Modelling an abalone fishery system - implications for management

by

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Declaration

The candidate hereby certifies that:

- except where due acknowledgement has been made, the work is that of the author alone;
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- the content of the thesis is the result of work which has been carried out since the official commencement date of the approved research program;
- any editorial work, paid or unpaid, carried out by a third party is acknowledged; and,
- ethics procedures and guidelines have been followed.

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Raymond A. Bedford
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### Abbreviations

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<th>Description</th>
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<tbody>
<tr>
<td>AGGP</td>
<td>aggregate system perspective</td>
</tr>
<tr>
<td>CPUE</td>
<td>catch per unit of effort</td>
</tr>
<tr>
<td>ITQ</td>
<td>individual transferrable quota</td>
</tr>
<tr>
<td>HTM</td>
<td>harvest threshold mechanism</td>
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<tr>
<td>LML</td>
<td>legal minimum length</td>
</tr>
<tr>
<td>MASY</td>
<td>maximum achievable sustainable yield</td>
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<td>MSY</td>
<td>maximum sustainable yield</td>
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<td>SSB</td>
<td>spawning stock biomass</td>
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<tr>
<td>SUBP</td>
<td>sub-system perspective</td>
</tr>
<tr>
<td>TACC</td>
<td>total allowable commercial catch</td>
</tr>
<tr>
<td>TAE</td>
<td>total allowable effort</td>
</tr>
<tr>
<td>VBL</td>
<td>von Bertalanffy</td>
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Abstract

Australia currently has the largest commercial wild abalone fisheries in the world. However this industry is coming under increasing pressure due to declining profitability, mainly attributable to a sustained appreciation in the Australian dollar and rising operational costs (e.g. fuel, labor). Reduced harvests have also contributed to dwindling profit levels. Therefore it has become imperative to examine all management strategies to ensure that the wild abalone industry remains both profitable and sustainable. In particular management have expressed concern about meeting key management objectives in relation to i) biomass recovery following a high mortality event ii) alternative spatial scales of management and iii) the fishery effects of diver behaviour. These are the foci of this dissertation research.

To address these issues a detailed abalone population dynamics model adapted from biological research data was developed. The model, which categorizes individuals by shell length, was initially used to compare the efficacy of two newly proposed harvest control rules. The objective was to find which rule adapts best to changing fishery conditions. It was found that the proposed harvest threshold mechanism (HTM) rule was ineffective in regard to recovering stock following a high mortality event. Alternatively the proposed mean length of commercial catch (MLCC) control rule was able to respond to a large downward shift in biomass through setting appropriate harvest adjustments.
To investigate system behaviour at alternative spatial scales of management the population dynamics model was extended to include spatial differentiation. The main aim of this work was to measure the effect of preferential harvesting within a system by separating the system into two reefs of equal size where one reef was preferred by divers for harvesting. The finding was that the cost and effort associated with implementing management at finer spatial scales would not be justified, in general. However in the case of fisheries with reefs that are strongly preferred by divers, the adverse effects upon management objectives may necessitate such measures.

There is evidence to suggest that divers move among reefs in response to changing profit at each location. To investigate the effect of this behaviour on the whole system a bioeconomic model was constructed that explicitly included this diver movement. The findings of this model indicated that fisheries that operate at low biomass density and contain a substantial range of locational harvesting costs are susceptible to rapid biomass loss at some locations.

The models developed in this thesis and the findings achieved have yielded insight into an abalone system. This in turn, has led to a better understanding of some management strategies under consideration. It is hoped this will contribute towards improved management practices and hence, sustainability of Australian abalone fisheries. Furthermore, the findings may extend to certain other renewable resource systems.
1. Introduction

1.1 Background

Some renewable resources that are utilised by humans continue to be placed under considerable pressure. The rapid increase in world population over the last century combined with our commitment to industrialisation and technological advancement has led to unprecedented levels of human resource consumption. The depletion of timber resources in Ghana (Eshun, Potting, & Leemans, 2010) is explained by a combination of overutilization and the enormous wastage of wood in the production process. Depletion of natural resources is not exclusively a consequence of human consumption with predation from competitor species posing an additional threat to some resources (Read & Brownstein, 2003). In many instances natural events can have a profound effect on biologically reproducible resources, such as exposure of cattle populations to drought (Lesnoff, Corniaux, & Hiernaux, 2012). The renewable resource central to this thesis is a species of wild abalone that is a member of the marine gastropod family.

Wild abalone have biological features that are distinguishable from the majority of other exploited marine species and some of these traits require consideration when assessing abalone stocks. Although abalone can crawl over short distances they are mostly known as sedentary creatures. Therefore, population biomass flows from immigration and emigration can be ignored in a system model. The shell length of fully grown abalone varies markedly between different species and following intense
commercial exploitation over many decades it is now uncommon to find specimens from the largest species with a shell length in excess of 25 centimetres where there were once thriving commercial fisheries. The body of meat that occupies an abalone shell is an ovoid shaped muscular foot that can clamp securely onto rocky surfaces.

Abalone meat is highly regarded in Asian cultures which explain why commercial harvests of these species continue to fetch high unit prices (Raemaekers & Britz, 2009). The rapid emergence of farmed abalone production in recent years means that wild abalone production now accounts for less than 25% of the world market. Local aquaculture of abalone has played a growing role in the Australian export market over the past decade (Figure 1.1).

**AUSTRALIAN ABALONE FISHERIES: 2000 - 2010**

![Graph showing abalone production and prices](image)

**Figure 1.1** The combined reported total commercial catch and farmed abalone production in Australia over the previous decade are illustrated by the vertical bars. In 2010 the Australian commercial catch accounted for more than 50% of world wild abalone production while Australian farmed production comprised less than 4% of
world production. The increasing competition from aquafarming produce combined with the steady fall in the real unit price of abalone (black line) has placed the Australian wild abalone market under pressure.

The relative ease with which abalone are captured that arises from their lack of mobility is further compounded by the tendency of these shellfish to aggregate in clusters (Officer, Dixon, & Gorfine, 2001). Today the largest wild abalone fisheries exist in south-eastern Australia, where blacklip abalone is the most prevalent species. This species can be difficult for divers to detect due to a preference for inhabiting narrow rock crevices or hiding beneath rocky structures. However, from a commercial harvesting viewpoint, the component of abalone biomass that inhabits cryptic space mostly consists of individuals that have yet to reach a legally harvestable size.

The shoreline habitat in which abalone is mostly found is ideal for diver access to the fishery and it allows spatial identification of the fishery by land reference. In a wildlife sustainability study Weinbaum, Brashares, Golden, & Getz, (2013) recognise the loss of wildlife habitat as the greatest threat to many of our food resources. With respect to abalone the main threats to habitat are considered to be from environmental change (Neuman, Tissot, & Vanblaricom, 2010) and the presence of competitor species (Strain, Johnson, & Thomson, 2013). The close proximity of abalone habitat to land means that abalone fisheries continue to be central to ongoing debates about the ownership and stewardship of wild marine resources. A partial solution was formed internationally with the implementation of wide ranging laws by the United Nations Law of the Sea
Convention (1949 - 1982). During this time nations were enabled greater jurisdiction over their adjacent sea fisheries and given increased sovereign rights over existing marine resources within their territorial waters. The commencement of commercial exploitation of abalone in Australian waters coincided with the early stages of the United Nations overhaul of international fishing laws.

Central to this thesis is the management of commercial abalone fisheries that provide highly sought products to a number of seafood markets throughout the world. In particular, the Australian state of Victoria has a large abalone fishery by international standards. Within the first decade of production the reported catches in 1968 reached an all-time peak of almost 2,900 t; about 10% of world production (Mayfield, Mundy, Gorfine, Hart, & Worthington, 2012). Due to the rapid growth of the wild abalone industry, the Victorian government swiftly implemented limited entry of commercial divers, minimum legal sizes of harvestable abalone and zonation of the coast line under fisheries legislation. These measures were complemented by additional management tools such as compulsory commercial logbooks and acquisition of biological and catch per unit of effort (CPUE) data prior to 1970.

By the late 1980s Australia had defied overseas trends of declining production by becoming the premier supplier of wild abalone to the world market. At this time, following years of stable catch in the Victorian fishery, the management agency sought to provide further protection to abalone biomass with the introduction of an annual total
allowable commercial catch (TACC). This harvest control measure involved individual transferrable quotas (ITQs) equally divided amongst licence owners. As the fishery maintained healthy performance indicators during the 1990s, further management measures were introduced during this decade to shore up the valuable resource. These measures included expanding existing biological research, introduction of fishery-independent surveys sites and development of a stochastic population dynamics model (Gorfine, Taylor, & Walker, 2001). Notably, during this decade the declaration of State ownership of wild fish stocks in the 1995 Victorian Fisheries Act included severe penalties for unlicensed fishing (Kailis, 2013).

Critical to the recent past of Australian abalone fisheries has been the increasing role played by adaptive management strategies. This approach involves adapting statistical controls to a resource system in order to gain iterative knowledge about management objectives through system monitoring. The involvement of different stakeholder groups means that the process has both a scientific and social aspect to it. Practical implementation involves participatory workshops between fishery management, quota owners and divers. This process has enabled focus to be placed on important areas of emerging fishery management policy. Notably, assessment of the fishery at finer spatial scales from legislated management zones now involves the pooling of current knowledge by all parties. While this practice is in an infant state, the lack of precision associated with the assessment of stocks means that the setting of catch caps and minimum lengths remain voluntary (Prince, Peeters, Gorfine, & Day, 2008).
Industry and management continue to allocate resources to develop an increased understanding of how the managed zones function at these smaller spatial scales. Of particular interest to management is expanding current knowledge of the spatial behaviour of abalone divers (Gorfine & Dixon, 2001) through the use of technological advances in digital acquisition of information. This improves capacity for elucidation of the fishery effects caused by dynamic spatial movement of divers in their search for highly productive stocks.

