. They are tapered, lens-shaped PVC, pink, white or red in colour, 159 mm long, 25 mm in diameter at the front and 14.6 mm at the rear.

5.2.2 Experiments

Preliminary trials indicated that the basic tool design concept was acceptable in that struck lures (as evidenced by bent hook shafts) frequently yielded tissue samples, and provided a set of observations on which hypotheses about design factors could be developed.

For the sake of the experiments described here, I define "success" as an instance where a struck lure retained an observable tissue sample. The hook gape diameter, hook pliability and whether either one or two broaches were incorporated in the tip were all identified as potential design factors bearing upon success rates, and were examined in Experiment 1. I hypothesised that

success rate would be improved by employing a multiple hook design. Thus in Experiment 2, I compared the relative performance of several multiple-hook designs and the best-performing single hook design from Experiment 1. Experiment 3 provided an additional test of hook shaft pliability as a factor in success rates, as well as a further comparison of single hook and multiple hook performance. The use of three line positions during Experiment 3 also provided opportunity to demonstrate whether deployment factors had a significant effect on success rate.

Trials were conducted during daylight fishing aboard the licensed Spanish mackerel troller *Taroona II*, during otherwise routine operations on established fishing grounds near Darwin, in Australia's Northern Territory (approximate locations 11° S, 130° E). In Experiments 1 and 2, only one of the vessel's lines was available for charter during commercial fishing operations. Consequently, Experiments 1 and 2 below describe sequential deployments of the different configurations of device designs. Experiment 1 used Line 4 of the vessel, and Experiment 2 used Line 2 (Figure 5.3). Three lines (Lines 1, 2 and 3; Figure 5.3) were available for charter during Experiment 3, providing for between-line comparisons. Note that lines 3 and 4 were rigged identically (Figure 5.4). For each configuration, the gear was deployed until apparently struck, shown by jerking of the line, or the unravelling of a slipknot put in the line as a telltale. Unstruck lines were retrieved and inspected at intervals of around 10 minutes. If, when the lure was retrieved, the hook shafts were not bent, the gear was immediately redeployed without changing the lure.
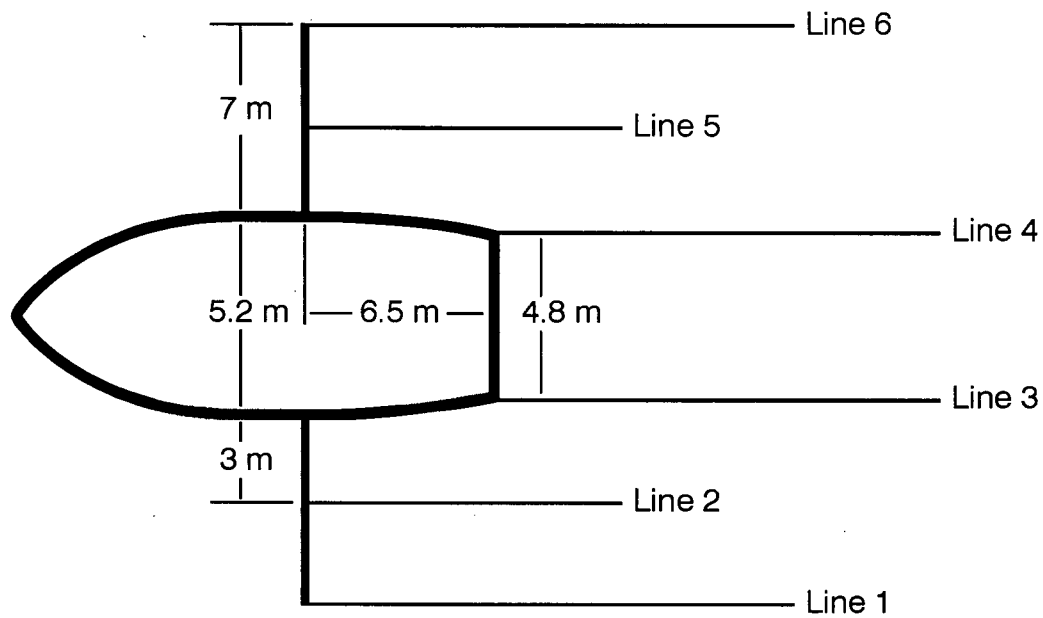
Figure 5.3. Arrangement of lines aboard *F. V. Tarroona II* during experiments to test designs of a device for remote tissue collection.
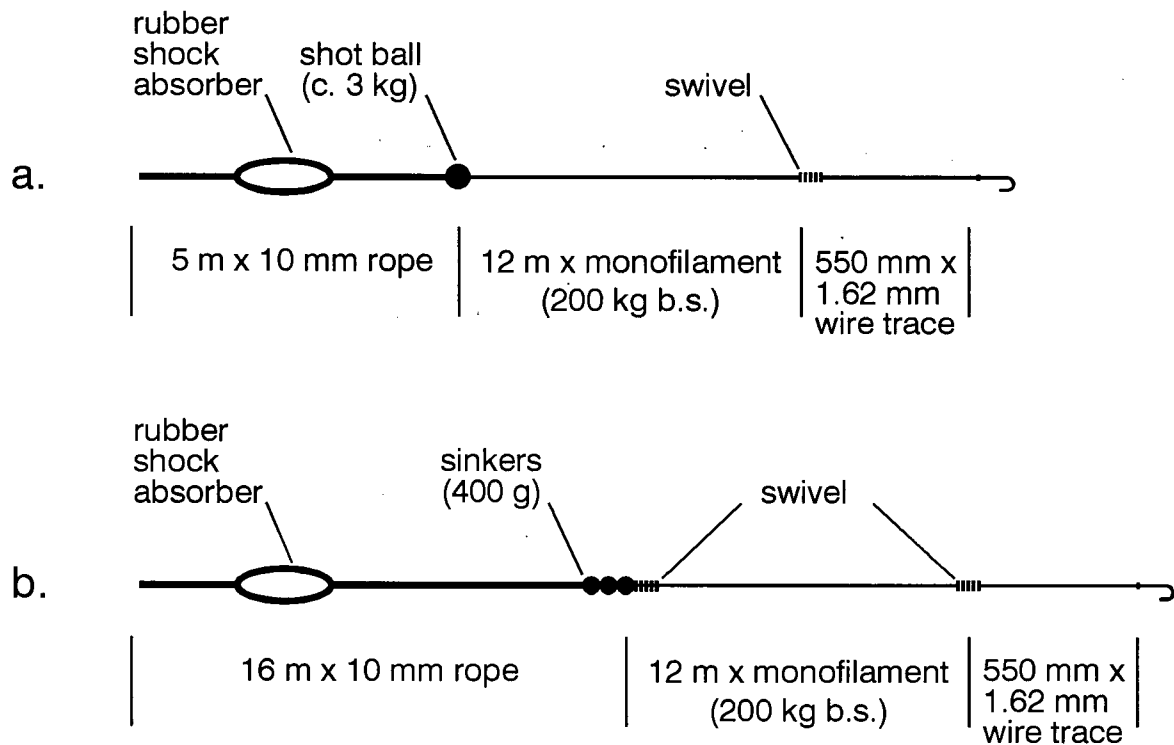
Figure 5.4. Line configurations during experiments to test designs of a device for remote tissue collection, with line numbers as identified on Figure 5.3. **a,** Lines 2 and 5; **b,** Lines 1, 3, 4 and 6.

Experimental design

Experiment 1

In this first experiment, I attempted to test basic design attributes that I hypothesised had bearing on success. All lures used in Experiment 1 were fit with a Design 1 form of the device (Figure 5.1). With the factor GAPE, the success rates of three hook gapes were compared, Small (inside diameter, mean 23.0 mm, SD 1.94 mm), Medium (39.0 mm, 5.09 mm) and Large (53.0 mm, 6.94 mm). Hooks were either braced or not (factor BRACE), in an attempt to alter hook pliability, and had either one or two broaches mounted in the tip (factor BROACH), based on the prediction that, for retaining harvested tissue, two broaches should be more effective than one. Difficulty with sourcing dental broaches also meant that it was necessary to use size 470 broaches for five lures (Broach Type = "Fine" in Appendix Table 5.1).

The different Gape levels were arranged in randomised blocks, each block containing one lure of each gape size (nested within Latin squares of the BRACE factor levels, and nested again within a Latin Square of the BROACH factor levels (Table 5.1). Labelling errors meant that the design was not accurately followed, with double broaches replacing single on

Table 5.1 Design of Experiment 1. Each Block included one lure of each Gape size. *A double-broached lure was inadvertently included in each of these blocks. Latin squares in Bracing factor are outlined.

Block	Bracing	Broach	Block	Bracing	Broach
1	B	D	9	B	S
2	N	D	10	N	S*
3	N	D	11	N	S*
4	B	D	12	B	S
5	B	S	13	N	D
6	N	S*	14	B	D
7	N	S*	15	B	D
8	B	S*	16	N	D
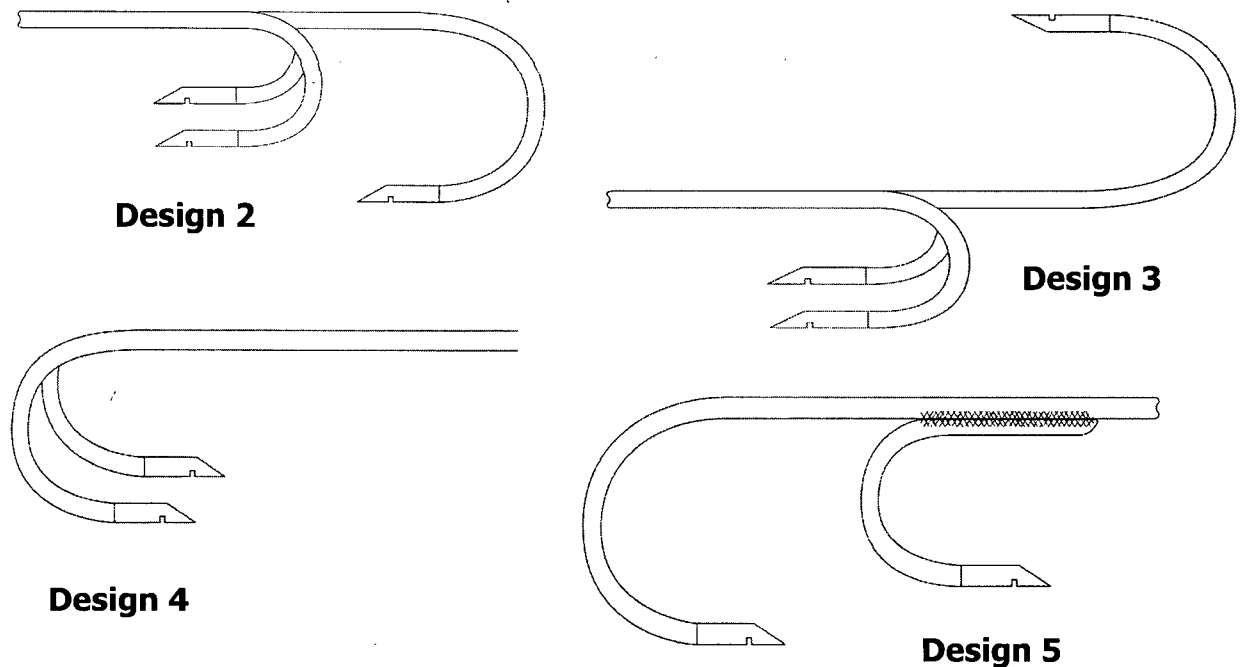
Figure 5.5. Designs of multi-hook devices. See text for details.

five lures (Lure Numbers 17, 20, 22, 30, 31 in Appendix Table 5.1). Experiment 1 could not be completed during a single trip, so that BROACH-TYPE and TRIP (1 or 2) were also included as factors in analyses.

Experiment 2

The second experiment compared the relative performance of the best-performing single hook design from Experiment 1 (Design 1 with small gape), plus four multi-hook designs (Figure 5.5). Design 2 consisted of a large gape hook (as for Experiment 1) with two small gape hooks each arranged radially at about 45° from the vertical, soldered in place forward of the central hook; the tips of the hooks were all oriented in the same direction (Figure 5.5). Design 3 was very similar to Design 2, with the variation that the large hook was rotated about the central axis of the hook shaft by 180° in relation to its position in Design 2 (Figure 5.5). Design 4 comprised two small gape hooks joined so that they each lay at an angle of around 45° from the

vertical plane (Figure 5.5). In Design 5, a small gape hook was soldered onto a large gape hook so that they lay in line in the same plane (Figure 5.5).

Lures were deployed sequentially in blocks that contained one of each of the lure types, with order randomised within blocks. All designs were unbraced and with two broaches included in the tips. These trials were conducted as in Experiment 1, with lures deployed sequentially on a single line. This experiment had to be continued across three trips. The analysis for Experiment 2 thus examined the effect of the factors DESIGN and TRIP upon success rate.

Experiment 3

In the third experiment, I compared the performance of the most successful designs from Experiment 1 and Experiment 2 (Designs 1 and 4 respectively). In a further attempt to explore the effect of the pliability of the hook on success rates, the simple double-hook design used in Experiment 2 (Design 4) was made from wire rather than tube, so that the hook shafts were more pliable, producing Design 6. As three lines were used simultaneously throughout this experiment, the effect of the factor LINE on success rate was also evaluated.

Statistical analyses

Each experiment was analysed using a generalized linear model (GLM) with a binomial error structure and a logit link function, in the S-Plus™ environment. The procedure followed was to fit models with all possibly important factors, then to reduce model complexity by eliminating the factors with only marginal effect. The Akaike information criterion,

AIC = - 2 maximized loglikelihood + 2 # parameters,

was used to examine the effect on the model fit of each factor by deleting each factor in turn from the initial model (Venables and Ripley 1999). The chosen model was that which minimized the AIC.

The first experiment was initially analysed using the factor terms GAPE, BRACE, BROACH, TRIP, BROACH-TYPE, BAG and BRACELS (Brace Latin Square).

Experiment 2 was analysed using the factor terms DESIGN and TRIP, while Experiment 3, the basic factors examined were DESIGN, LINE and BAG.

5.3. Results

Experiment 1

Complete results for Experiment 1 are tabulated in Appendix Table 5.1 and summarized in Table 5.2. The first experiment produced an overall success rate of 20.8%, out of 96 fish strikes. Design attributes strongly influenced the success rate: the number of broaches was particularly important, with all 10 successes coming from the 29 strikes (34.5%) with double broaches. This was in complete contrast to the zero successes of the one-broach configuration. There were also strong differences between the gape sizes with the small gapes producing an overall success rate of 43.8 % (or 58.3 % of double-broach strikes), compared to only 12.5 % (22.2% of double broach strikes) and 6.3 %, (12.5 %) respectively.

Table 5.2. Summary results for Experiment 1.

Gape		Broach				Grand Totals	
		Double	% Success	Single	% Success		% Success
L	Successes	1	12.5	0	0.0	1	6.3
	Strikes	8		8		16	
M	Successes	2	22.2	0	0.0	2	12.5
	Strikes	9		7		16	
S	Successes	7	58.3	0	0.0	7	43.8
	Strikes	12		4		16	
	Total Successes	10	34.5	0	0.0	10	20.8
	Total Strikes	29		19		48	

The model could not converge when the factors BLOCK and BRACELS were included, as there were too many extreme values (0s). These factors were therefore deleted before analysing an initial model with the remaining factors GAPE, BRACE, BROACH, TRIP and BROACH-TYPE.

The factors TRIP, BROACH-TYPE and BRACING were progressively eliminated from the model, reducing the AIC from 44.50 to 39.87 (Table 5.3), with GAPE and BROACH being only the factors included in the final model. Coefficients for the remaining factors in the model are presented as Table 5.. A null deviance of 49.13 on 47 degrees of freedom indicated that the model was not over-dispersed. The analysis of deviance indicates the high impact of these factors on the success rates of the designs used in Experiment 1, with the factor BROACH being particularly important ($p < 0.002$; Table 5.).

Table 5.3. Elimination of uninformative factors from the GLM for Experiments 1, 2 and 3 using the AIC. NB for Experiment 3, no factors were removed from the initial model. See text for detailed explanation.

Expt.	Step	Factor removed	df increment	Deviance increase	Residual df	Residual deviance	AIC
1	1	(Initial model)			41	30.52030	44.52030
	2	TRIP	1	0.0947300	42	30.61503	42.61503
	3	BROACH -TYPE	1	0.3342426	43	30.94927	40.94927
	4	BRACING	1	0.9159642	44	31.86523	39.86523
2	1	(Initial model)			34	43.52884	75.52884
	2	BAG	9	14.92753	43	58.45637	72.45637
	3	TRIP	2	1.10019	45	59.55656	69.55656
3		(Initial model)			47	55.22583	73.22583

Table 5.4. Coefficients and analysis of deviance for the informative factors in Experiment 1.
* The intercept incorporates Large GAPE and Single BROACH

GLM coefficients				Analysis of Deviance				
	Value	Std. Error	t value	df	Deviance	Residual df	Residual Deviance	Pr(Chi)
Intercept*	-11.27	21.74	-0.518			47	49.13	
GAPE				2	7.7	45	41.47	0.0217
M	0.70	1.34	0.519					
S	2.28	1.22	1.873					
BROACH	9.32	21.72	0.429	1	9.6	44	31.87	0.0019

Table 5.5. Predicted success rates for combinations of hook GAPE and Broach configurations.* Confidence intervals for combinations include single broaches, for which all experimental results were zero, were not defined.

Gape	Broach	Estimate	95% c.i., lower	95% c.i., upper
Large	Single	1.28E-05	*	*
Large	Double	1.25E-01	1.63E-02	5.52E-01
Medium	Single	2.55E-05	*	*
Medium	Double	2.22E-01	5.37E-02	5.90E-01
Small	Single	1.25E-04	*	*
Small	Double	5.83E-01	3.01E-01	8.20E-01

Predicted success rates calculated from the coefficients (Table 5.5) ranged from 58.3% for the combination of a double broach and a small hook gape, down through 22.2% and 12.5% for double broached hooks with medium and large gapes, respectively. Predicted success rates for single-broached hooks were close to zero for all gape sizes.

Experiment 2

Results for Experiment 2 are tabulated in Appendix Table 5 and in summarized in Table 5.6. Although there was an overall success rate of just 44.0%, there was an appreciable difference among the relative success rates of the single hook and the various multi-hook combinations. The single hook combination (Design 1) achieved only a 20% success rate (2 successes of 10 strikes), as did Design 5. The other 3 designs all achieved success rates of 50% or more, up to 70% for Design 4 (Table 5.6). Although the overall success rates improved through the series of trips, this was marginal given the number of strikes in the later 2 trips (Table 5.6), and factors BLOCK and TRIP were both eliminated from the model, reducing the AIC from 79.53 to 55.54. Only DESIGN remained in the final model.

Table 5.6. Summary of success rates for Experiment 2.

Trip	Design	1	%	2	%	3	%	4	%	5	%	Total	%
1	Successes	1	16.7	3	42.9	3	50	4	57.1	2	33.3	13	40.6
	Strikes	6		7		6		7		6		32	
2	Successes	0	0	1	50	2	100	2	100	0	0	5	45.4
	Strikes	2		2		2		2		3		11	
3	Successes	1	50	1	100	1	50	1	100	0	0	4	57.1
	Strikes	2		1		2		1		1		7	
	Total Successes	2	20	5	50	6	60	7	70	2	20	22	44.0
	Total Strikes	10		10		10		10		10		50	
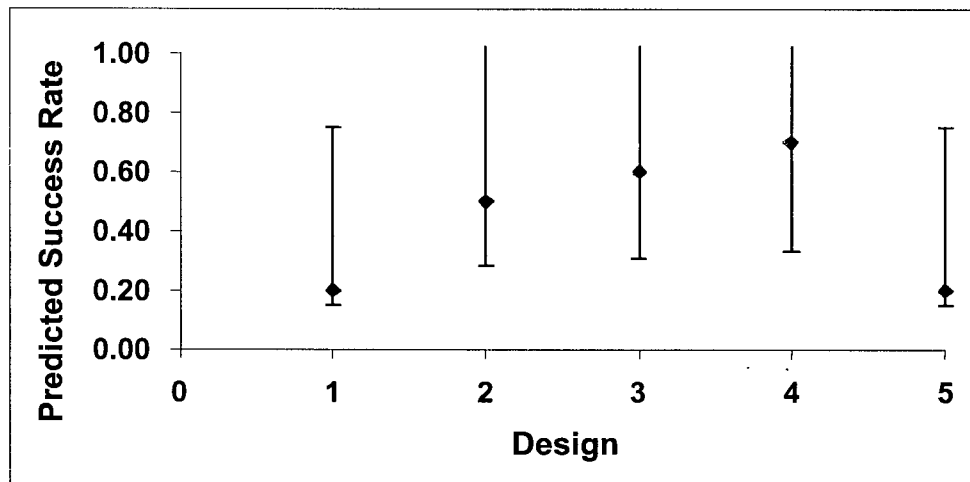
Figure 5.6. Predicted success rates for the different lure designs of Experiment 2. Error bars are 95% confidence limits.

Coefficients for the factor (DESIGN) remaining in the model for Experiment 2 are presented in Table 5.7. A null deviance of 68.59 on 49 degrees of freedom indicated that the model was not over-dispersed. The analysis of deviance indicates sufficient variation between design types, ($p \cong 0.06$; Table 5.7) that further investigation is warranted.

Predicted success rates calculated from the coefficients (Figure 5.6) ranged from 20.0%, for Designs 1 and 5 (the single hook, Figure 5.1 and the double in-line hook of Figure 5.5), to 70 %, for the simple double hook of Design 4 (Figure 5.5). However, the wide 95% confidence limits for the predictions (Figure 5.6) indicate the lack of a strong result for Experiment 2.

Table 5.7. Coefficients and analysis of deviance for the informative factors in Experiment 2.
* The intercept incorporates Design 1.

GLM coefficients	Value	Std. Error	t value	Analysis of Deviance df	Deviance	Residual df	Residual Deviance	Pr(Chi)
Intercept*	-1.39	0.79	-1.76			49.00	68.59	
Design				4	9.04	45.00	59.56	0.060
2	1.39	1.01	1.37					
3	1.79	1.02	1.76					
4	2.23	1.05	2.13					
5	9.30E-17	1.11	8.34E-17					

Experiment 3

In this experiment I compared the success rates of the best-performing designs from Experiment 2 (Design 4) and Experiment 1 (Design 1; note that this was also the worst-performing design of Experiment 2), as well as a two-hooked design manufactured with a more pliable material (Design 6). Results for Experiment 3 are tabulated in Appendix Table 5.3 and are summarized in Table 5.8. The overall success rate, 42.9 %, was similar to that for Experiment 2. In this experiment there was again an apparent difference in success rate between hook designs. Both Designs 1 and 6 had overall success rates of 33.3% (6 of 18 strikes successful), while Design 4 achieved a 60.0% success rate (12 successes of 20 strikes; Table 5.8). There was also a notable difference between success rates across lines, with them varying between 26.3% for Line 1, through 31.3 % (Line 2) to 66.7% for line 3 (Table 5.8). There was apparently interaction between designs and lines, with the double-hooked designs being much more successful on Line 3 than on the other lines, but only Design 4 showing a particular improvement in success rate on Line 2 (relative to Line 1; Figure 5.7).

Table 5.8. Summary of success rates for Experiment 3.