While the workshop process is a useful tool for a fishery that remains in a healthy state, the true worth became apparent several years ago upon the discovery of disease throughout the Western Victorian zone that caused catastrophic losses of stock (Mayfield et al., 2011). In this circumstance the management system provided a structured response to the problem as the lines of communication already existed enabling detailed documentation of the damage to stock. Prior to the discovery of disease in the Western zone, the combined 2006 Victorian TACC was set to 1,328 t. Seven years later the annual TACC is persisting below 800 t (coincidentally still close to 10% of current world production) while the Western Zone is still coming to terms with what is hoped to be the early stages of a recovery period. In the next section, previous fisheries management reactions to the impacts of severe stock loss are reviewed with emphasis on subsequent changes to policy.
1.2 Management of depleted stocks

The increasingly sophisticated approach to scientific analysis of fisheries has enabled management agencies to better address instances of marine stock depletion which had become a growing issue among many commercial fisheries since the late nineteenth century. While it can be difficult to determine whether a single cause or multiple factors are to blame for the collapse of a particular fishery it is generally accepted that overfishing has played a prominent part in many collapses. The underlying causes of overfishing are investigated by Finley & Oreskes, (2013) who question whether it is the combined behaviour of individuals or government policy that should shoulder most responsibility for excessive harvesting.

The process of correctly identifying and taking steps to eliminate root causes of fishery depletion continues to be an elusive goal for fishery managers. One step in the process has been the development of population dynamics models to capture the core dynamics of changes in biomass, recruitment, growth, mortality, immigration and emigration. The estimation of these key parameters coupled with the interpretation of catch and biological data provide management with the information necessary to implement conservative harvest strategies (Richards & Megrey, 1994).

The chronology of literature regarding the collapse of the Peruvian anchovy fishery in 1972 highlights the evolution of modern fisheries management approaches. An assessment model was developed by Boerema & Gulland, (1973) to estimate the
maximum sustainable yield (MSY) of the fishery by fitting the combined predatory
effort of humans and birds to the catch data. Due to the slow recovery of the fishery a
separate stock assessment model was developed by Barrett, Caulkins, & Yates, (1985)
addressing possible causes of the collapse. This model measured the contributory
effects of the extreme El Nino event in 1972 combined with the ever present threat of
predation by birds and the authors proposed a seasonal harvest strategy and the
introduction of a quota system. Recently, the 1972 fishery collapse was modelled as a
bioeconomic system with simulation of the cognitive reasoning of stakeholders (Duncan,
Hepburn, & Papachristodoulou, 2011). The model outputs suggested that some of the
human behaviour at the time led to severe consequences for the fishery.

Whilst abalone have been commercially harvested for more than a century it was the
advent of compressed air diving equipment after World War II that led to greatly
increased harvesting capacity in sub tidal habitat. The high unit market value of abalone
immediately attracted high levels of commercial and illegal effort in abalone fisheries. It
is now apparent that this increased accessibility of abalone stocks led to unsustainable
fishing of stocks. This culminated in an all-time peak of world abalone production
during the late 1960s estimated at around 25,000 t (Cook & Gordon, 2010). In the decade
that followed, lucrative abalone fisheries throughout the world showed signs of
substantial stock depletion.
The Californian coastline harboured a complex abalone system containing five commercially harvestable species. The fishery was eventually decommercialised in 1997 (Hobday, Tegner, & Haaker, 2001) after a long struggle to maintain economic viability. While overfishing is cited as a chief cause, the authors acknowledged the difficulties in concurrently managing multiple species as a single fishery. After fifty years of commercial harvesting the South African abalone fishery was closed under different circumstances. In this case a single species that initially produced extremely high yield succumbed to illegal harvesting activity (Dichmont, Butterworth, & Cochrane, 2000).

The Mexican fishery which contains two commercially exploitable species demonstrates that collapsed abalone fisheries can be rehabilitated. Twice there have been strong recoveries following fishery closures due to stock depletion (Shepherd, Turrubiates-Morales, & Hall, 1998).

In all these cases of abalone fishery depletion there were varying management attempts to conserve stocks in this time. In South Africa a legal minimum length of catch was applied in 1953, followed by various catch controls in the 1960s and seasonal fishery closure from 1985. In Mexico the influence of fishery management prior to the first stock collapse was diluted due to the lack of size limits enforcement (Shepherd et al., 1998). Hobday et al., (2001) apportion much of the blame for the sequence of Californian fishery collapses culminating in the listing of white abalone as a threatened species on a lack of management resources. Unsophisticated generic approaches to management
resulted in the implementation of blanket strategies that ignored varying biological characteristics particular to individual species.

The economic and cultural implications arising from the decline of many world abalone fisheries were strong enough to trigger a more detailed inspection of abalone fisheries. Management in Mexico introduced fishery-independent transect surveys in 1988 to complement CPUE data that was used to estimate abundance. Gathering of key biological population data that could be fed into stock assessment models also became a priority. Growth studies using the tag-recapture method (Haaker, Parker, & Chun, 1995; Tarr, 1995; Worthington, Andrew, & Hamer, 1995) and length frequency data (Siddeek & Johnson, 1997) all captured informative biological information. Many of the lessons learned from these fisheries were subsequently applied to the management of Australian and New Zealand abalone. In the following section a review of the literature concerning recent developments in key areas of fisheries management is undertaken.
1.3 Recent fisheries management developments

Overview

During the years following the conclusion of the United Nations Convention on the Law of the Sea, countries began exercising their exclusive rights to an economic zone of up to 200 nautical miles from shore. At this time, the level of sophistication in fisheries management varied greatly between countries as they prepared to escalate existing management of their fishing waters. In a review of marine fisheries management during the period 1985–2010, Hilborn, (2012) acknowledges that the difficulty faced by all countries during this period of management transformation was compounded by signs of overexploitation from decades past. At the commencement of the review period the management of marine stocks was already well underway in some fisheries due to the existence of scientific institutions for marine data collection, assessment and management. At the other end of the scale, some developing countries had no fisheries management in place at that time.

The growing sophistication of harvest strategies and computer simulation techniques over the last three decades is identified by Hilborn, (2012) as a significant area of fishery management evolution. During this period a rapid increase in the breadth and depth of fisheries analysis was also enabled by increased computing power combined with new and improved statistical methods (Richards & Megrey, 1994). This led to techniques such as management strategy evaluation becoming a useful tool that assesses the impact
that harvest strategy has on fisheries (Bastardie, Nielsen, & Kraus, 2010). This methodology involves the design of an operating model that combines population dynamics, data collection and analysis, stock assessment method and a harvest control rule that drives the actions of fisheries management.

**Harvest control rules**

The implementation of harvest control rules is a fundamental requirement of fisheries management policy. Deroba & Bence, (2008) summarise control rules as falling into three main categories;

- constant catch (fixed tonnage)
- constant catch rate (fixed rate relative to biomass density)
- constant escapement (involves taking all biomass above a specified target level)

Hybrid control rules have been developed from these primary control types to strengthen a vulnerable area of a rule with the intention of providing further protection to fishery biomass. These include the threshold harvest control rule which is the constant catch rate rule with provision for fishery closure if fishery abundance falls below a specified biomass threshold and a conditional constant catch rule which switches to the constant catch rate rule when the catch rate reaches a pre-determined upper threshold rate. It is common fisheries science practice to synchronise threshold levels designated by management with biological reference points (Katsukawa, 2004).
More recently, the threshold rule has been further refined into a sub category of biomass-based rules which essentially scale the catch rate downwards when population biomass falls below a threshold and upwards in response to biomass gains above a threshold level. One commonly used biomass-based rule linearly indexes the prescribed catch rate against a target catch rate on proportionate change to biomass abundance within a specified target biomass range. In the assessment of the Australian Commonwealth Harvest Strategy Policy by Wayte & Klaer, (2010) the upper threshold is determined by the estimated equilibrium biomass size that sustains maximum yield and the low threshold harvest biomass size is set to 50% of the upper threshold. The prescribed catch rate is indexed against a target catch rate that can sustain biomass at a size that exceeds the upper threshold.

In a study of lobster fishery management, Zhang, Chen, & Wilson, (2011) compare the system behaviour of discrete catch rate adjustments under the threshold control rule with the continuous catch rate adjustments of the more recently developed linear biomass-based rule. The model results found that the continuous rule proved to be superior with regards to longer term sustainability of the fishery. This finding was further explored by Zhang & Chen, (2012) who measured the relative performance of nonlinear variations of the continuous linear biomass-based rule. In a theoretical study Franco & Peran, (2013) consider a newly proposed target oriented harvest control rule. This rule is a refinement of the escapement rule where the off-take of biomass that exceeds a target level is regulated by a multiplicative control parameter. Importantly the
control parameter provides management with added flexibility making the rule suitable for adaption to multiple fisheries.

**Management strategy evaluation**

The evaluation of fisheries management strategies is particularly useful in identifying distinct differences in system behaviour under alternative strategies. Alternative strategies can consist of two or more variations of a basic strategy or a number of independent strategies that bear little resemblance with each other. These evaluations typically compare the effects of harvest control rules on a fishery with simulation of the interaction between the operational component of a fishery system and a management module that can be modified to incorporate alternative control rules. It is common practice to compare the performance of an existing control rule with modified rules that capture a certain feature of a fishery system over multiple scenarios that are relevant to the modification.

The influence of the environment on fisheries has been a source of harvest control rule improvisation in recent times. The effect of sea temperature on the Japanese sardine (Hurtado-Ferro, Hiramatsu, & Shirakihara, 2010) was evaluated by the system performances of the constant catch rate control rule and an environmental harvest control rule that relies on the sea temperature as feedback to the catch rate adjustment process. It was found the environmental rule outperformed the constant fishing mortality rule over various recruitment scenarios with respect to long term catch.
Further work on the interactions between harvest control rules and environmental conditions was performed by Brunel, Piet, van Hal, & Rockmann, (2010) who compared the constant catch rate rule under deterministic recruitment with an environmental control rule that varies harvest in line with recruitment trends resulting from environmental change. It was found that following detrimental environmental change the environmental rule markedly outperforms the constant catch rate rule. In contrast to the previous findings Walters & Parma, (1996) contend that it may be more cost effective to concentrate on understanding how to successfully apply fixed harvest strategies to fishery systems than to invest in understanding the complexity of climate change and its effects on fisheries.

A recent theme in management strategy evaluation literature is the incorporation of uncertainty that surrounds the fishery and management processes within the modelling framework. A particular aim of these studies has been to identify management strategies that are robust to uncertainty (Dichmont et al., 2008). Some areas of uncertainty that have come under scrutiny pertain to model parameters, implementation of management strategy, life history of fish species and environmental conditions (Hurtado-Ferro et al., 2010; Ianelli, Hollowed, Haynie, Mueter, & Bond, 2011; Tong, Chen, & Chen, 2013). While most evaluation studies involve simulation of more than one management strategy under the same operational conditions, there exist other plausible evaluation methods. In the study by Yakubu, Li, Conrad, & Zeeman, (2011) a comparison between the responses of two fisheries to the constant fishing mortality rule demonstrates that
species with distinct biological differences react differently when subjected to the same harvest control rule.

**Management at fine spatial scales**

The analysis of fisheries at more spatially explicit scales is a key area of fisheries management that has received some attention in the literature in recent years. By example Ramirez-Rodriguez & Ojeda-Ruiz, (2012) set out to determine the boundaries of finer management zones based on the landed catch of multiple species, whilst a study on the varying states of maturity of the common whelk, *Buccinum undatum* L., on a spatial scale concluded that fine scale management should come under serious consideration (Shelmerdine, Adamson, Laurenson, & Leslie, 2007). An analysis about the effectiveness of a global Total Allowable Catch (TAC), where mixing and migration occurs amongst subpopulations found only guarded support for finer-scaled TACs (Holland & Herrera, 2012). In contrast Hobday et al., (2001) directly lay blame for the serial depletion of the white abalone species on the lack of spatial analysis of commercial catch. Wilson, Hayden, & Kersula, (2013) observed that ground fishing management in New England has failed to manage at a relevant spatial scale.

Studies on the management of fisheries at fine scales have also centred on measuring the effects of marine spatial enclosures. Dichmont et al., (2013) believe that while enclosures achieve their initial objective the benefits may not flow to higher level objectives of the fishery. Furthermore, the authors suggest that fisheries would benefit
most from implementation of a variety of management tools rather than overcommitting to one strategy. A spatial study on the age structure of a fishery by Edwards & Plaganyi, (2011) found that younger sections of the population need protection in shallow areas to ensure sustained recruitment into the older cohorts of fish.

Spatial analysis of the Galapagos fishery (Bucaram, White, Sanchirico, & Wilen, 2013) predicts potential consequences of fisherman behaviour under management at a finer scale in a fishery that has suffered from ineffective management techniques. The authors suggest that as management implement new policy they should be mindful of both fleets sharing the same preferred fishing ground which is closer to the home island of one fleet. The diversity of views on fine scale management suggests each study needs to be taken in the context intended on a case by case basis.

Fishery bioeconomics

Another area of fisheries management that is receiving growing attention is the role played by economic factors. Fishery bioeconomic theory links original fisheries evaluation methodology concerning the interaction between fishing intensity and biological constraints to the underlying dynamic economic forces that drive a fishery. In an overview of bioeconomic resource modelling Conrad & Smith, (2012) highlight the growing importance of spatial bioeconomic models that give recognition to location as a part of the management process. Much of this work has been motivated by the introduction of no-take nature reserves where the spatial migration of fish in and out of
reserves adds complexity to the modelling process. Flaaten & Mjolhus, (2010) discuss the further difficulty of incorporating involuntary migration caused by sea currents from one area of a fishery system into another. However the inclusion of migration is unnecessary in modelling of abalone populations due to the sedentary behaviour of abalone species that is characteristic from the early juvenile stage through to full maturity. In reality, abalone populations share some ecological characteristics in common with terrestrial species (such as plants). This includes clumped distribution patterns closely related to fine scale topography and localised dispersal of fertilised gametes. It is in this context that abalone fisheries management differs from the mainstream finfish management strategies reviewed in this section.

There has been a growing body of literature about fisheries bioeconomic studies in recent times. The economic feasibility of stock enhancement to an abalone fishery (Hart, Strain, & Hesp, 2013) was modelled over scenarios that varied by release volume, creature size at release and fishing mortality rates. The results were highlighted by strong economic growth following the injection of an abundance of two year old abalone into the fishery from a hatchery. Bioeconomic impact is used as a fishery performance measure in the management strategy evaluation conducted by Garcia, Urtizberea, Diez, Gil, & Marchal, (2013) where three harvest control rules are assessed to determine the best management outcomes for data-poor deep water fisheries. Bunnell, Lipton, & Miller, (2010) simulate economic scenarios that eliminate harvest for specific markets in a crab fishery with enforcement of seasonal closure and maximum size limits. It was
established that economic gain could result from the early closure of the female crab fishery. The economic study undertaken in this dissertation differs from these studies in that it deals with the effects of economic change on fishery performance instead of its effects on fisheries management policy. Further to this, most of these studies consider a fishery as a whole whereas in this dissertation a spatial component is also incorporated into the system model.

A bioeconomic model was developed to reconstruct the historical events of the collapse of an anchovy fishery in 2005 (Lazkano, Nostbakken, & Prellezo, 2013) demonstrating the subtle, yet important, role that economic dynamics play in a fishery. The methodologies applied to the internationally managed fishery comprising two fleets are along similar lines to those used in the Galapagos fishery model (Bucaram et al., 2013), highlighting that discrete models can be entirely appropriate for modelling some fishery characteristics. While the cause of the fishery collapse was due to a number of poorly set TAC amounts, the model results highlight the gross economic inefficiency that took place. The study aim was to identify how quota could be shared by the two countries upon resumption of harvesting so that the fishery is managed in a more economically viable manner. A review of the fisheries management methodologies used to formulate the models developed in this thesis follows in the next section.
1.4 Fisheries management methodology

Stock Assessment

Fisheries stock assessments are processes that use population dynamics models to estimate the trend in population abundance of a fishery. Much of the motivation behind performing stock assessments on fisheries came from the incidence of marine stock depletion with the advent of commercial fishing. One of the first successful implementations of stock assessment into a commercial fishery was applied to the Pacific Halibut species in 1923 (Hilborn, 2012). Decades later, the theoretical work on fishery dynamics produced by Beverton & Holt, (1957) became influential in fisheries management. The highly mathematical approach empowered fisheries management with a new set of tools that paved the way for further development of stock assessment modelling. One of these tools was the use of biological reference points as performance measures of the assessment process.

Modern stock assessment methods in fisheries management also incorporate flexibility that can cater for almost any kind of observation. In particular there is a growing trend for stakeholder input into the assessment process (Dunlop & Mann, 2013; Gilman, Owens, & Kraft, 2014). The benefits of these projects have wider positive implications for fisheries as they lay the foundation for improved relationships between fishery scientists and industry (Armstrong, Payne, Deas, & Catchpole, 2013).
Currently one of the most widely used stock assessment techniques is based on a statistical catch-at-age or catch-at-length approach. Modelling of wild abalone populations involves length-based populations since there is no simple, reliable and cost-effective technique for determining age (McShane & Smith, 1992). Length-based population modelling has previously been applied to Tasmanian rock lobster species (Punt & Kennedy, 1997), Australian northern prawns (Punt et al., 2010) and estimating growth of mangrove cockle (Flores, 2011). However, an age structured assessment model has been used to explain illegal catch effects on the South African abalone fishery (E. Plaganyi, Butterworth, & Burgener, 2011; E. E. Plaganyi & Butterworth, 2010). In this instance, the age-length relationship must be inferred from growth.

Following decades of research on abalone species it was acknowledged that some biological detail is not well understood from a quantitative perspective. Knowledge of the recruitment dynamics of abalone populations is severely limited due in part to the difficulty in quantifying recruits in cryptic habitat (McShane, 1995). Further difficulty in this area was encountered by Shepherd, Rodda, & Vargas, (2001) who contend that even small localised areas of an abalone fishery contain a family of stock-recruitment relations. In addition, wide temporal and age cohort variations are discovered in the process of quantifying instantaneous natural mortality rates (McShane & Smith, 1989).

In recent years the development of Bayesian techniques has enabled the estimation of unknown probability distributions of uncertain biological model parameters (Stewart et
al., 2013). Statistical computational advancements such as these techniques have led to several applications of models to abalone populations in recent years, including a length-based model by Breen, Kim, & Andrew, (2003) in New Zealand and Gorfine et al., (2001) in Australia.