Line		Design 1	%	4	%	6	%	Total	%
1	*Successes*	2	40.0	2	28.6	1	14.3	5	26.3
	Strikes	5		7		7		19	
2	*Successes*	1	20.0	4	66.7	0	0.0	5	31.3
	Strikes	5		6		5		16	
3	*Successes*	3	37.5	6	85.7	5	83.3	14	66.7
	Strikes	8		7		6		21	
	Total Successes 6		33.3	12	60.0	6	33.3	24	42.9
	Total Strikes 18			20		18		56	
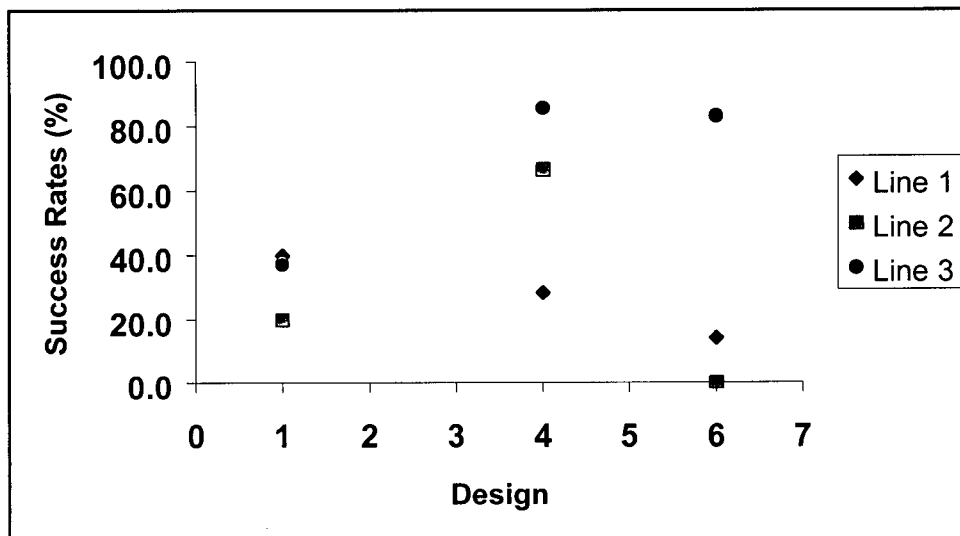
Figure 5.7. Success rates for Experiment 3 by Design and by Line.

There was again trouble including the factor BAG in the GLM, due to the frequent zero values, so the model was fit with the factors DESIGN, LINE and a (DESIGNxLINE) interaction term. Removal of factors did not reduce the AIC, hence all factors were appropriate for inclusion in the model. A null deviance of 76.49 on 55 degrees of freedom indicated that the model was not over-dispersed. Analysis of deviance for the GLM (Table 5.9) indicates the strong effect of LINE upon success rate, and that DESIGN and its interaction with the LINE factor were similarly important. However, several interaction terms were not defined. For this reason, a model including for onlythe main factors, DESIGN and LINE, was used to calculate the factor coefficients presented in Table 5.9 and the predictied success rates of Table 5.10.

Predicted success rates calculated from the coefficients (Table 5.10) ranged from lows of 0.14 and 0.18 for Design 1 on lines 1 and 2, up to 0.85 and 0.61, for Designs 4 and 6 respectively, on Line 3. Design 4 would be expected to exceed the desired level of 40% success rate on all lines, although confidence intervals were wide for lines 1 and 2. Lures of each of the three designs of Experiment 3 could be expected to perform well on Line 3, with predicted success rates of 0.55 or more (Table 5.10).

Table 5.9. Coefficients for main factors and analysis of deviance for the informative factors in Experiment 3. Note that factor coefficients were calculated from the model including DESIGN and LINE but not their interaction. *The intercept incorporates Design 1 and Line 1.

Factor	Coefficient	Std. Error	t value	Analysis of Deviance df	Deviance	Residual df	Residual Deviance	Pr (Chi)
Intercept	-1.79	0.78	-2.29			55	76.49	
Line				2	7.98	53	68.51	0.019
2	0.27	0.78	0.34					
3	1.99	0.79	2.62					
Design				2	5.04	51	63.47	0.081
4	1.53	0.77	1.98					
6	0.25	0.78	0.33					
Line X Design				4	8.24	47	55.23	0.083

Table 5.10. Predicted success rate for combinations of Design and Line from Experiment 3.

Design	Line	Estimate	95% c.i., lower	95% c.i., upper
1	1	0.14	0.03	0.44
1	2	0.18	0.04	0.51
1	3	0.55	0.26	0.81
4	1	0.44	0.18	0.73
4	2	0.50	0.21	0.79
4	3	0.85	0.58	0.96
6	1	0.18	0.05	0.48
6	2	0.22	0.06	0.56
6	3	0.61	0.29	0.86

5.4. Discussion

The results of this study clearly indicate the feasibility of remotely collecting small tissue biopsies; the simple double hook variation, Design 4, is predicted to exceed the criterion of a 40% success rate, at least for the different line positions used in this study. However, both design features and deployment conditions had strong influence on the success rates achieved, so that further development of the design concept should address not only design features of the sampling device, but also deployment protocols that maximize success rates.

While these small experiments should mark only the beginning of experimentation with these devices, it was clear from the first experiment that incorporation of two dental broaches, rather than one, into the device tips markedly affected the success rates. Consequently, the subsequent experiments reported here used double-broached tips in all designs. The mechanism by which the second broach increased performance was not investigated, but it may be related to the space occupied by the broaches relative to the bore of the tip. The extra broach, in addition to providing extra grip, would act to reduce the distance between the broach, the other broach, and the bore of the tip. Designs using a smaller tip with a single broach might also be feasible, and should be tested as size reduction of the device might extend its utility for smaller individuals or species. In addition, the number of broaches (@Aust. $0.60) contributes significantly to the material cost of the tool.

Hook gape diameter also had a large effect on catch rates. I suggest that this is a simple mechanical effect: with the larger gape devices, the probable action was that the side of the fish's head came within the gape of the device, straightening out the hook – so that the lure was struck and the device de-activated rather than the tip coming into contact with the side of the fish, where a sample could be taken. This interpretation implies that the success rate would also

then be dependent upon the size of the fish, and in future could be tested by comparing the size composition of recaptured fish with the size composition of concurrent landings. Success rates of smaller gapes than those tested here should also be examined. Future work could be aimed at tailoring gape size to fish size composition of target groups in response, for example, to observed size of fish in landings (to target the section of the population vulnerable to fishing), or to large spawning females (say in studies of movements related to spawning). Bracing had little effect on the success rates and, given the results of Experiment 3, may not have affected pliability at all.

The hypothesis that appropriate multi-hook designs would provide increased success rates was tested in Experiment 2. Most of the multi-hook designs did increase the success rate of the device, relative to the single-hooked design. There was a range of success rates among the multi-hook designs, with the relatively simple double hook design being the most successful. Substantially more experimentation would be required to closely define the relative performance among the different designs. Experiment 2 simply indicated that some multi-hook designs are likely to be more successful than the single-hook design, and it was beyond the scope of this study to establish the characteristics that cause the performance differences. However the complexity of a design and the number of broaches included are important factors in the difficulty of manufacture and cost of different applications to which this gear might be directed. It is also important to consider that the design may affect the appearance to a fish, or swimming action of a lure, affecting strike rates or the vigour of attack, so that variations need to be tested against standard designs that mimic conventional lures. For the purpose of developing a device that could predictably achieve a success rate of 40% or more, the simple double-hook configuration (Design 4) was chosen on the basis not only of performance (it had the highest success rate in Experiment 2), but as the easiest to make among the multi-hook designs tested.

The success rate of the most successful design configuration from Experiment 1 declined from 58% in the first experiment to 20% in the second. Extrinsic factors such as size of the fish, the effect of minor variations in trolling speed, or unobserved factors that led to variation in the vigour of attack, may have caused or contributed to the difference. However, the only procedural variation that may have caused this difference is that the lines on which the gear was trolled differed between the two experiments.

Substantial support for this last interpretation was provided by the results of the third experiment. Not only were there strong differences in performance between lines, the interaction between lines was just as strong a factor as device design. The two-hooked tool manufactured from tube outperformed the single-hooked version in Experiment 3, as predicted by the second experiment. It also outperformed the more-pliable two-hooked tool manufactured from wire. However, for Line 3, the success rates for both versions of the two-hooked tool were similar. One interpretation of this result that is amenable to future experimentation is that success rate depends on the elasticity of the line/ device system. This could be examined by comparing success rates of the lines with different shock absorber configurations. Additional (not mutually exclusive) hypotheses relate the success of the tool to the vigour of attack by fish, and this in turn to the position of a line near the propeller wash, or the boat, or to the fact that, as a boat turns, the line on the outside of the arc will move more quickly and sweep a greater area than lines inside the arc. Circling after a strike is a common tactic in this fishery, with the purpose of keeping in proximity to the school from which the striking fish came. Some fishers habitually turn in one direction. However we have observed that the skipper of the *Taroona II* has no strong habit in this respect.

In planning, I assumed that strikes rates would probably differ between lines, but that the success rate should not. Thus experimental procedures reflected the expectation that variation in success rates was likely to be coincident with time- or space-related size or behavioural effects (eg spatial correlation in fish size, changed behaviour with water clarity). However this had a minor effect on the statistical analysis or the interpretation of results.

This series of experiments was conducted with the intention of demonstrating the feasibility of sampling fish tissue remotely, to develop a basic device that could be used for further experimentation, and to initiate experiments on genetic tagging. The constraints of conducting research during commercial fishing activities meant that factors such as the choice of line used depended upon convenience for the crew, especially given no *a priori* appreciation that choice of line might be a dominant factor in these studies.

The success rates of best-performing designs here are probably already adequate to undertake mark–recapture experiments in *S. commerson* fisheries. However the high variability of success rates even within designs indicates that further research into design features and into deployment methods would be fruitful.

Whether the designs are more widely applicable to other species will depend largely on their attack behaviour. I have observed that *S. commerson* tend to strike the rear of a lure or bait, from the side. Designs could be used with little change for species that attack in a similar manner, such as other *Scomberomorus* species. Additionally, whether striking a biopsy device affects subsequent fish health and behaviour relative to fishing gear is a consideration that has important bearing on application of the gear in mark-recapture studies and should be investigated. It would, however, be reasonable to assume that any such effect would have to be less than caused by the capture of a fish with conventional fishing gear before tagging. It is also

conceivable that *in situ* sampling and genetic-based mark-recapture may simply be less expensive than conventional tagging approaches, when limiting costs relate to the number of samples that can be taken at sea.

The approach I have outlined here addresses three of the main problems of using mark-recapture to monitor fishing mortality rates: by remotely sampling tissue and using genetic identification, tag shedding and tag-induced mortality should be rendered negligible. The problem of knowing reporting rates should be reduced to a sampling problem, given that total catch is known. Finally, the simplicity of this approach means that incorporation of tactics such as industry participation that requires daily tagging of a set number of fish, by fishers operating over wide areas and long seasons, would help address the problem of ensuring all members of the fished population have a similar probability of being tagged.

5.5. Appendix 1

Appendix Table 5.1. Data for Experiment 1, testing of design attributes of remote tissue sampling tool. Lure number indicates the consecutive order in which lures were deployed.   Gapes were L= large, M = medium; S = small; Bracing: B = Braced, N = Not Braced, Broach: S = Single, D = Double; Success: 1 = visible tissue, 0 = no visible tissue.