**System Dynamics methodology**

The thesis includes components of System Dynamics methodology as an underlying approach to address some issues currently faced in the Victorian abalone fishery. This mathematical modelling technique was chosen in order to increase understanding of the effects that stem from a variety of sources on the dynamic behaviour of a fishery system. The methodology of system dynamics can provide elegance in illustrating differences between the consequences that arise from alternative management strategies in instances where there are clear contrasts in their effects (Randers, 1980; Sterman, 2000). According to this methodology, the system response to a small shock or disturbance is often enough to reveal problems with the inherent structure of a system or to reveal why one system should be preferred to another. This approach is different from ‘forecasting’ models where the output (i.e. the forecast) is dependent on the precise predictions of the parameter values. Further, as revealed by many System Dynamics case studies, the dominant modes of system behaviour are usually not very sensitive to parameter values (e.g. Schroeder, 1975).
While there is no evidence of an application of system dynamics to abalone fisheries in the literature, the methodology has been applied to other shellfish fisheries (Bald et al., 2009; Bueno & Basurto, 2009; Yang, Fu, & Cullen, 2013). The analysis of the resilience of a Mexican shellfish fishery found that these fisheries can rapidly collapse upon reaching critical threshold levels and the occurrence of incremental system behaviour nearing a tipping point provides less warning signs of impending system danger than substantial environmental system shocks. In a study of the Manila clam fishery the dynamic system responses to key management strategies were measured where it was found that minimum legal size for capture is of most importance. In the New Zealand Bluff oyster fishery the dynamic effects of ITQ-based self-governance upon the system are measured.

In the case of the oyster fishery, Yang et al., (2013) identify System Dynamics as a suitable approach for measuring fisheries management as it allows for the inclusion of multiple fishery aspects and importantly this methodology provides feedback from critical parameters that influence the system. The incorporation of bioeconomics into the system model found this form of ITQ-based self-governance to be desirable due to correlation between profitability and stock abundance. The usefulness of the system dynamics approach in providing insight into system behaviour that stems from management policy is demonstrated by Garrity, (2011) who constructed a number of models in a qualitative analysis of fisheries with ITQ-based management.
1.5 Thesis outline

The economically lucrative wild abalone fisheries found in south-eastern Australia form the foundation of an industry that is of high importance to many people. It is evident from the literature that the management of ocean fisheries is faced with a range of significant problems. Abalone fisheries are particularly vulnerable to some of these issues as highlighted by the disappearance of some large-scale commercial fisheries over the last four decades. The aim of this thesis is to explore the problems that confront management in the context of key management objectives; sustainability of biomass and harvest rates, consistency of yield and cost efficiency.

Any substantial loss of biomass in a commercial fishery due to a high-mortality event presents many challenges for management as they attempt to find a balance between biomass recovery and the commercial needs of industry. The stock assessment model presented in Chapter 2 is designed to evaluate two separate management strategies. One strategy uses a threshold-based harvest control rule to determine commercial catch for the upcoming season while the alternative strategy uses the mean length of commercial catch as a novel performance measure. The implications for management are based on; which harvest control rule will recover sufficient biomass that produces consistent harvest rates into the future?

The proximity of abalone species to shoreline in combination with their sedentary nature makes abalone fisheries suitable for monitoring at finer spatial scales. However,
there have been difficulties turning this opportunity into a meaningful advantage by obtaining comprehensive accuracy of biological information or stock assessment. In Chapter 3 a comparison of system behaviour at different management scales is performed. In this case, the stock assessment model simulates the practice of large scale spatial management of an area that consists of smaller scale spatial harvest regions and an alternative practice, whereby a management area and a harvest region are synchronised at a similarly small scale. This additional layer of governance provides motivation for the thesis to address the issue of which spatial scale of management is most suitable for attainment of management objectives.

The literature demonstrated that in the case of the Peruvian anchovy fishery the behaviour of humans is perceived to have had dire consequences for the fishery. This provides motivation to analyse the behaviour of abalone divers in Chapter 4. The inclusion of economic detail into the model enables specific measurement of system effects that are caused by the profit-driven behaviour of the divers. The effects on key variables relating to management objectives are assessed over both small and large spatial scales. Finally, this work is followed by a discussion of the key findings of the thesis with implications for management which appears in Chapter 5.
2. Evaluating alternative management strategies for abalone 1

2.1 Introduction

There has been a considerable number of northern hemisphere wild abalone fishery collapses over the last half century (Hobday et al., 2001; Lessard & Campbell, 2007; Prince & Delproo, 1993). In Australia, recent years have been challenging for the wild abalone industry. The Victorian fishery in particular, has seen the 2002 annual TACC of 1440t reduce to 741t in 2011. The drop in TACC, along with unfavourable changes among economic factors, has resulted in a AUD$50M loss in gross value of production (GVP). Despite being in decline, Australia currently has the world’s largest commercial wild abalone fishery (Cook & Gordon, 2010) with Victoria contributing just under 10% of total world production.

As a consequence of the current position faced by Australian abalone fisheries, it is more important than ever that effective management strategies are implemented. In this study, we consider two alternative strategies that determine the intended commercial catch for the next fishing season. Both strategies are evaluated by the following performance criteria:

- yield;
- consistency of yield;

- spawning stock biomass (SSB).

Wild abalone populations can experience substantial population losses ("shocks"), with catastrophic effects on their fisheries, for reasons other than commercial harvesting. During 2006 there was a novel outbreak of the disease, abalone viral ganglioneuritis (AVG), afflicting abalone in western Victoria, Australia, (Corbeil et al., 2010; Gilmour, Dwyer, & Day, 2011), decimating abalone stocks throughout most of this region of the State over a four-year period. This is another reason to determine the response of the system to a mortality "shock", with particular emphasis placed on the completeness of the recovery as measured against the performance criteria.

The first method for determining commercial catch for the upcoming season considered in this paper forms the basis for a revised Victoria abalone fishery management plan. This steers away from reliance on conventional fishery measures such as CPUE as reported by industry, which has been widely criticised as being hyperstable to stock depletion in abalone fisheries (Hart & Gorfine, 1997; Prince & Delproo, 1993). This hyperstability arises from the propensity of abalone to maintain an aggregated spatial distribution despite reductions in overall stock abundance.

Instead, a hierarchical decision-tree (Prince et al., 2008) is employed in a subjective risk assessment to determine the planned catch and minimum size limits for the ensuing fishing season. The aggregate of planned catches for individual abalone populations becomes the legislated TACC for the fishery.
The foundation for the second method, is the use of a shell length threshold for the commercial catch, based on the approach by Mayfield, (2010). Sampling the length structure of the commercial catch, either aboard divers’ boats or in processing factories, is more efficient and less costly than fishery independent sampling. This method is novel because it employs the mean shell length in the commercial catch as a performance measure. In this instance a low mean shell length in the simulated population results in a reduction of the planned harvest and an increase in mean shell length can lead to an increase in planned harvest. Similar work to this study has been conducted in the Norwegian spring-spawning herring fishery by Myrseth, Enberg, Heino, & Fiksen, (2011) where the existing two-stage management strategy was compared with a proportional harvest threshold strategy. Prior to this, Enberg, (2005) focused on the benefits of threshold strategies in the same fishery.

Detailed information about the simulation model formulation is presented in Section 2.2. The description of the management strategy for growth sensitivity within the model and an explanation of model simulations all appear in Section 2.3. Section 2.4 shows the variation in performance between the harvest management strategies. Two sets of results are produced for a thirty year period. One result set is conducted under unchanging environmental conditions, whilst the other set measures recovery from a dramatic stock loss. A discussion of the results appears in Section 2.5.
2.2 Model formulation

2.2.1. Length-based population model

The simulation modelling of alternative harvest strategies performed in this study uses some key abalone population dynamics adapted from the length-based stock assessment model developed by Breen et al., (2003) and research data from Victorian blacklip abalone populations (see Appendix). The purpose of the model is to compare the results between two strategies that determine the harvest for the upcoming season within a hypothetical abalone fishery. Length class intervals are 2mm, with length classes ranging from 30mm to 210mm. In addition to wide variation among the lengths of individual fully-grown abalone, there is substantial variation among individual growth rates (Day & Fleming, 1992). This secondary level of variation stems from different food supply levels, as well as other environmental factors. Therefore a stochastic version of the von Bertalanffy (VBL) growth equation (Fabens, 1965) is used to handle variation in growth (Troynikov & Gorfine, 1998).

Recruitment into the length-based abalone model occurs at two years of age since information about abalone biomass below this age is limited, largely because small abalone tend to prefer cryptic habitat. Upon recruitment, individuals are assigned individual VBL growth parameters from the following Gaussian bivariate distribution (Zhang, Lessard, & Campbell, 2009):
\[ L_\infty \sim N(\mu_{L_\infty}, \sigma_{L_\infty}) \quad k \sim N(\mu_k, \sigma_k), \quad \rho = 0.75 \]

where \( L_\infty \) is the fully grown length of an individual animal, \( k \) is the growth rate of an individual animal and \( \rho \) is the correlation coefficient between \( L_\infty \) and \( k \). The number of recruits for each time step is randomly distributed into a \( (L_\infty \times k) \) growth-class array containing \((28 \times 7)\) elements. The \( L_\infty \) intervals are \( 0.25\sigma_{L_\infty} \) in length and the growth rate intervals are \( \sigma_k \) in size. Each element of this population array, represented as \( r_{v,w,x} \), contain animals whose \( E(L_\infty) \) is within the range of the \( vth \ L_\infty \) class, \( E(k) \) within the range of the \( wth \ k \) class and time of recruitment into the fishery occurring in the \( xth \) time interval.

The length of animals at time \( t \) can be calculated as:

\[
l_{v,w,x}(t) = L_{\infty,\overline{v}}[1 - \exp\left(-k_{\overline{v}}(t - (t_x - 2))\right)] \tag{2.1}
\]

where \( L_{\infty,\overline{v}} \) is the midpoint of the \( vth \ L_\infty \) class and \( k_{\overline{v}} \) is the midpoint of the of the \( wth \ k \) class. The number of animals in each cohort is given by the equation:

\[
r_{v,w,x}(t + 1) = \left(1 - (shf_t)\right) r_{v,w,x}(t)e^{(-M/p)} \tag{2.2}
\]

where \( s = s_t(l) \) represents the fishing selectivity proportions at length at time \( t \).

\( h = h(l) \) is the time invariant probability of emergence from cryptic habitat at length \( l \),

\( f_t \) represents the fishing mortality rate applied to exploitable biomass during period \((t, t + 1)\), \( M \) is the annual instantaneous mortality rate and \( p \) is the annual periodicity of the model.
The annual instantaneous rate of mortality is modeled with time and length invariance setting $M = 0.2$. This value was taken from a field study by Beinssen & Powell, (1979) and is generally accepted as an estimate of $M$ for abalone (Shepherd & Breen, 1992). The Beverton-Holt recruitment function has been previously used by Zhao, Hirayama, & Yamada, (1991) to model abalone. This function is commonly used in length–based models where there is strong density dependent mortality among juveniles when the spawning stock is large (Haddon, 2001), as occurs with abalone (McShane, 1991). The recruitment productivity of this function is governed by a steepness coefficient $h$. Invertebrates such as abalone are generally considered to have high steepness in their recruitment owing to their high levels of fecundity (McShane, 1995).

### 2.2.2. Reference points and levels

The model adopts the fishery SSB and harvest reference points specified in the Australian Commonwealth Harvest Strategy Policy applied by Wayte & Klaer, (2010). The populations in this study begin each simulation at the limit reference point of $SSB_{20}$ (20% of pre-fished SSB), below which, harvesting should cease. The proportion of $SSB_{20}$ that is harvestable depends on the legal minimum length (LML). It is important to note that management does not have the tools to provide an accurate measurement of SSB and furthermore, during a high mortality event, management is unaware of the scale and length of the event. The $SSB_{20}$ threshold was chosen as a starting point because the
focus is solely on the recovery behaviour of the population below this point. A fishing mortality rate of $F_{20}$ is the initial rate applied to simulated populations because on average, it maintains an equilibrium level of $SSB_{20}$. An important assumption of the model is that the harvestable biomass of the simulated populations can be fully removed by divers.

### 2.2.3. Growth Sensitivity

The *medium* growth rate (see Table 2.1), is the estimated growth rate for blacklip abalone in this study. At five years of age, blacklip abalone in Victoria are expected to have reached sexual maturity and be emergent from cryptic habitat (McShane, 1995). The average length at this age is 91.4mm, with mean population VBL parameters, $E(L_{\infty}) = 154mm$ and $E(k) = 0.18$ for fully grown abalone. A LML setting of 110mm was chosen for the simulations, based on the originally legislated LML for harvesting blacklip from populations along the central Victorian coast (VAFMP, 2002). This setting provides five-year-old abalone with an average 24 months protection before attaining the legal minimum size. This period is critical to the sustainability of the fishery because it is important to ensure that young adult abalone can contribute to breeding before they are exposed to fishing mortality.

*Low* and *high* growth rates represent the extremities of the range of growth considered in this study. The rationale being that growth exhibits high spatial heterogeneity at reef scales (Troynikov & Gorfine, 1998), to the extent that averages do
not adequately characterize growth at the scale of management. Consequently, we perturbed the estimate of growth (by 5%) to investigate its effect on model outputs.

Table 2.1: Variable SSB values for each growth rate (represented in tonnes) for a low stock-recruitment \((h = 0.6)\) population at the beginning of the management simulation period. Expected individual abalone length (represented in mm) has a 10% coefficient of variance. Expected lengths of maturity and emergence from cryptic habitat are represented in mm.

<table>
<thead>
<tr>
<th>Growth rate</th>
<th>(E(L_{\infty})) ((95% \ PI))</th>
<th>E(maturity)</th>
<th>E(emergence)</th>
<th>(SSB_{20})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Medium</td>
<td>154.0 (123.82, 184.18)</td>
<td>91.4</td>
<td>90.0</td>
<td>26.71</td>
</tr>
<tr>
<td>Low</td>
<td>146.3 (117.63, 174.97)</td>
<td>86.8</td>
<td>85.5</td>
<td>22.77</td>
</tr>
<tr>
<td>High</td>
<td>161.7 (130.01, 193.39)</td>
<td>96.0</td>
<td>94.5</td>
<td>30.75</td>
</tr>
</tbody>
</table>

Table 2.1 shows the different levels of SSB at the beginning of the management simulation period for medium, low and high growth. Exploitation at the same LML setting of 110mm, combined with the difference in average size, hence weight for the three populations, explains the different compositions of SSB attributable to growth.
2.3 Methods

Simulations are used to compare the performance of two management strategies. The primary purpose of both strategies is to determine the intended fishing harvest for the following fishing season. The first strategy, termed harvest threshold mechanism (HTM), is based on annual catch relative to pre-specified target and limit values used as reference points linked to catch control rules. The second strategy, termed mean length of commercial catch (MLCC), features mean shell length relative to a target parameter.

2.3.1. Harvest threshold mechanism (HTM)

To determine a recommended catch for the upcoming fishing season, the target and limit reference points and catch control rules were as follows:

i) the optimal harvest target (OHT) represents the intended annual catch for a region;

ii) the annual upper harvest limit (UHL) and the annual lower harvest limit (LHL) are set to an equidistant level (30% in this study), either side of OHT;

iii) if completed annual catch falls below LHL, OHT is reset to the completed annual catch amount, whilst UHL and LHL are reset accordingly;

iv) if the UHL is reached during a fishing year, fishing ceases for the remainder of the season and OHT, UHL and LHL are raised by 5% for the following season if all other indicators within the region are positive (Figure 2.1).
v) it is assumed that during recovery from a high mortality event, stakeholders attempt to harvest at the UHL level after annual harvest falls below $F_{20}$.

Figure 2.2: Flowchart representing catch control rules implemented for the HTM strategy.
2.3.2. Mean length of commercial catch (MLCC).

With the advent of equipment used for electronically measuring and logging abalone shell size during the handling of the catch whilst at sea, it is now feasible to use the mean shell length in the commercial catch as a measure of fishery performance as well as stock selectivity (Mayfield, 2010). The sensitivity of the mean length of a commercially exploitable abalone population with the computation of deterministic equilibrium values is illustrated in Figure 2.2.

![Figure 2.3: The mean length of an exploitable abalone stock at deterministic equilibrium levels. The population parameter settings used are LML = 110mm, $E(L_\infty) = 154.0$mm and $h = 0.6$.](image)
The increased sensitivity in the mean length of a low exploitable stock level (Figure 2.2) suggests the MLCC strategy will detect changes in stock abundance under these circumstances. Figure 2.3 illustrates the impact that fishing selectivity and emergence from cryptic habitat has on numbers at length in the 110mm-124mm range, hindering the flow of growth from the sub-exploitable population to the exploitable population. At pre-fished equilibrium, only $\approx 26\%$ of exploitable abalone lie in this length range. However, a population containing 10$\%$ of unfished exploitable abalone comprises $\approx 74\%$ within the same range. The lack of replacement growth following over-harvesting of the smaller population has a substantial downward effect on its mean length.

Figure 2.4: Distribution of exploitable abalone numbers at length. The larger population is the pre-fished equilibrium population, whilst the smaller equilibrium population represents the removal of 90$\%$ of the pre-fished equilibrium population. The population parameters used were \( L_{\text{ML}} = 110\text{mm} \), \( E(L_{\infty}) = 154.0\text{mm} \) and \( h = 0.6 \).
The MLCC harvest strategy relies on two parameters \((q \text{ and } r)\), where \(q\) is the target mean length of commercial catch for the population and \(r\) is the maximum applicable ratio of change in annual harvest. Under the MLCC approach, the equation for determining the planned annual harvest for the following season based on the actual annual harvest from the previous season is:

\[
H_{t+1} = \begin{cases} 
H_t \times (1 + (r \times \text{tanh}(MCC_t - q))), & \text{if } MCC_t < q \\
H_{ref}, & \text{otherwise}
\end{cases}
\]  

(2.3)

where \(H_t\) is the annual harvest from the previous season, \(MCC_t\) is the current mean length of commercial catch, \(H_{ref} = H_{20}\) and \(0 < r < 1\).

This method controls the composition of SSB in the model by seeking to ensure the mean length of commercial catch remains near to \(q\). Since the range of the tanh function is \((-1, 1)\), the range of potential seasonal change in harvest is \((-r, \infty)\). When \(MCC_t\) is less than one millimetre below \(q\), the range of potential annual change in harvest imposed by management is \((-0.762r, 0)\). The parameter values for mean length of commercial catch combined with an LML setting of 110mm provide a two-dimensional layer of protection to SSB.

The mean length of the harvestable population fluctuates about \(q\). When the mean commercial catch length falls below \(q\), the previous annual harvest is reduced by the proportion of change ratio \(r\), as determined by Equation 2.3. This protection given to the SSB gradually serves to raise the mean length of harvest above \(q\). Therafter, the annual
harvest is restored to a reference level until the mean commercial catch length falls below $q$ again.

### 2.3.3. Simulation

All simulations were run with a time-step of one month. The first set of simulations were conducted on populations of one million abalone in unchanging environmental conditions. To achieve this, initial harvest and SSB ($F_{20}$ and $SSB_{20}$) were found computationally for the following parameter settings:

- Beverton-Holt recruitment steepness curve ($h$) = 0.6 and 0.8.
- for each steepness curve parameter value, growth was simulated for low, medium and high growth populations.

One thousand simulations were performed on the HTM and MLCC strategies for the six parameter combinations by applying the derived $F_{20}$ and $SSB_{20}$ levels over a thirty year period. As expected, all simulation populations and annual yield levels remained stable, near their $SSB_{20}$ and $F_{20}$ levels.