Trip	Lure number	Block	Gape	Brace Latin Square	Bracing	Broach	BroachType	Success
1	1	1	M	1	B	D	Fine	1
1	2	1	L	1	B	D	Coarse	0
1	3	1	S	1	B	D	Coarse	1
1	4	2	S	1	N	D	Coarse	0
1	5	2	L	1	N	D	Coarse	0
1	6	2	M	1	N	D	Fine	0
1	7	3	M	1	N	D	Fine	0
1	8	3	L	1	N	D	Coarse	0
1	9	3	S	1	N	D	Coarse	1
1	10	4	S	1	B	D	Coarse	1
1	11	4	M	1	B	D	Fine	0
1	12	4	L	1	B	D	Coarse	0
1	13	5	M	2	B	S	Coarse	0
1	14	5	L	2	B	S	Coarse	0
1	15	5	S	2	B	S	Coarse	0
1	16	6	M	2	N	S	Coarse	0
1	17	6	S	2	N	D	Coarse	0
1	18	6	L	2	N	S	Coarse	0
1	19	7	M	2	N	S	Coarse	0
1	20	7	S	2	N	D	Coarse	0
1	21	7	L	2	N	S	Coarse	0
1	22	8	M	2	B	D	Coarse	0
1	23	8	S	2	B	S	Coarse	0
1	24	8	L	2	B	S	Coarse	0
1	25	9	L	3	B	S	Coarse	0
1	26	9	M	3	B	S	Coarse	0
1	27	9	S	3	B	S	Coarse	0
1	28	10	L	3	N	S	Coarse	0
1	29	10	M	3	N	S	Coarse	0
1	30	10	S	3	N	D	Coarse	1
1	31	11	S	3	N	D	Coarse	1
2	32	11	M	3	N	S	Coarse	0
2	33	11	L	3	N	S	Coarse	0
2	34	12	S	3	B	S	Coarse	0
2	35	12	M	3	B	S	Coarse	0
2	36	12	L	3	B	S	Coarse	0
2	37	13	M	4	N	D	Coarse	0
2	38	13	S	4	N	D	Coarse	0
2	39	13	L	4	N	D	Coarse	0
2	40	14	L	4	B	D	Coarse	0
2	41	14	S	4	B	D	Coarse	0
2	42	14	M	4	B	D	Coarse	0
2	43	15	L	4	B	D	Coarse	1
2	44	15	S	4	B	D	Fine	1
2	45	15	M	4	B	D	Coarse	0
2	46	16	L	4	N	D	Coarse	0
2	47	16	M	4	N	D	Coarse	1
2	48	16	S	4	N	D	Coarse	1

Appendix Table 5.2. Data for Experiment 2, comparison of relative performance of different single-hook and multi-hook lure designs.   Success: 1 = visible tissue, 0 = no visible tissue.

Trip	Lure number	Block	Design	Success	Trip	Lure number	Block	Design	Success
1	1	1	1	0	1	26	6	4	0
1	2	1	2	0	1	27	6	2	0
1	3	1	3	1	1	28	6	5	0
1	4	1	4	0	1	29	6	1	0
1	5	1	5	0	1	30	6	3	0
1	6	2	4	1	1	31	7	2	0
1	7	2	5	0	1	32	7	4	1
1	8	2	2	1	2	33	7	1	0
1	9	2	3	1	2	34	7	5	0
1	10	2	1	0	2	35	7	3	1
1	11	3	4	1	2	36	8	1	0
1	12	3	2	1	2	37	8	3	1
1	13	3	5	1	2	38	8	2	0
1	14	3	1	0	2	39	8	5	0
1	15	3	3	0	2	40	8	4	1
1	16	4	3	1	2	41	9	2	1
1	17	4	1	0	2	42	9	5	0
1	18	4	4	0	2	43	9	4	1
1	19	4	2	0	3	44	9	1	1
1	20	4	5	1	3	45	9	3	1
1	21	5	2	1	3	46	10	1	0
1	22	5	1	1	3	47	10	2	1
1	23	5	5	0	3	48	10	5	0
1	24	5	4	1	3	49	10	3	0
1	25	5	3	0	3	50	10	4	1

Appendix Table 5.3. Data for Experiment 3, comparison of relative performance of basic single and double hook lure designs, and shaft manufacture material.   Success: 1 = visible tissue, 0 = no visible tissue.

Line	Lure number	Block	Design	Success	Line	Lure number	Block	Design	Success
1	1	1	6	0	2	29	12	1	0
2	2	1	1	0	3	30	12	6	1
3	3	1	4	1	3	31	12	4	1
2	4	2	4	1	3	32	13	6	1
2	5	2	1	0	2	33	13	4	1
1	6	3	4	1	1	34	13	1	0
1	7	3	6	0	1	35	14	4	0
3	8	3	1	0	2	36	14	6	0
2	9	4	4	1	3	37	14	1	0
2	10	4	6	0	1	38	15	4	0
3	11	4	1	1	2	39	15	1	0
1	12	5	4	0	1	40	15	6	0
1	13	5	1	0	2	41	16	6	0
3	14	6	4	1	3	42	16	4	1
3	15	6	6	1	2	43	17	4	1
1	16	6	1	1	3	44	17	1	0
3	17	8	1	0	1	45	17	6	1
3	18	8	4	0	3	46	18	1	1
3	19	8	6	0	2	47	18	4	0
3	20	9	1	1	1	48	18	6	0
1	21	9	4	1	1	49	19	6	0
2	22	9	6	0	3	50	19	1	0
3	23	10	4	1	2	51	19	4	0
1	24	10	1	0	2	52	20	1	1
2	25	10	6	0	3	53	20	4	1
1	26	11	6	0	3	54	20	6	1
1	27	11	4	0	1	55	21	4	0
1	28	11	1	1	3	56	21	6	1

6. Should genetic tagging be used to monitor the Northern Territory Spanish mackerel fishery?[5]

6.1. Introduction

Managers of fisheries must make decisions about how a fishery is to be monitored. These decisions will invariably be a compromise between the quality of information provided by that monitoring - the predicted impact that it will have on management performance - and cost. Thus monitoring measures and their attendant costs can be compared on the basis of expected performance, and a manager can make an evaluation against available funding. In previous chapters I have indicated the acute need in the NT Spanish mackerel fishery, and many other fisheries, for a monitoring method that provides direct information on fishing mortality rates. I have suggested the implementation of genetic mark-recapture as a monitoring method, to supply that information.

The technical feasibility of genetic tagging has now been established. In Chapter 5, I described a device for *in situ* sampling, that should overcome the tag loss problems (shedding and mortality) that impede the use of conventional tags for monitoring fishing mortality rates. I indicated that the requisite molecular genetics techniques also exist and have been applied in mark-recapture experiments for a range of species, but have not been used as yet for monitoring fishing mortality rates in fish species.

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[5] A successful application to Australia's Fisheries Research & Development Corporation, FRDC 2002/011, *GENETAG: Genetic mark-recapture for real-time harvest rate monitoring. Pilot studies in northern Australia Spanish mackerel fisheries*, was based on preliminary work for this section of the thesis project. Preliminary analyses of the combined use of genetic and conventional tags were presented by Buckworth and Martell (2003).

Whether or not genetic tagging is adopted as a monitoring method for any fishery will depend upon decisions that weigh the benefits of using the methodology against its costs. My purpose in this chapter is to provide the basis for such a decision for the NT Spanish mackerel fishery, and similar fisheries. A limit reference point of 450 tonnes total catch is in place for this fishery. As indicated by the previous chapter, the status of this fishery is poorly known; while there is some danger that the reference point is too high, it may also be an underestimate of the real potential of the fishery and may hence be unnecessarily preventing its economic growth. .

Mark-recapture has not been used to monitor this fishery previously because of the high expense of capturing *Scomberomorus commerson*, Spanish mackerel. The fish are large and difficult to handle, so that even in commercial operations they are caught at rates of tens and kg of fish per day, not thousands and tonnes. Tag mortality has been considered a significant problem: a highly active predator, a mackerel is often damaged when striking at baited hooks or lures, attracting subsequent attack from sharks or other mackerels. The loss of hooked mackerel to sharks is a constant problem for troll fishermen and anglers. Additionally, the fighting behaviour so respected by sports fishers probably creates a substantial physiological stress response that, even if sublethal in itself, probably compromises subsequent survival (Gustaveson *et al.* 1991; Taylor *et al.* 2001) Many of the fish tagged would be lost as a result of the capture and tagging, but just how many would be difficult to ascertain. Mark recapture with *S. commerson* has produced poor recapture rates of around 2% (McPherson 1992; Mackie *et al.* 2003).

The genetic tagging approach is envisioned as a means by which the marking of several hundred fish per year can be achieved, without the tag loss problems, by using *in situ* marking by the commercial troll fishermen, as suggested in Chapter 5. Screening can be accomplished by

paying for the collection of fins. There is also potential for conventional tagging programs to contribute to the estimation of fishing mortality rates and catchability when they are conducted in concert with a genetic tagging protocol (Buckworth and Martell 2003). The NT has a rapidly-growing guided-fishing sector and catch-release culture, in which a significant number of Spanish mackerel are now caught and released annually (> 2000 fish in 2001: NT Fisheries logbook data). The opportunity thus exists for a well-coordinated, small conventional tagging program, that minimises the cost of capture.

I used the FISHMOD implementation described in Chapter 2 to conduct evaluations of the performance of fishery monitoring based on genetic tagging, with and without conventional tagging, relative to the performance of CPUE and mean weight monitoring. The output of the tagging program is used to track fishing mortality rates and catchability, the Tq approach successfully applied in Chapter 3. The simulated fishery is spatially similar to the NT Spanish mackerel fishery. The simulations are necessarily for a hypothetical fishery – the spatial dynamics of the NT Spanish mackerel population are not known, and the historical magnitude and spatial distribution of catch and effort are poorly known. I demonstrate that the genetic mark-recapture approach is expected to give superior performance to CPUE and mean weight monitoring by all criteria used, and note that the cost for a small genetic tagging program is similar to that for collection and processing of otoliths for the estimation of age composition. I also discuss a range of additional types of information that might be accumulated in the process of conducting a genetic tagging program.

6.2. Methods

To mimic the state of the NT Spanish mackerel fishery, in each simulation with FISHMOD I first used a historical effort sequence to generate a fishery that had been fished down to some extent. I then simulated forward over a 20 year period. Management and monitoring of the fishery began with the forward simulation, so that catch and effort information from the historical sequence were not applied in any way, except for the impact that it had on the reference population.

For the FISHMOD reference population, I set B_0 at 5000 tonnes, and adopted the population fecundity at age schedule used in Chapter 3 (Table 2.2), and the relative vulnerability at age schedule of Chapter 4 (Table 4.2). Recruitment variability was set high at $u_r = 0.6$. The reference population was divided into 5 spatially distinct substocks, with a spatial distribution indicated in Figure 6.1. Each substock had the same productivity. Movement parameters, identical between substocks, were as described in Chapter 2. These were set as the weight on random moves, $W_k = 100$, and weight on directed moves as $W_i = W_j = 10$.