Following the System Dynamics methodology discussed earlier, the simulation process outlined in the previous paragraph was then repeated with the only difference being that a shock is applied to the system. This shock was in the form of a high mortality event in the first year of simulation, where $M = 0.6$. The thirty year simulation period measures the recovery performance of the HTM and MLCC harvest rules. For the MLCC strategy, the target average length of commercial catch $q$ was determined for
each combination of growth and recruitment levels at $SSB_{20}$, with $q$ ranging between (112.5, 126.5)mm, whilst the maximum annual reduction in harvest target was limited to 10%, with $r = 0.1$. 
2.4. Results

2.4.1 Harvest threshold mechanism (HTM)

The initial simulations of recovery from a twelve month high mortality event \( (M = 0.6) \) were based on populations with low recruitment steepness \( (h=0.6) \). For a medium growth population, SSB and annual yield did not recover and continued to decline (Figure 2.4A). The initial abundance of harvestable biomass enabled harvesting to remain at \( F_{20} \) for six years following the onset of the high mortality event, at which time SSB had reduced by more than 50%. Thereafter, the strategy control rules merely followed annual harvest on its downward trend, reducing OT and resetting the LHL and UHL whenever annual harvest fell below the LHL (Figure 2.1). Three decades into the recovery phase, SSB had fallen below \( SSB_4 \), with the population on the verge of collapse. The recovery results for populations with low recruitment steepness with both a high and low growth (not shown), confirmed that populations with poor recruitment have a low chance of recovery following a considerable loss of SSB under this strategy.

In contrast, Figure 2.4B presents a medium growth population with a high stock-recruitment relationship \( (h=0.8) \) with stabilised levels of SSB and annual harvest at \( SSB_{12} \) and \( F_{12} \). In this scenario there was only a three year period before harvest reduced below \( F_{20} \). The synchronous relationship between harvest and SSB in Figure 2.4B shows that the HTM strategy again does not facilitate harvest recovery. From the fourth year onwards, annual yield was derived mainly from continuous growth of sub-
harvestable biomass into harvestable biomass, due to the population containing a very low level of harvestable biomass.

Figure 2.5: The HTM strategy results for a medium growth population with A) low and B) high stock-recruitment. SSB and annual yield recovery during a thirty year recovery period from a high mortality event (M = 0.6 during Year 1), are compared with
the initial levels of $SSB_{20}$ and annual yield $F_{20}$ that would persist under unchanging environmental conditions.

Intuitively, it may be expected that a high growth population with strong stock-recruitment would recover better than the medium growth population under the HTM strategy but this was not the case (Figure 2.5). Despite a healthy recruitment level, SSB plummeted below $SSB_{7}$. The reason for the failed recovery of this population stems from the LML (110mm) being set too close to the expected size at which abalone mature.

![Graph: High growth - High recruitment]

Figure 2.6: HTM strategy results for a high growth population with strong stock-recruitment. SSB and annual yield recovery during a thirty year recovery period from a high mortality event ($M = 0.6$ during Year 1), are compared with the initial levels of $SSB_{20}$ and annual yield $F_{20}$ that would persist under unchanging environmental conditions.
A low growth population with high stock-recruitment scenario (Figure 2.6) provided the most suitable conditions for recovery from a high mortality event. Two key factors of this recovery were low growth, which meant that the population was effectively protected by the LML and strong recruitment. Another important recovery factor was the immediate sensitivity of harvest to the high mortality event because under this scenario the population contains a very low level of harvestable biomass at $B_{20}$.

Figure 2.7: HTM strategy results for a low growth population with strong stock-recruitment. SSB and annual yield recovery during a thirty year recovery period from a high mortality event ($M = 0.6$ during Year 1), are compared with the initial levels of $SSB_{20}$ and annual yield $F_{20}$ that would persist under unchanging environmental conditions.
2.4.2 Mean length of commercial catch strategy (MLCC)

The result in Figure 2.7 compares the best harvest recovery result achieved under the HTM strategy with that of the MLCC harvest strategy. The HTM strategy provided the most desirable and consistent recovery path compared to the erratic recovery of the MLCC strategy. The fact that the target length of mean commercial catch (112.5mm) was very close to the LML size meant that this strategy did not have the required sensitivity to facilitate a smoother recovery.

![Low growth - High recruitment](image)

Figure 2.8: Yield recoveries for a low growth population with strong stock-recruitment under the HTM and MLCC strategies.

The MLCC harvest recoveries for the remaining simulation scenarios were all similar in nature and there was only marginal difference in the speed of recovery and loss of yield. Of these scenarios, a population with strong stock-recruitment and high growth experienced the greatest loss of yield and longest length of recovery during the first
decade (Figure 2.8). This was followed by a shorter period of yield loss towards the end of the second decade, where recovery was almost complete. The most critical stage of recovery was the first decade where yield decreased substantially in comparison to the HTM strategy.

![Diagram: High growth - High recruitment](image)

Figure 2.9: Yield recoveries for a high growth population with strong stock-recruitment under the HTM and MLCC strategies.

The relationship between harvest and SSB under the MLCC strategy demonstrated a lag in harvest level response to change in SSB (Figure 2.9). After the year of high mortality, the reduced abundance level of SSB was too low to maintain a fishing mortality rate of $F_{20}$. As harvest was reduced below $F_{20}$ a correction to SSB followed. The MLCC strategy experiences a time lag of less than one year in response change in the mean length of commercial catch because its harvest control rule is implemented annually. The harvest recovery under this strategy generated a series of corrections to
harvest that diminished in size three decades after the high mortality event. The high stock-recruitment simulation populations experienced the largest magnitudes of correction.

Figure 2.10: MLCC strategy results for a high growth population with strong stock-recruitment. SSB and annual yield recovery during a thirty year recovery period from a high mortality event ($M = 0.6$ during Year 1), are compared with the initial levels of $SSB_{20}$ and annual yield $F_{20}$ that would persist under unchanging environmental conditions.
2.5 Discussion

The primary goal of this study was to compare the HTM harvest strategy with the MLCC harvest management strategy. Both strategies were able to maintain simulation populations in unchanging environmental conditions at their reference levels of $SSB_{20}$ and annual harvest $F_{20}$. The behaviour of both strategies was then measured in response to environmental change in the form of a high mortality event. Strategy evaluation was based on SSB and yield levels over the ensuing thirty year recovery period, with consideration given to the consistency of yield.

A key harvest rule of the HTM strategy reduces the target level of yield, based on yield obtained from the previous season that falls below a LHL threshold (70% of existing annual harvest target was used in this study). This rule was ineffective in regard to recovering SSB stock under the conditions of the simulations containing a high mortality event because harvestable biomass was depleted upon harvest reaching the LHL threshold. Under most recovery scenarios, harvest needed to be reduced well below the yield obtained from the previous season upon reaching the LHL threshold. The only simulation scenario where the UHL threshold was reached was for the population recovery in Figure 2.6. It was reached after nine years, at which time the target and threshold values were increased by 5%. For the remainder of the recovery period, annual harvest grew by 11%, therefore the threshold values were increased a further two times. The SSB did not benefit from the HTM strategy in this or any other simulation scenario.
It is important to note that under computer simulation, the HTM strategy harvest control rules could only be simulated in automatic mode. In practice, the strategy has a much more elaborate set of risk management rules that may have been applied well before the completion of thirty years. However, judging by these results, a considerable amount of damage is done before the strategy detects a substantial shift in SSB. Recovery under this strategy is difficult to achieve since management are unaware of levels of SSB and are purely reliant on commercial catch and LML to manage the fishery. Abalone populations that carry any level of harvestable biomass are particularly vulnerable to this strategy.

The MLCC strategy provides a two-tiered level of protection to SSB, using a LML (110mm in the case of this study) in conjunction with mean length of harvest specifications (Section 3.2). As a consequence, the MLCC strategy is more likely to provide management with more control over the level of SSB, than the one-dimensional LML settings under the HTM regime. The MLCC strategy is based on the use of a key statistical input as a proxy for SSB. The monitoring of this input enables estimation of next seasons’ harvest with the intention of maintaining SSB at a reference level.

The rationale behind the MLCC strategy is to reduce fishing pressure when the mean length of commercial harvest becomes lower than its reference target, giving rise to long-term harvesting gains from these savings. Following substantial population loss, the MLCC method continually reduces annual yield until the mean annual length of
commercial catch eventually exceeds the reference target. Once this benchmark has been reached, annual harvest reverts back to the original reference harvest level. The results showed that in most recovery scenarios, a correction occurs upon resumption of the original harvest level necessitating further reductions in harvest. However, each subsequent correction is much smaller than its predecessor.

Incomes from abalone fishing may fluctuate markedly in accordance with market conditions (Cook & Gordon, 2010), even under constant TACC. This is beyond the scope of the present study; nonetheless achieving consistency in yield and supply is important in maintaining market access and share. The scenarios modeled under the HTM strategy show, however, that stability in yield that would be expected to provide consistent supply were accompanied by overall long-term depletion in SSB. Although the MLCC resulted in inconsistent yields, these were compensated by recovery in SSB.

For both management strategies under consideration in this paper, we tested the ability to provide strong yield and viable populations following disturbances or mortality shocks to the population. The MLCC strategy provided the best recovery path to a population in recovery phase, as far as these criteria are concerned. The MLCC strategy is designed to fully recover losses of yield and SSB with the aid of electronically logging the lengths of some or all of annual commercial catch. In the Victorian fishery, the commercial catch during 2012 will equate approximately to three million abalone. Not only can logging data be used to support a strategy that involves an approach such
as MLCC, a substantial amount of information can be derived from their shell measurements, giving management a much clearer picture of the composition of abalone populations within their jurisdiction.
3. Implications of the dichotomy between the spatial scales at which abalone are managed and harvested

3.1 Introduction

Due to time and financial constraints, studies of key abalone population characteristics, such as abundance (Officer, Haddon, & Gorfine, 2001), mortality (McShane & Naylor, 1997), recruitment (Rogers-Bennett, Allen, & Davis, 2004) and growth (Preece & Mladenov, 1999) are usually conducted at spatial scales no larger than a reef or less than 10km of coastline. This is also the scale at which divers make decisions as to where to harvest (Gilmour et al., 2011; Leaf, Rogers-Bennett, & Haaker, 2007; Saunders, Mayfield, & Hogg, 2008; Stuart-Smith et al., 2008; Worthington et al., 1995).

By contrast, since the inception of commercial fishing of abalone in the 20th Century, single management zones have normally spanned hundreds of kilometres of coastline. This zone-level scale is used currently by management agencies in the Australian abalone fisheries to enforce TACC and LML limits. However, over the past decade commercial catches have been assessed at a finer sub-zonal scale (< 40 km) in some states (Prince et al., 2008). The following questions thus arise: How does the dichotomy of scale between management and commercial divers affect standing biomass and harvest? Should the fishery be managed at a scale that corresponds with the divers’ spatial preferences?

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To gain insight into these issues of spatial scale, a simplified system of a single management sub-zone (Section 3.2) comprising two spatially identical reefs, with each reef containing half of the sub-zone population distribution was considered. Management of the sub-zone is simulated at two scales; firstly at the existing sub-zone level and secondly, at an even finer reef level, where one reef is preferred for harvesting and when it becomes low on stock, harvesting commences in the non-preferred reef whilst the preferred reef recovers its stock. The results in Section 3.3 focus on the long-term effects of management scale and diver preference behaviour on population and yield. The implications for future management practice are discussed in Section 3.4.
3.2 Methods

3.2.1 Management system

Abalone management zones often span extensive lengths of coastline. A zone normally contains many regional sub-zones, each of which can be further separated at reef level. Large-scale zone management does not discriminate among reef populations, nor does it discriminate among larger regional populations contained within a sub-zone. Critically, this scale of management is insensitive to the contribution to annual commercial harvest made by each sub-zone or reef.

The management system simulated in this study represents a sub-zone abalone fishery. This mirrors recent proposed changes to management practice, which attempts to address the heterogeneous distribution of fishing effort at a zone level. This study was designed to measure the effectiveness of fine scale management at the sub-zone level, where the annual harvest target has no limits in place at the reef level. To achieve this, the management area was separated into two identical reefs (A and B). Reef A was designated as the preferred harvesting site each fishing period, giving rise to potential non-homogeneity of spatial fishing effort over the management area.

To measure the effect of preferential harvesting within the system, it was necessary to obtain baseline results by simulating the management sub-zone with fine-scale spatial distribution of fishing effort. This outcome could occur without management intervention in the case where there is no harvest preference for either reef. However, to
be certain, management can enforce increased uniformity of spatial fishing effort by managing a fishery at a finer scale (reef level). This allows for harvest targets to be set for each reef, thereby eliminating the possibility that the annual harvest target for the managed sub-zone is removed from one reef only. In this system, both reefs were allotted a 50% share of the sub-zone annual harvest target to achieve the baseline results.

3.2.2 Fine scale management at reef level

Prior to simulation of the system, a post-exploitation equilibrium state population was required at the commencement of the simulation period. This was achieved by applying a 50-year harvesting curve to a pre-exploitation abalone population and refining the curve by trial and error until population equilibrium was achieved. The equilibrium sub-zone population biomass of 59 t (29.5 t per reef) was able to sustain annually 12 t (6 t per reef) of harvest. The number of abalone contained within the 8 t that were legally harvestable at any point in time represented approximately 6% of the corresponding pre-exploitation number. There were actually 13 t of abalone that could be legally harvested residing within the sub-zone at this time, however, 5 t of these are protected from short-term capture due to a combination of being beyond divers’ visual range, growth beyond LML due to harvest cycle, and selectivity at or just above the LML.

As expected, under fine-scale management of harvesting effort, diver behaviour was neutralised, since neither reef deviated from equilibrium over a 40-year period. This
equilibrium baseline result made for easy interpretation of the spatial effects arising from preferential harvesting of a sub-zone. The effects were simply measured by any deviation from any corresponding equilibrium points throughout simulation.

3.2.3 Diver behaviour at sub-zone level

The following harvesting algorithm was applied to the management area to model the spatial effects caused by the preferential fishing effort of divers within the system. Firstly, always harvest the preferred reef A and leave non-preferred reef B unfished. When reef A is fished to a point where it is below a threshold level of productivity, fishing effort is switched to reef B. This reef is then fished until reef A has recovered beyond its recovery threshold. In the case where both reefs are still recovering, no fishing occurs.

A major difficulty in modelling an abalone fishery was establishing when the increasing effort required to obtain a catch becomes a contributing factor that leads to fishermen going elsewhere to catch their quota. The tendency of the majority of abalone to form aggregations means that they are easy to locate until the overall population becomes heavily depleted. McShane, (1996) observed that only abundant aggregations are commercially fished and these account for a small proportion of all aggregations in an unfished population and an even smaller proportion in a fished population.

Given this aggregative spatial behaviour of abalone, the assumption that the number of aggregations on a reef decreases at a similar rate to the average number of abalone
within an aggregation was used. Aggregation density was therefore based on the proportion of the current number of abalone that could be legally harvested ($P_{curr}$) against the corresponding pre-exploitation number ($P_0$), and is measured as $\sqrt{P_{curr}/P_0}$, where $P_{curr}/P_0$ is a relative measure of harvestable biomass. Therefore, the Preference Index is an inversely proportional function of aggregation density $(P_{curr}/P_0)^{-0.5}$ and it measures of the level of preference for a fishing area in relation to standing harvestable biomass, at the point when divers decide the area is not worth fishing any more.

Figure 3.10 illustrates the persistence of aggregations that occurs during population decline. However, as harvestable numbers deplete to low levels, the level of site preference required to continue fishing at the same site, begins to rise rapidly. The model captured the behaviour of divers near this point, by predicting when they will switch fishing effort away from a region. Three thresholds levels, indicating the preference that a fishing area has over alternative sites were chosen to trigger to a harvesting switch, when the relative harvestable biomass (calculated by the model) reached 5%, 4% and 3%. The relative biomass at the point of switching for the first time was stored as a tipping point for the remainder of the simulation period, invoking temporary cessation of fishing whenever it was reached.
Figure 3.11: Fishing site preference as a function of the number of abalone that could be legally harvested in proportion to the corresponding pre-exploitation number. The preference index is measured as \( (P_{\text{curr}}/P_0)^{0.5} \), where \( P_{\text{curr}}/P_0 \) is a relative measure of exploitable abalone. The function reflects that preference for a harvesting site is increasingly sensitive to reducing numbers of exploitable abalone.

Switching harvest effort away from an area that is only marginally preferred to neighbouring areas when the relative harvestable biomass reaches 5%, could arise from multiple factors. For instance, there may be an availability of neighbouring sites perceived as more profitable or equally accessible at this point of relatively high harvestable biomass. However, if these factors are not as favourable, divers will be disinclined to switch harvest effort, instead harvesting the same area some or all of the way until very low levels of relative harvestable abalone, nearing the point where 3% remain.
Modelling the point at which the fishery regains its perceived harvesting viability was also problematic. The point at which the relative harvestable biomass exceeded 150% of the tipping point was arbitrarily chosen to be when the fishery would be reopened. This means that for a moderately preferred site where $P_{curr}/P_0 = 6\%$, that divers would abandon harvesting when $P_{curr}/P_0 = 4\%$. After a recovery period, harvesting resumes when exploitable stock reaches its original level, $P_{curr}/P_0 = 6\%$. Since harvest resumption at a marginally preferred site (7.5%) occurs above the original 6% level and resumption at a strongly preferred site occurs at 4.5%, there is enough sensitivity in this experiment to illustrate the effects of heterogenous spatial harvest effort.

### 3.2.4 Sub-zone population model

In recent years there have been several models developed to capture the spatial dynamics of fish populations. In Chile, Gelcich, Edwards-Jones, & Kaiser, (2007) measured the willingness of shellfish harvesters to leave heavily fished areas unharvested for future gains. More specific to abalone species, in South Australia the whole abalone fishery was divided into selected fishing areas to perform spatial modelling of several effort estimation methods (Burch, Mayfield, Stobart, Chick, & McGarvey, 2011) and several commercial catch sampling methods (Burch, Mayfield, & Chick, 2010). A spatial model was developed to capture illegal catch effects in the South African abalone fishery (E. Plaganyi et al., 2011; E. E. Plaganyi & Butterworth, 2010).
Because abalone have a very limited spatial movement range during their lifetime (Officer, Dixon, et al., 2001) and limited larval dispersal (Prince, Sellers, Ford, & Talbot, 1987), it was reasonable to model each spatial population independently. Each individual recruit is assigned individual VBL growth parameters from the following Gaussian bivariate distribution:

\[ L_\infty \sim N(154.0 mm, 15.4 mm^2), k \sim N(0.18, 0.02^2), \rho = 0.75 \]

This rate of growth is represented in Chapter 2 as medium growth (Table 2.1) where the expected abalone length at full sexual maturity and emergence from cryptic habitat is assumed to be approximately 90 mm. Under medium growth an abalone has reached approximately 5 years of age at this length. The annual instantaneous rate of mortality was set with length invariance where \( M = 0.2 \) (Sanders & Beinssen, 1996). An LML setting of 114 mm was considered appropriate, because the expected age of these abalone at this length is greater than 7 years, implying that on average an abalone will have spent enough time to contribute to population sustainability as a mature spawner before potential capture.