The ratio $R_c = F_0 / F_{opt}$, the capacity of the fishery to exceed F_{opt}, was set at 2.0 and the catchability growth factor, R_h, was set at 4.0, to represent a very hyperstable fishery. This is expected of an aggregative species such as *S. commerson*.
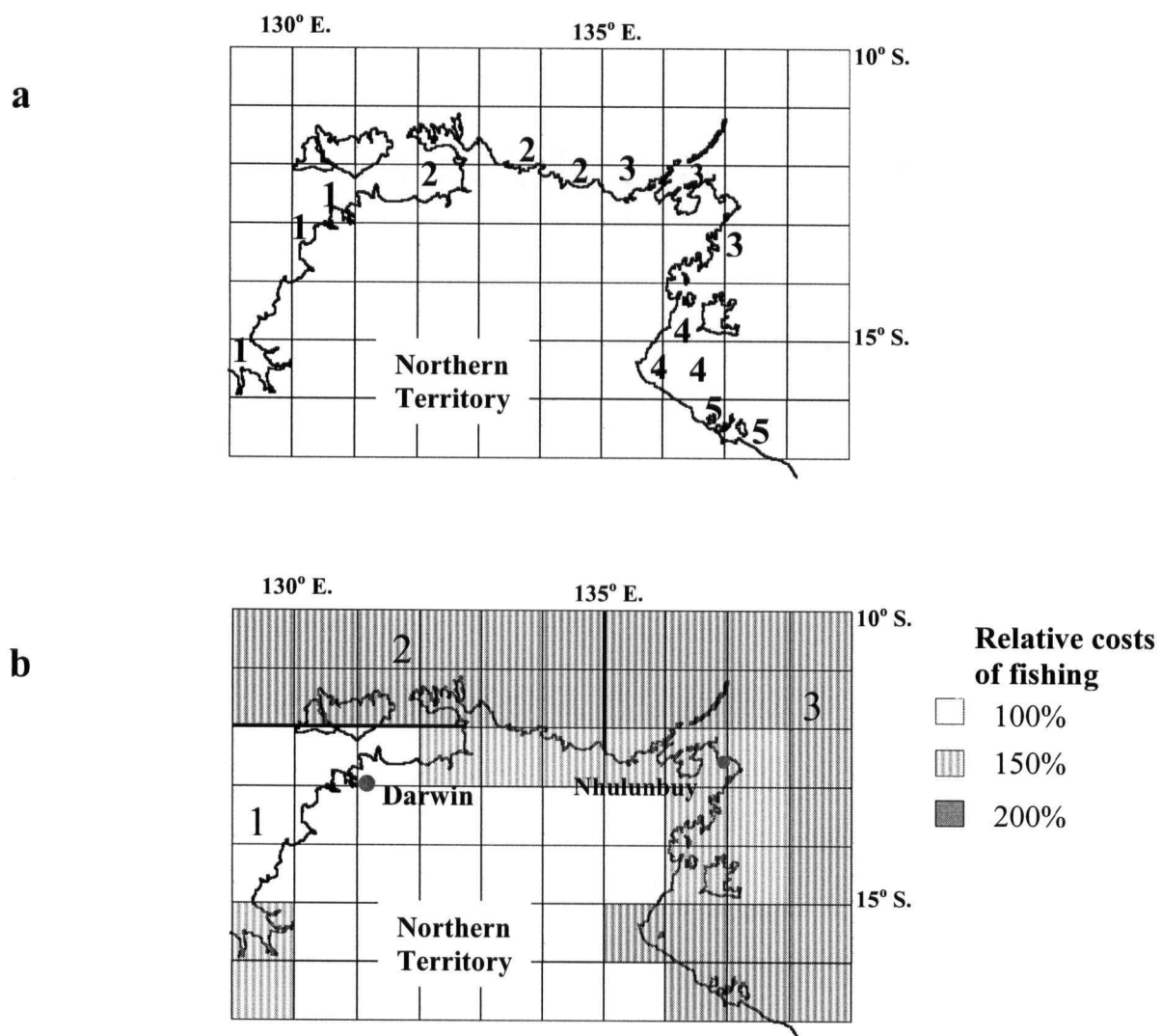
Figure 6.1. Maps of the Northern Territory indicating the spatial distribution of **a**, the hypothetical Spanish mackerel substocks based on nursery area location, with centres of adult spatial distributions being 1 grid cell offshore from nursery areas; and, **b**, the relative costs of fishing, and also showing the statistical areas, indicated by the large numerals, used for the fisheries managed as multiple areas.

Management control and monitoring

The fishery was managed as either 1 or 3 statistical areas, to establish whether there was any major difference in performance arising from this source. The statistical areas were defined on an "operational" or political basis and did not relate to the location of the substocks in any way.

Management was applied using the effort control option described in Chapter 2. This was modified to damp potentially catastrophic effort increases in any month by applying a constraint on the regulation factor of Chapter 2, reg_k. If at the end of any month, the effort that had been applied to the fishery in the area k was more than 10% greater than the effort for the previous year, then the fishery was shut down for the remainder of the year, ie $reg_k = 0$. Maximum effort for the fishery was set at 300 boatdays per month.

The management of the first three years of the forward simulation was modified. The TAC control option was applied for these years with an annual TAC of 450 tonnes. This provided an initial period of monitoring information for the forward simulation.

I chose monitoring methods and levels which are currently feasible for the NT Spanish mackerel fishery: a spatial CPUE (as in Chapter 2), collection of mean weight information for a sample of either 2000 or 5000 fish, and a mark-recapture program with various levels of genetic tagging (200, 400, 600, 800 or 1000 genetic tags released), and catch screening (5%, 10% 15% or 20%), combined with conventional tagging.

Parameters for the assessment model were the same as in Table 4.1, excepting that I set $B_{0min} = 1000$ tonnes, the minimum hypothetical biomass, and the number of hypotheses examined over the assessment grid as $n_{B0} = 200$.

For the conventional tagging, 0, 200, 400 or 600 tagged fish were released, each of which was also genetically tagged. The number of genetic and conventional tags released annually were divided evenly among statistical areas, as in Chapter 2, and the proportion of the catch screened was the same for all areas.

I assumed that the total catch was known through the logbook program. It was assumed that there would be complete reporting of the conventional tag recaptures.

Annual assessment of the fishery using the mean weight proceeded as in Chapter 2. The CPUE assessment was modified in that the power model approach suggested in Chapter 2 was implemented, as in Chapter 4 (Eq 4.1). The power term was again set at $b = 0.5$.

Tracking catchability using fishing rate information from combined genetic and conventional mark-recapture

For using genetic tagging as a monitoring method I followed the basic methodology presented in Chapter 2, where mark-recapture is used to track catchability, q. An annual observation from the mark-recapture program is used to update the estimate of catchability, using the estimate from the previous year in a Kalman filter (Walters 1986; Schnute 1994).

Throughout the following description, the subscripts k and t reference statistical area and year, respectively, as in Chapter 2.

An initial estimate of the catchability each year was $\hat{q}_{k,t} = -\ln(1 - X_g)/Effort_{k,t}$ (ie Eq 2.34), with $X_g = gtags_{ck,t}/gtags_{rk,t}/P_s$, an estimate of the annual harvest rate, and where $gtags_{ck,t}$ is the genetic tag release in that area and year.

The corresponding number of genetic tags detected, $gtags_{rk,t}$ was simulated by drawing from a binomial distribution with a probability of P_sU_{kt} ie the proportion screened multiplied by the harvest rate, which was estimated as the annual catch for the area divided by the vulnerable biomass at the beginning of the fishing year. This is as in Chapter 2, but accommodating the proportion of the catch, P_s that is screened. This is analogous to having a known reporting rate (e.g. Martell and Walters 2001).

In the absence of a conventional tagging component, the annual estimation of catchability from genetic mark recapture proceeded as described in Chapter 2, but weighting the number of tags released in all calculations by the proportion of the catch screened. The number of "effective tags" was considered to be $P_s \, gtags_{rk,t}$.

When a conventional tagging program was included in addition to the genetic tagging program, it was assumed that the samples for genetically tagged fish and conventionally tagged fish were discrete i.e. the set of genetically tagged fish were not subsequently sampled for conventional tagging. However, all conventionally tagged fish were additionally genetically tagged (e.g. from tissue retained from a fin clip), so that screening of the catch produced recaptures from both sets. Thus recaptures from the conventionally tagged fish could be of fish retaining conventional tags, or of fish which had shed their tag but were detected in the genetic screening process.

With information from the genetic tagging program, recaptures of conventionally tagged animals retaining tags, and also genetic detection of conventionally tagged fish that had shed tags, then it is feasible to annually update estimates of rates of tag shedding and mortality (due to the conventional tagging process) and make an additional harvest rate estimate from the conventional tag data. (Note that this process requires an input of the genetic tagging estimate

of the harvest rate.) A weighted mean of the genetic and conventional tag harvest rate estimates is then used to provide the annual observation of catchability (Eq. 6.1), and so update the state estimate of catchability with the Kalman filter. Details of the updating process follow.

I assumed that rates of tag shedding and mortality, which arose from the conventional tagging process, were constant over the duration of the simulated fishery monitoring program. I simulated the recapture of conventionally tagged fish as sequential binomial processes. For each of the $tags_{r,k,t}$ tagged fish:

1. The rate of mortality due to capture, marking and release, g_m was supplied as an input, so that the rate of survival of tagging was estimated by drawing from a binomial distribution with probability of $(1 - g_m)$;

2. For those that survived the tagging process, tag shedding was modelled by drawing from a binomial distribution with probability g_s, the tag shedding rate, that was supplied as an input; and,

3. For those that survived tagging, subsequent recapture was drawn from a binomial distribution with probability $U_{k,t}$, for fish retaining that tag, and $P_s U_{k,t}$, for those that had shed the tag and were detected in the genetic screening process.

This process generated $H_{c,k,t}$, the number of fish recaptured with conventional tags, and $H_{gk,t}$, the number of fish which had been tagged conventionally but were recaptured with tags shed. The recapture rate of conventional tags was

$$X_{c1k} = H_{c,k,t} / tags_{r,k,t} , \qquad \text{Eq 6.1}$$

and the annual estimate of the tag shedding rate was

$$\hat{g}_{sk,t} = \frac{H_{gk,t}/P_s}{H_{gk,t}/P_s + H_{ck,t}}. \qquad \text{Eq 6.2}$$

The estimate of the tag shedding rate was updated as the mean of the annual estimates, ie

$$\hat{g}_{sk,t|t} = \sum_{i=1}^{t} \hat{g}_{sk,i}/t,$$

I employed this estimate of the shedding rate with the conventional tag recapture rate and the estimate of the harvest rate from the genetic tagging program to calculate an annual estimate of the rate of mortality due to tagging,

$$\hat{g}_{mk,t} = 1 - \frac{X_{c1k}/(1 - \hat{g}_{sk,t|t})}{X_{gk}},$$ then calculating a mean mortality rate over the course of the

monitoring program as $\hat{g}_{mk,t|t} = \sum_{i=1}^{t} \hat{g}_{mk,i}/t$. The annual conventional tag estimate of the harvest

rate was

$$X_{c2k} = \frac{X_{c1k}/(1 - \hat{g}_{sk,t|t})}{1 - \hat{g}_{mk,t}}.$$

The estimated annual harvest rate for application in (Eq 2.34), to produce the annual catchability estimate was then

$$X_{gck} = W_g X_{gk} + (1 - W_g)X_{c2k} \qquad \text{Eq 6.3}$$

where the term W_g is the weight given to the genetic tag information. Initial trials indicated that a value of $W_g = 0.7$ was effective, so this value was used throughout the simulations here.

The annual estimate of catchability was

$$\hat{q}_{k,t} = -\ln(1 - X_{gck}) / Effort_{k,t} \quad \text{Eq 6.4}$$

and the update of the catchability estimate with the Kalman filter proceeded with

$$\hat{q}_{k,t|t} = \hat{q}_{k,t|t-1} + \lambda(\hat{q}_{k,t} - \hat{q}_{k,t|t-1})$$

(i.e. Eq 2.35), where the Kalman gain (Eq 2.36) is calculated as

$$\lambda = \sigma_V^2 / (\sigma_V^2 + \sigma_U^2).$$

I modified the expression given in Chapter 2 for the variance arising from the mark-recapture estimate of catchability estimate (Eq 2.37) to reflect the two types of tagging input,

$$\sigma_U^2 = X_{gck,t}(1 - X_{gck,t}) / [tags_{rk,t}(1 - W_g) / W_g + gtags_{rk,t} / P_s] / (Effort_{k,t})^2. \quad \text{Eq 6.5}$$

As in Chapters 2 and 3, I set $\sigma_V = 0.2$ for all trials.

An important assumption underlying the estimation of catchability with this process is that genetically or conventionally tagged fish (those that survive tagging in the conventional tag case) are unaffected by the tagging, so that their subsequent probability of capture is the same as that of the untagged population.

Effort series

The NT Spanish mackerel population has been subject to what is believed to be a moderate to heavy level of fishing since at the least the mid-1970s, with the maximum impact in the late 1970s-early 1980s period, and in the 1999-2001 period (Chapter 4). To mimic a population that had been fished down to a corresponding degree, I composed an effort and catch rate series that roughly parallels the effort history of the NT Spanish mackerel fishery, in the following way:

1. I began with the monthly effort series for the NT Spanish mackerel troll fishery referenced by spatial grid, for the period 1983 to 2001;

2. the values for 1980 to 1986 were increased by adding to them the values for the period 1995 to 2001 (ie the 1980 to 1982 distribution of effort was set equivalent to the 1995 to 1997 values, and the 1983 values were set equal to the sum of the NT Fisheries log book data distributions for 1983 and 1998, and so on); and,

3. the values for 1974 to 1979 were set to the logbook values for 1996 to 2001.

Simulations

The fishery was simulated 100 times for each combination of management arrangements (number of statistical areas) and monitoring methods and levels. For each simulation, performance indices were calculated over the period of the fourth to final (twentieth) years of the forward simulation sequence. These indices were L_{rel}, as defined in Chapter 2, the mean annual catch, the mean annual fishing mortality rate, and the number of years in which each substock was less than 20% of its unfished biomass. The mean and standard deviation over the 100 simulations of each of these indices were stored.

6.3. Results

6.3.1 Performance of monitoring methods

Mean loss L_{rel} values over the 20 year forward simulation period in the single statistical area fishery ranged from 3.84 (SD = 0.86), for 600 genetic and 200 conventional tags released, with 20% screening, up to 8.60 (SD = 1.45) for spatial CPUE monitoring. Even without additional

information from conventional tags, the genetic tagging methodology consistently produced

substantially lower L_{rel} values than the more conventional monitoring methods (Figure 6.2).

Even just 200 genetic tags and 5% of the catch screened produced an L_{rel} of 7.97 (1.52 SD) –

less than the value for CPUE monitoring. For the genetic tagging monitoring, these results were

consistent whether the fishery was managed as a single administration, or three (Figure 6.2a, b).

Performance of the mean weight and CPUE monitoring methods was poorer (larger L_{rel} values)

in the 3 statistical area fishery (Figure 6.2b), with mean weight monitoring with 2000 fish/year
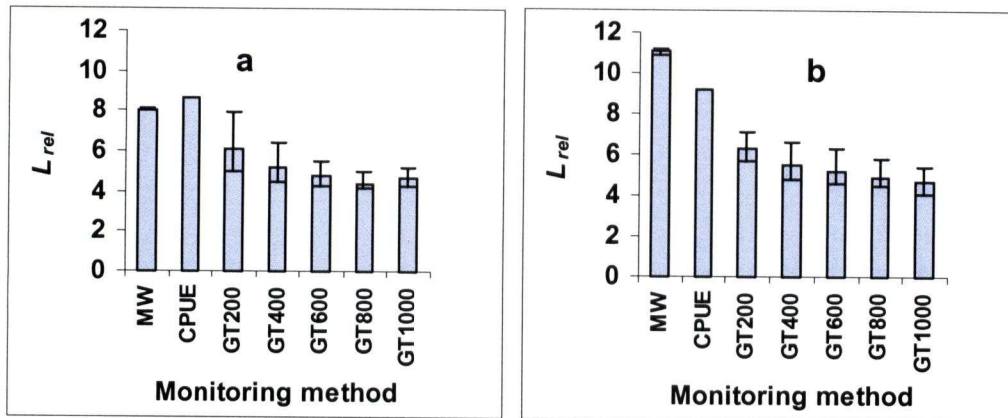
Figure 6.2. L_{rel} for the 20 year forward simulation, averaged over the various monitoring method scenarios. MW signifies mean weight monitoring, CPUE is CPUE monitoring and genetic tagging is the GT series, where the numerals indicate the number of genetic tags released. The solid bars are the mean L_{rel} over the range of sampling levels (of number of fish in weight samples, and the percentage of the population screened for genetic tags), and the error bars indicate the maximum and minimum of the means for the different sampling levels. For these scenarios there was no conventional tagging. **a**, Fishery managed within a single statistical area; **b**, Fishery managed in 3 statistical areas.

sampled producing a loss value of L_{rel} = 11.19 (1.26 SD), while L_{rel} for CPUE was 9.21

(SD 1.11).   As the genetic tagging results were very similar for both statistical area

configurations, I concentrate hereafter on the single statistical area fishery.

Example catch trajectories (Figure 6.3) indicate prolonged overfishing in the first decade of the

forward simulation when monitoring was with the CPUE and mean weight methodologies.

Nevertheless, the genetic tagging methodology resulted in only a modest increase in mean

catches (< 5%) over the 20 year forward simulation (Figure 6.4). However, these catches were achieved with a much lower average F in the genetic tagging scenarios – the mean of F for 200 genetic tag releases was 0.47; in the case of larger genetic tag releases, means were less than 0.40. Relative to the "true" $F_{opt} = 0.89$, the fishery was slightly underfished when management included the genetic tagging monitoring methodology.
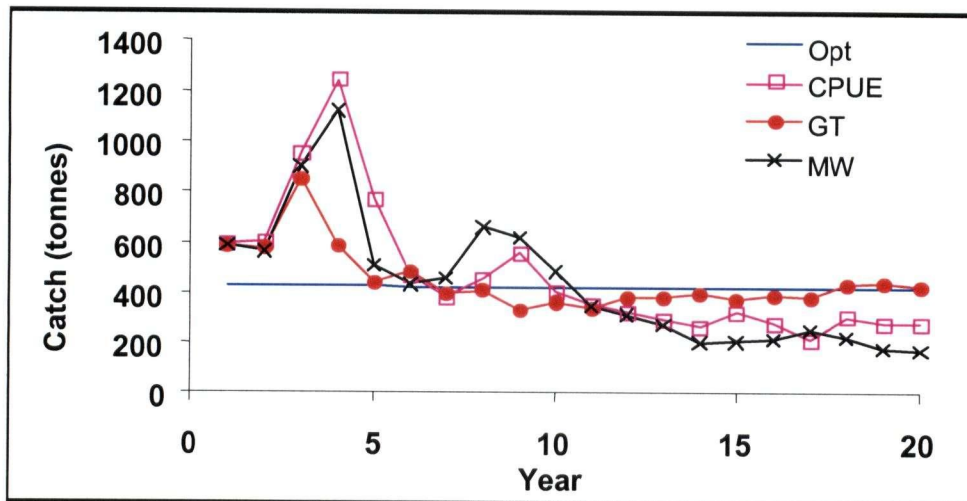
Figure 6.3. Example catch trajectories over a 20 year forward simulation in a 5 stock fishery managed as a single statistical area. Fisheries were monitored with CPUE; MW, mean weight with annual sampling of 2000 fish; and GT, genetic tagging with 400 tags released, 20 % screening and 200 conventional tags released. Opt indicates the optimum catch series.
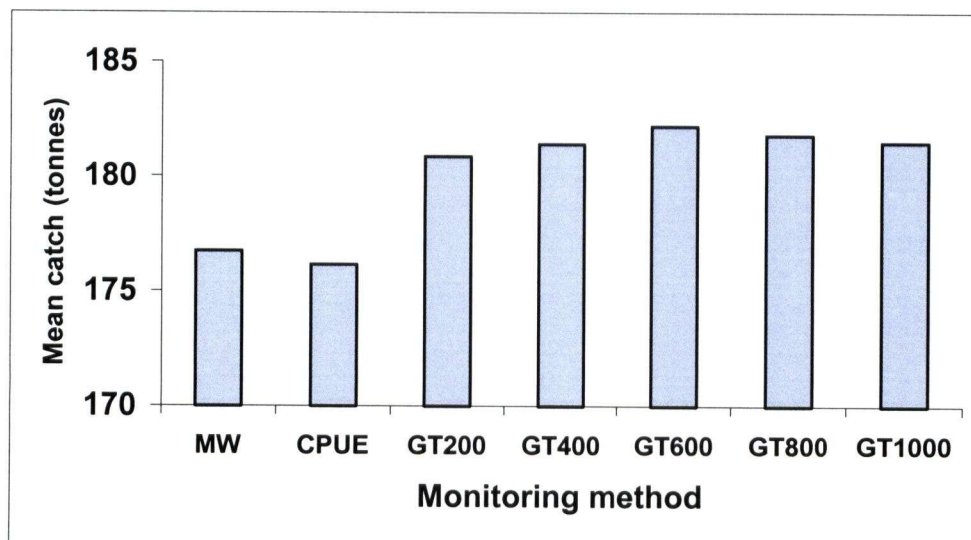
Figure 6.4. Mean catch over the 20 year forward simulation, averaged across monitoring method scenarios. Notation is as for Figure 6.2. Note that range of means for the levels within monitoring methods was ≈ 1% of these means.

For monitoring with the conventional methods, there was substantial overfishing, indicated by mean F values of 2.39 and 2.22 for CPUE and mean weight methods, respectively (Figure 6.5). These were substantially greater than $F_{opt} = 0.89$.

Overfishing in the fisheries using the conventional monitoring methods was also strongly indicated by the number of years for which substocks were depleted. These were summarized by taking the mean across substocks of the mean number of years for which each substock was less than 20% of initial biomass.
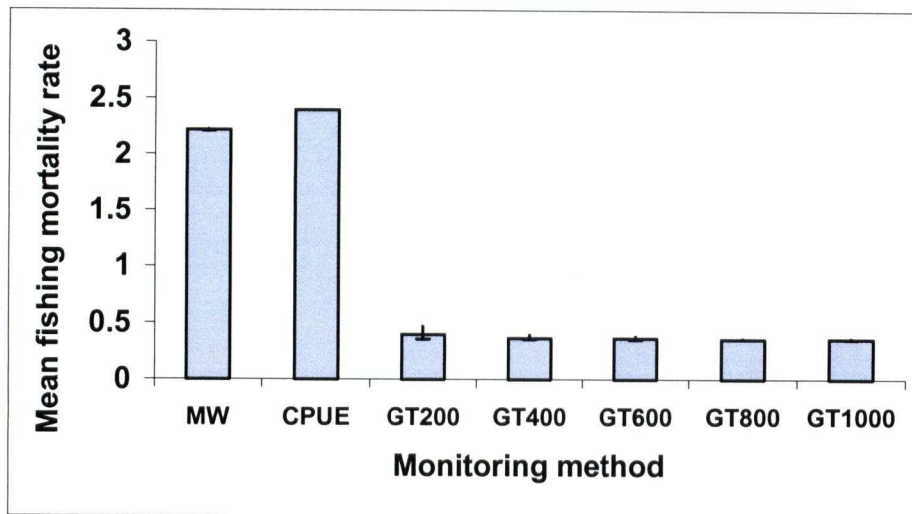
Figure 6.5. Mean fishing mortality rate F over the 20 year forward simulation, averaged across monitoring method scenarios. Notation is as for Figure 6.2.
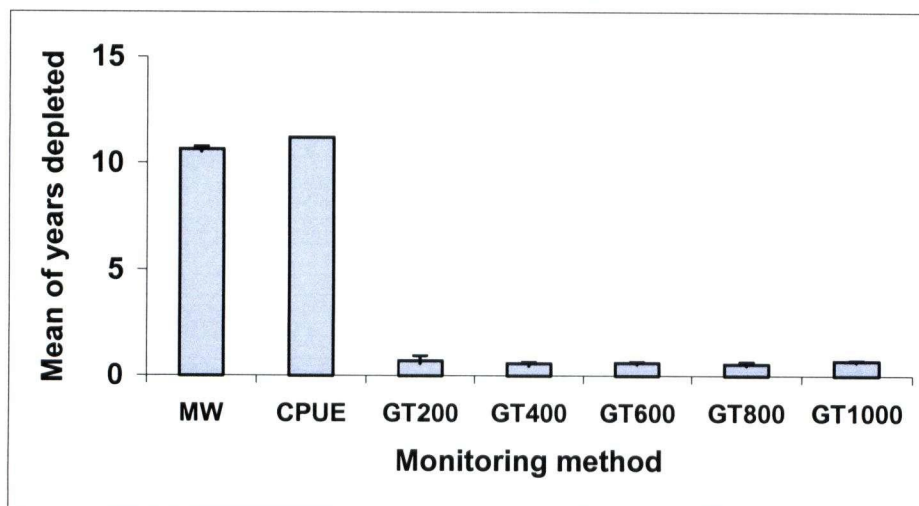
Figure 6.6. Mean of number of years in which substocks were depleted, over the 20 year forward simulation, averaged across monitoring method scenarios. Notation is as for Figure 6.2.

This mean was 10.6 years for the average weight monitoring and 11.2 years for the CPUE monitoring (Figure 6.6), indicating prolonged depletion of most substocks in each case. In contrast, the genetic tagging approach produced means across screening levels for depletions of less than 1.0 year for 200–1000 genetic tags released. However, the frequency of depletions increased as the number of tags released and the proportion screened decreased, so that there was a mean depletion of 3.55 years for one substock in the fishery monitored with the minimum 200 genetic tag release, 5% of the catch screened and no conventional tagging.
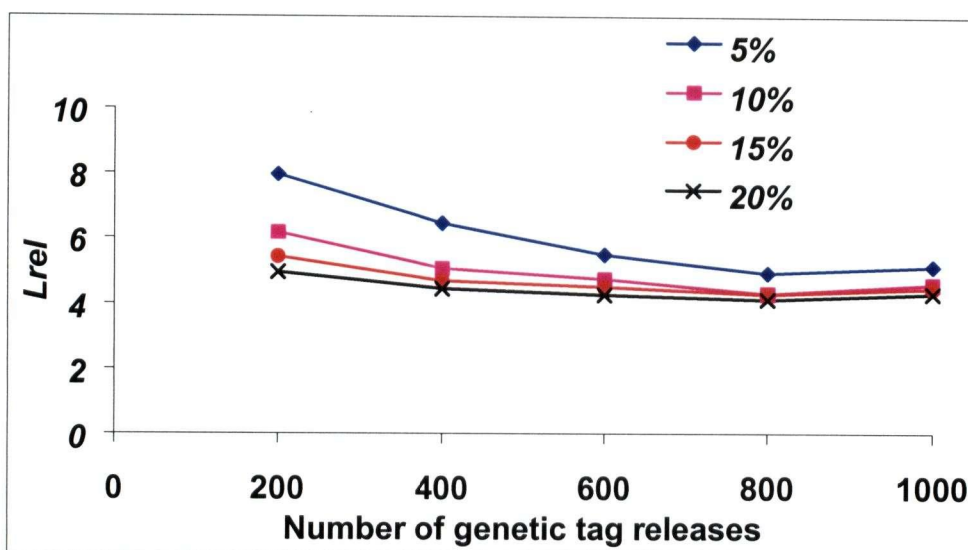
Figure 6.7. Mean relative loss measures, L_{rel}, over the 20 year forward simulation, for a range of numbers of genetic tags released, with 5%, 10%, 15% or 20% of the catch screened. These series do not include releases of conventional tags.

The relative performance (L_{rel}) of different combinations of the annual numbers of genetic tags released, and the proportion of the catch screened (Figure 6.7), indicated that performance generally improved (L_{rel} was smaller) with the number of tags released and the proportion of the catch screened. The performance with the minimum screening rate tested, 5%, was notably poorer than at the other screening levels, particularly for the releases of 200 and 400 tags. For the 200 genetic tag release, for example, L_{rel} was 7.97 for 5% screening, and 6.17 for the 10%

screening. However, the value of mean L_{rel} differed little when there were was a larger genetic tag release and more screening, i.e. there was an obvious diminishing return from increased monitoring investment for the high release-screening rate scenarios. For example, there was little difference in mean L_{rel} values between the combinations of 600 releases and 10% screening (L_{rel} = 4.76) and 1000 releases and 20% screening (4.30).

Combining conventional tags with genetic tagging improved performance (L_{rel}). The smallest number of conventional tags tested, 200, had the greatest relative impact, particularly for the lower numbers of genetic tags released (200-600) and the 5% level of screening (Figure 6.8). Thus for a genetic tag release of 200 with 5% screening, without release of conventional tags, L_{rel} declined from 6.13 to 5.17 with the additional release of 200 conventional tags. It is worthy of note that for some combinations, for example with genetic tag releases of 1000 (Figure 6.8), that an increase in the number of conventional tags actually led to a slight increase in L_{rel}.
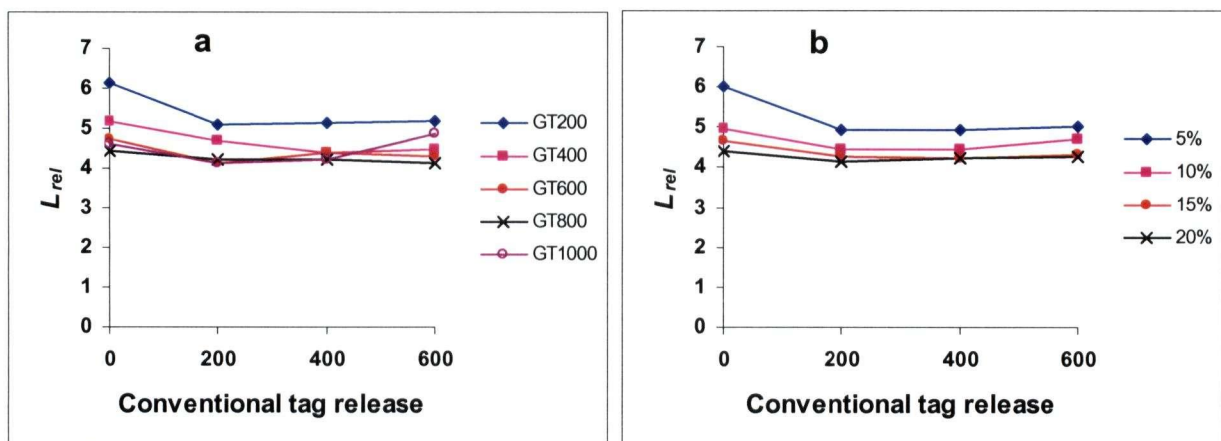
Figure 6.8. Mean relative loss measures, L_{rel}, over the 20 year forward simulation, for a range of numbers of conventional tags released **a,** Mean of L_{rel} over screening levels for genetic tag releases, with GT indicating genetic tagging and the numerals indicating the genetic tag release; **b,** Mean of L_{rel} over screening over levels of genetic tags released, the percentages indicating the proportion of the catch screened.

6.3.2 Costing of genetic mark-recapture for monitoring

In this section I provide indicative costs of genetic tagging as a monitoring method. Approximate costs for conducting genetic tagging of the NT Spanish mackerel fishery were assembled assuming the pre-existing infrastructure and pilot experimentation necessary to undertake routine monitoring. I have assumed that there would be pre-existing liaison and extension programs, and that the cost of logbook programs would be the same under any monitoring system, so the estimates presented here are additional marginal costs. I have excluded costs of analysis, data management and reporting, on the assumption that the ability to undertake these is part of available infrastructure. Thus I regard the marginal cost of obtaining catch and effort information as zero for CPUE monitoring; presumably mean weight monitoring, if not available through logbooks, could be undertaken for minimal additional cost.

I have understated the current cost of the remote tissue sampling devices in Table 6.1, on the assumption that they will rapidly reduce in price. Those described in Chapter 5 are prototypes, so that the real price should their use become routine cannot be known, but the materials used in their manufacture, excluding the re-useable lure bodies, were around Aust. $ 8.

I have assumed that the actual deployment of the sampling devices would be undertaken by the fishing industry. If this cannot be done then the cost would increase by about Aust. $50-80 per genetic tag, reflecting the cost of capture of Spanish mackerel in this fishery.

The dominant cost item is the genetic screening. The cost of $5 in Table 6.1 is based on routine sequencing being undertaken at a commercial facility, and efficient screening protocols currently under development (Dr J. Ovenden, Queensland Department of Primary Industry,

pers. comm.). With improved genetic techniques and increasing automation, I suggest that this cost is an upper limit.

I have assumed that the NT fishery will land 450 tonnes annually, at a mean weight of 7 kg per fish. I approximate the cost of genetic tag monitoring with the deployment of 600 genetic tags, screening of 10% of the catch, and an additional release of 200 conventional tags (also by volunteer fishers), at around Aust. $63000 annually (Table 6.1).

The variation in cost due to the genetic tag release and the proportion of the catch screened is presented in Table 6.2. The cost of even a small program is significant, for the small NT Spanish mackerel fishery. Future reduction in the cost of screening to $3 per sample is probable, achieving a substantial reduction in costs.

For the purpose of comparison, I have included the cost of obtaining age structure information for Spanish mackerel otoliths. A genetic tag release of 400 and 5% screened is similar in cost to routine age composition sampling based on 2000 fish annually (Table 6.3). This is substantially more than previous annual collections for the NT Spanish mackerel fishery (the 1991-93 age information of Chapter 4 was from 1547 fish), but might be necessary to accommodate spatial variation in age compostion.

The cost for preparation and interpretation of otoliths is at quasi-commercial rates (information provided by Mr John Ackerman, Central Ageing Facility, Marine and Freshwater Research Institute, Queenscliff, Victoria). Mackerel heads with information on size, sex and location can be provided by NT troll fishermen at a price of around $3.50 each (price provided by Mr Norman Hedditch, Chairman, NT Spanish Mackerel Troll Fishermen's Association).

Table 6.1. Approximate annual costing of genetic tagging as a routine monitoring method. Estimates are based on release of 600 genetic tags, 10% of the catch screened (6429 fish), and 200 conventional tags released.

Item	Quantity	Cost Per Item	Total Cost ($)
lab technician	1	5000	5000
lab expenses	1	5000	5000
fishery technician	1	5000	5000
genetic tagging devices	600	10	6000
fins collection	6429	0.25	1607
fins screening	6429	5	32143
genetic tags screening	600	5	3000
rewards, liaison and other incentives	600	5	3000
conventional tags	200	1	200
rewards & liaison	1	2000	2000
Total			**62950**

Table 6.2. Variation in cost (Aust. $ thousands) of genetic tagging as a routine monitoring method, as a function of the number of genetic tags released and % screened, for screening costs of $5 and $3.

% screened	Number of tags				
$5 screening	**200**	**400**	**600**	**800**	**1000**
5	38.1	42.1	46.1	50.1	54.1
10	55.0	59.0	63.0	67.0	71.0
15	71.8	75.8	79.8	83.8	87.8
20	88.7	92.7	96.7	100.7	104.7
$3 screening					
5	31.2	34.8	38.4	42.0	45.6
10	41.7	45.3	48.9	52.5	56.1
15	52.1	55.7	59.3	62.9	66.5
20	62.6	66.2	69.8	73.4	77.0

Table 6.3. Approximate annual costing of routine age analysis as a routine monitoring method. Estimates are based on purchase of 2000 heads, removal of otoliths and commercial preparation and interpretation of the material.

Item	Quantity	Cost Per Item	Total Cost ($)
otoliths	2000	16	32000
purchase of heads	2000	3.50	7000
technician and lab expenses	1	3000	3000
Total			**42000**

6.4. Discussion

The general results of the simulations of genetic tagging and other monitoring methods I have undertaken here were quite clear cut: by all the performance measures by which the methods might be evaluated, the genetic tagging approach was superior. For the simulated fisheries using mark-recapture monitoring, the primary performance measure, the loss function L_{rel}, was markedly smaller than for monitoring that relied on CPUE and mean weight, even for minimal genetic tag release and low rate of screening.

The improvement in mean catch with the genetic tagging scenarios was only marginal. Over the short time horizon of 20 years, the overfishing in the first decade in the CPUE and mean weight scenarios was not penalized by this measure. Simulation over a longer time horizon would have shown a larger relative mean catch by fisheries using the tagging approaches, as would overfishing of a less resilient species.

This was emphasised by the other performance measures: very high mean F values in the mean weight and CPUE monitoring contrasted with mean values less than F_{opt} for the genetic tagging scenarios. Equally impressive was the contrast in the depletion measure between the genetic tagging scenarios and the conventional methods: the mean across substocks of the number of years in which population biomass dropped below 20% of the unfished biomass was more than half of the time horizon for the conventional methods, and below 1.0 year for all but one scenario in the fisheries monitored with genetic tagging. There could be few fisheries where the monitoring and the management apparatus were sufficiently sophisticated to be able to monitor and control all substocks in a fishery. In the case of Spanish mackerel, it has only been with the

very recent application of an array of methods (Lester *et al.* 2001; Moore *et al.* 2003; Newman *et al.* 2003; Ovenden *et al.* 2003) that even the potential existence of the species' metapopulation structure has been revealed. Without mark-recapture and other techniques that capture spatial dynamics on fine scales, it will not be elucidated further. Thus far, it appears that mark-recapture (and genetic mark-recapture) is less likely to lead to the erosion of cryptic substocks than the other monitoring methods I have considered.

Although a conclusion from Chapter 3 was that multiple area management might improve performance, relative to single area management, that was not apparently the case in the results of this chapter. Performance in the mean weight- and CPUE-monitored fisheries was a little worse in the multiple area scenarios. As noted in Chapter 3, the effect of spatial complexities and their interactions with management arrangements are difficult to predict, but one of the general conclusions there was that multiple management areas may be a safer option than single administrations. This was not the case in this chapter. The difference may have arisen due to the extra constraint placed on the effort controls in the simulations of this chapter (the fishery was shut if the cumulative effort for the year exceeded the total effort for the previous year by more than 10%). Thus the advantage that multiple area fisheries had over single area fisheries, that effort concentration was limited, may have been removed in the scenarios of this chapter. The interplay of management arrangements and spatial complexity is difficult to predict, and the effects of spatial management arrangements need to be investigated thoroughly before being instituted. Also as in Chapter 3, the genetic tagging results were comparable between single and multiple administrations.

It was apparent that, for the fishery considered here, using ever-larger numbers of tags and screening ever-larger proportions of the catch would not continue to improve performance.

There was little reduction in L_{rel} for tag numbers greater than 600 and screening greater than 10%. Similarly, the real improvement in performance with the addition of conventional tags was when the genetic tag release was low. But this should not be surprising. Given the high F_{opt} of the simulated fisheries, small numbers of tags released are expected to be informative, because a large proportion are expected to be returned, with $F = 0.89$, the expected return when the fishery is fished optimally is 58% (the annual harvest rate) weighted by the proportion screened. For a less productive species or a fishery focussed on the capture of much younger fish, and thus having a lower F_{opt}, a larger tag release would be necessary to provide the same level of information (Martell and Walters 2001).

The purpose of this chapter was to provide a hypothetical manager with the information on which to base a decision whether to employ a genetic tagging protocol as a monitoring method in preference to other methods. Based on the comparisons here, genetic tagging was far more informative about the status of the fishery, and in all criteria but mean catch, this was sufficient to ensure superior performance. The weighting placed by the manager or management committee, on sustainability over short-term catch, might be a critical factor. Given that mean catches were much the same whether the fishery used CPUE monitoring or genetic tagging, it could be argued by some that there was no need to introduce a potentially costly new system. If sustainability were given prominence – as it should under the precautionary approach (FAO 1996), and as it must for an Australian fishery to gain export approval, then the genetic tagging approach should plainly be the choice over either CPUE or mean weight monitoring. Nevertheless, for a small fishery, the cost presented here is significant. One consideration is that, with an anticipated reduction in the cost of screening, fins or other material could be stored for future analysis, providing retrospective precision to harvest rate estimates.

There is substantial need for research into the means of extracting the maximum value and enhancing the information from the combined genetic/conventional tagging process described here, such as the weighting process for combining the genetic tag and conventional tag information, and the most effective means of updating the estimates of the tag loss rates. If rates change over time, for example, a Kalman filter or similar tracking technique might be appropriate. The calculation of the variance of the annual catchability estimate in the Kalman filter should be upgraded to accommodate conventional tag loss rates and the covariance component due to the use of the genetic tag harvest rate estimate to calculate the conventional tag harvest rate estimate. Techniques could be included for incorporating additional information on the conventional tag loss rates into the annual estimation process. Additional information from conventional tagging could be obtained at small cost, through the voluntary participation of anglers and fishing guides.

An alternative to a mark-recapture methodology would be the application of age composition information. The use of a delay-difference model here as the assessment model precluded the explicit evaluation of a monitoring system based on age. Substantial age sampling (2000 fish or double what might be a reasonable monitoring target) is likely to be similar to a small genetic tagging program. Such a large age sample might reflect attempts to compensate for spatial age composition variation. However such information may be of limited use. NT Spanish mackerel fishermen report substantial differences in size composition from locations separated by a few km. Even without age or size related spatial behaviour by the fish, differential fishing rates acting on different substocks would lead to different age compositions in those substocks, and hence to such large changes in sampled age-vulnerability schedules as to make the age composition information practically useless.

In the context of applying these results to the NT Spanish mackerel fishery, it should be emphasised that many of the input parameters of the simulations are uncertain. Above all, the spatial structure is largely unknown. A multiple substock structure was adopted to provide some agreement with what is now known of Spanish mackerel population structure, but the locations, movements, nursery locations and relative productivities are all unknown. The simulations of Chapter 3 demonstrated clearly that underlying spatial dynamics may interact strongly with monitoring and control systems. Many of the biological parameters, as discussed in Chapter 4, are poorly known.

A manager might also consider that there are other potential applications of the data derived from a genetic tagging monitoring program, analyses of which were beyond the scope of this thesis. One is that as tags are expected to accumulate over time, so that for a constant annual tag release, the number of fish that are tagged in the population will reach an equilibrium larger than the annual release (at least for fish with longevity greater than one year). The apparent total mortality rate needed to result in that equilibrium, along with the estimates of F, can then be used to estimate natural mortality rate (eg Dredge 1990). Genetically marked fish should accumulate in the population if a monitoring program is maintained over some years, with some tagged repeatedly over time, so that potentially it could be possible to derive information on the variability of natural mortality rates over time, and in relation to factors such as age and location.

Given good location data in logbook systems, it should also be possible to accumulate substantial information on spatial dynamics of the target population, potentially on a range of spatial scales. Fins collected for screening provide a very large, spatially-referenced database of genetic information, that may be used in analysis of spatial population genetics, and might also

be a source of information on size and age structure of catches. For a sexually dimorphic species such as *S. commerson*, even sex composition might be derived from fins.

Although this work is directed at the NT fishery, it clearly has potential application in other fisheries. Fisheries for Spanish mackerel elsewhere are likely to have the same management and monitoring problems as the NT fishery - preliminary evaluation of genetic tagging has already been included in management strategy evaluation for the Queensland Spanish mackerel fishery (Hoyle 2002) - and the ability to undertake genetic tagging programs should apply generally to large pelagics. The number of tag releases needed and screening required will depend upon the stock size and productivities of those fisheries. Development of the remote tissue sampling approach for demersal fisheries would also allow this methodology to be applied to reef fishes and other deeper water species, for which barotrauma is a problem that generally precludes effective use of conventional tagging.

7. CONCLUSIONS AND RECOMMENDATIONS

I began this thesis with a triplet of fundamental questions, that should be asked continuously throughout the course of a fishery. The current state of the world's fisheries is testimony to our inability to address these questions effectively. There are many reasons for this. I have focussed on one aspect - problems that arise from spatial complexity and inadequate data for stock assessment.

Simulating a series of small but spatially complex fisheries, I demonstrated that underlying spatial dynamics interact strongly with monitoring and control systems, profoundly affecting the behaviour of fisheries. Spatial dynamics of stocks and the distribution of fishing effort interact with management arrangements for those stocks at several scales. Fishing may be concentrated on aggregations of fish at scales much finer than a fishery is managed, with impact depending on the capacity of the fishery.

Concentration of effort on population units, occurring sequentially across value gradients, may result in serial depletion. To avoid this more subtle phenomenon, management requires information on the spatial behaviour of a fishery. Information on relatively fine scale dynamics allows the development of responses such as spatial and seasonal closures, or even larger-scale responses such as the use of large Marine Protected Areas, to contain fishing mortality rates at fine time and spatial scales.

Effort and TAC controls were equally effective in the more benign fishery scenarios that I simulated. But in those scenarios with large recruitment variation, high fishing effort capacity

and high hyperstability in catch rates, the quality of monitoring information became critical to fishery performance. TAC control was not resilient, but very dependent on the accuracy of stock assessments: poor information on abundance, such as that provided by CPUE, led to continued over-optimistic TACs and consequent entrenched overfishing. Effort control provided a more damped response to overestimation. Nevertheless, effective effort control also requires good definition of effort and catchability.

Quality information is necessary to ensure good management performance. This might be in the form of accurate indices of abundance, but I suggest that directly monitoring fishing mortality rate F and tracking changes in catchability q may be a more effective approach, and abundance estimation is not required except to calculate TACs or quotas. Simulated annual mark-recapture programs were used to annually estimate F and update an estimate of catchability, and so determine annual effort limits. This approach was generally superior to the other monitoring/ management schemes considered. It may be generally more effective to use the tagging information, making the simple set of assumptions required to ascertain the state of the fishery along with annual updates of catchability, than it is to employ the complex machinery of catch rate (commercial and survey) and size-age based stock assessment, with its encumbrances of assumptions about model structure and specification. This is an area that warrants further research.

Nevertheless, mark-recapture monitoring also comes with a set of assumptions that are difficult to meet. These assumptions are that tag-shedding and tag-induced mortality are well-quantified, that the tagging is representative over the vulnerable population, and that the reporting rate is exactly known. Moreover, many species of fish are too difficult to tag with the economy required of an ongoing, routine monitoring method. I recommend that genetic mark-recapture

based on DNA-fingerprinting techniques and remote tissue collection be evaluated for use as a monitoring tool, overcoming some of the limitations in conventional tagging.

The NT Spanish mackerel fishery exemplifies the problems of assessment of many fisheries. I found that uncertainty about the status of the fishery arises partly from its uncertain catch history. The available indices of abundance and composition, CPUE, mean weight and a small age composition sample, were not adequate to determine fishery status. There was no evidence to suggest that the fishery should be allowed to increase in size, and some indications that the fishery potential truly is quite small, perhaps even smaller than the current limit reference point implies. I recommend that more direct evidence of the fishery's status, such as that supplied by a tagging program, should be sought.

In response to this requirement, I have developed a methodology that should enable the use of a genetic mark-recapture monitoring program for this fishery. I demonstrated the feasibility of a new device for collecting fish tissue remotely, with the intention of providing the technical basis for genetic tagging as a monitoring tool. I applied this approach in a series of experiments to develop the basic device. I suggest that the designs could be used with little change for other species that attack lures and baits in a similar manner, such as other *Scomberomorus* species, and may be further developed for a wide range of species. The approach is simple. I therefore suggest that a genetic tagging program using the device could employ industry participation. This would entail daily tagging of a set number of fish, to address the problem of ensuring representative sampling of the fished population, by ensuring wide spatial and temporal distribution of tags, and may make monitoring economically feasible.

In a simulated, spatially-complex Spanish mackerel fishery, genetic tagging was markedly superior to CPUE and mean weight monitoring. Additional conventional tagging was able to

contribute useful information and would be most valuable in a monitoring program in which the number of genetic tags released and number of fish screened for fingerprints were fairly low. On the basis of simulated performance, I strongly recommend that the genetic tagging methodology be developed for monitoring the NT Spanish mackerel and similar fisheries. The cost of routine monitoring should be similar to a modest program based on the use of sectioned otoliths to provide age composition information.

I am unable to provide a definitive answer to the question of the current the status of the NT Spanish mackerel fishery, but in the absence of better information, it would be unwise to let the fishery grow larger. I have recommended a clear and relatively inexpensive means by which that information may be obtained, through direct measurement of F using genetic tagging.

How a fishery should be managed depends on the information obtainable. Of the options that I examined here, effort control was less dependent upon quality information than a TAC control. For the NT Spanish mackerel fishery, I recommend that the fishery move toward a management system where management targets, and information on status, are expressed and measured in terms of fishing mortality rates. In a broader sense, management of the impact of fishing, of fishing mortality rates, is what fishery control is all about. Logically, this suggests that management methods that control F by as direct a method as possible, having fewer assumptions, will be safer.

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