The sub-zone population model consists of two instances of the length-based population model that was introduced in Chapter 2. System equilibrium is attained by both reef populations running in parallel with fine-scale spatial distribution of fishing effort (Section 3.2.2). Measurement of the spatial effects arising from preferential harvesting of a sub-zone only occurs when the diver population removes the periodic harvest allocation of 1t from one reef only, assuming there is 1t of exploitable biomass.
available at either reef. The subsequent system interaction that takes place between the two reefs is regulated by the following distribution of harvest effort:

Where $f^u(t)$ represents the fishing mortality rate applied to reef $u$ during period $(t, t + 1)$ and $b^E(t)$ represents the exploitable biomass during period $(t, t + 1)$,

$$f^A(t) = \begin{cases} 
0, & \text{if reef A is in recovery mode} \\
1/b^E(t), & \text{otherwise}
\end{cases}$$

$$f^B(t) = \begin{cases} 
0, & \text{if reef B is in recovery mode} \\
0, & \text{if reef A is not in recovery mode} \\
1/b^E(t), & \text{otherwise}
\end{cases}$$

Under fine scale management at reef level, $f^A(t) = f^B(t) = 0.5/b^E(t)$.

### 3.2.5 Simulations

All simulations were run with a time-step of one month. The simulation period was 2011–2050, with 1000 simulations performed for each scenario. Simulation under the fine-scale harvesting scenario produced equilibrium harvest and population biomass results, appearing in Section 3.2.2. Simulation of the population effects resulting from preferential harvesting is measured at three different levels of preference, (marginal, moderate and strong). The multi-population simulation model employed was developed in Visual Studio using C source code. The model is restricted by very slow runtimes, due to individual tracking of each recruit.
3.3 Results

 Preferential harvesting (2011–2050)

Harvest was barely affected when reef A was only marginally preferred to reef B, in those instances where switching of harvest effort occurred whilst the harvestable biomass was still strong (Table 3.2). Intuitively, this result was not surprising since switching harvest sites at this level of preference closely mimics fine-scale harvesting, with only one sixth of the equilibrium harvestable biomass being removed before switching occurred. Because harvestable biomass oscillated in both reefs between the lower switching threshold of 5% of harvestable biomass and the recovery threshold of 7.5% of harvestable biomass meant that the equilibrium harvestable biomass of 6% was close to the mid-point of the oscillatory range.

To achieve near-optimal fine-scale harvesting, 6 switches per annum from reef A over the 40 year period would be required, with the population remaining in equilibrium. Here, there was an average of 1.25 switches per annum, where a consistent switching pattern occurred, with each reef being harvested for almost five months whilst the other recovered. Noticeably, the cumulative harvest for reef B exceeded that of reef A by 2.8%, contrary to the aim of the divers.
Table 3.2: Results [tonnes (% change)] for management simulation period 2011–2050, where % change indicates difference from comparative equilibrium results of the same starting population fished at a reef management scale. These results reflect simulation of the preferential fishing distribution. The \( P_{\text{curr}}/P_0 =\) (%) value is the point when divers switch fishing effort away from reef A.

<table>
<thead>
<tr>
<th>Harvest switching strategy</th>
<th>Reef A (% change)</th>
<th>Reef B (% change)</th>
<th>Sub-zone (% change)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Marginal preference (( P_{\text{curr}}/P_0 = 5% ))</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cumulative Harvest</td>
<td>235.8 (-1.6)</td>
<td>242.5 (1.2)</td>
<td>478.3 (-0.2)</td>
</tr>
<tr>
<td>Population Biomass</td>
<td>28.0 (-4.7)</td>
<td>30.7 (4.6)</td>
<td>58.7 (-0.1)</td>
</tr>
<tr>
<td>Harvestable biomass</td>
<td>3.8 (0.0)</td>
<td>4.0 (4.6)</td>
<td>7.8 (2.3)</td>
</tr>
<tr>
<td>Moderate preference (( P_{\text{curr}}/P_0 = 4% ))</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cumulative Harvest</td>
<td>231.4 (-3.5)</td>
<td>245.7 (2.5)</td>
<td>477.0 (-0.5)</td>
</tr>
<tr>
<td>Population Biomass</td>
<td>26.9 (-8.6)</td>
<td>29.0 (-1.3)</td>
<td>55.8 (-4.9)</td>
</tr>
<tr>
<td>Harvestable biomass</td>
<td>3.0 (-20.9)</td>
<td>3.5 (-9.1)</td>
<td>6.5 (-15.0)</td>
</tr>
<tr>
<td>Strong preference (( P_{\text{curr}}/P_0 = 3% ))</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cumulative Harvest</td>
<td>226.8 (-5.4)</td>
<td>247.8 (3.4)</td>
<td>474.6 (-1.0)</td>
</tr>
<tr>
<td>Population Biomass</td>
<td>25.3 (-13.7)</td>
<td>26.7 (-8.9)</td>
<td>52.1 (-11.3)</td>
</tr>
<tr>
<td>Harvestable biomass</td>
<td>2.5 (-34.9)</td>
<td>2.6 (-32.5)</td>
<td>5.1 (-33.7)</td>
</tr>
</tbody>
</table>

Switching of harvest effort at the middle preference level indicates that reef A is moderately preferred over reef B. This result set also portrayed little effect on cumulative harvest for the sub-zone, but the difference in cumulative harvest for reef B
over reef A was further exacerbated from 2.8% to 6.2% (Table 3.2). Under this strategy, there was an average of 1.35 switches per annum. This increased switching frequency was due to a narrower band between the lower switching threshold of 4% and the recovery threshold of 6% of harvestable biomass.

Only when reef A was subjected to greater harvesting pressure due to its strong appeal, was a substantial reduction in harvest seen in the latter years of the simulation period (Figure 3.11A). By the year 2050, both reefs were producing 94% of their equilibrium yield, whilst yield continued to trend downwards. After 30 years of short recovery periods, both reefs fell out of harvesting synchronisation and were unable to sustain a timely recovery thereafter. The average number of switches per annum from reef A to reef B further increased to 1.45. This means that increased frequency of switching harvest effort is only beneficial when the harvestable biomass is close to its original post-exploitation equilibrium point. In this case, the increased frequency of switching harvest effort relative to the other two switching thresholds occurred because reef A was fished to a low level every four months, meaning that the divers had no choice but to move to reef B.
Figure 3.12: Mean annual harvest A), mean population biomass B) and mean harvestable biomass C) results when reef A was strongly preferred to reef B as a harvesting site. The equilibrium result arose from harvesting the population at a finer scale of management over both reefs. The remaining results stemmed from preferential harvesting effort of the sub-zone. Graphical results for switching strategies at medium and low levels of reef preference are not shown, as they were similar in behaviour to these results, with less fluctuation.

In keeping with the harvest result, population biomass and harvestable biomass showed little response when reef A was only marginally preferred to reef B (Table 3.2). However, reef B marginally benefitted from its non-preferred harvesting site status, as it held more standing biomass at the conclusion of the simulation period. Harvesting each reef where reef A was moderately preferred to reef B gave the first indication of biomass depletion caused by preferential distribution of fishing effort. The 15% loss in standing harvestable biomass resulted in a population biomass loss of 4.9% over forty years.
Whilst this did not appear catastrophic, the population was still trending downward due to it no longer being in equilibrium.

When reef A was continually fished heavily due to the strong preference given to it, a more substantial depletion of harvestable biomass occurred (Figure 3.11C). The harvestable biomass on reef A initially plummeted by 35% as a result of the first harvest cycle and never recovered. Over the first 30 years of the simulation period, the harvestable biomass for reef B gradually succumbed to the pressure placed on it by the poor yield from reef A. From this point, there was no population biomass recovery due to constantly high harvesting pressure. Figure 3.11B illustrates a gradual depletion of more than 10% of total population biomass under this scenario of harvesting reef A down to a low level of harvestable biomass.
3.4 Discussion

The aim of this study was to provide insight into the dichotomy between the spatial harvesting pattern of divers and fine-scale management at the sub-zone level, with particular focus on any long-term effects on standing biomass and harvest. This was achieved by measuring the effect of preferential distribution of fishing effort on an abalone management area separated into two spatially equivalent reefs, both containing similar population size and structure. It has been recently suggested by Holland & Herrera, (2012) and Wilson et al., (2013) that fine scale management will fail as a result of inefficient governance and that fine scale management under effective governance should still be pursued. However, as far as the management of blacklip abalone in Victoria is concerned, this was not found to be the case.

It is important to note that by dividing the sub-zone biomass equally into a preferred and non-preferred area within the management system the model serially depletes the sub-zone biomass at the maximum rate possible. Therefore, any unequal distribution of the sub-zone biomass within the two areas of harvesting preference will result in less serial depletion (Shepherd et al., 2001) over a 40-year period than shown in this study. In particular, if the preferred or non-preferred area contains only a small proportion of sub-zone biomass, even strong preferential harvesting patterns of divers will result in negligible long-term stock depletion. It is also important to mention that the modelling of two spatial harvesting preference levels simplifies abalone management zones, which are complex systems that may contain a sliding preferential harvest scale of multiple
spatial areas. These environmental conditions would only serve to dilute the level of serial depletion found, providing management with less incentive to manage at a finer scale.

Critical to the understanding of the preferential spatial harvesting effects on an abalone fishery is the preference index introduced in Figure 3.10. Some of the rationale behind why a preferred site experiencing more perceived fishing effort than alternative sites can still be considered a more attractive harvest option is outlined in the introduction. It was determined that when the harvestable abalone population falls below 5% of the pre-exploitation population at a preferred site, the rate of change in the preference index would cause divers to consider switching harvest effort to a less preferred site where catch rates are perceived to be higher. Only if alternative harvest areas are completely unappealing to divers, would they continue to harvest a preferred area until the relative harvestable population approached 3%. Here, the increasing catch effort reflects the increased diver time required to search for aggregations of abalone. Effort at this point is beginning to grow exponentially, necessitating a switch of harvest effort by divers elsewhere. Results from the case where the preference for a particular harvest site is strong suggest there might be some value in fine scale management.

If the level of preference for a preferred harvest site is deemed to be only moderate, divers will switch their effort from the preferred site earlier. The midpoint of the preference index range where the relative harvestable population at a preferred site is depleted to 4% was chosen to represent the scenario where a moderately preferred site
becomes less attractive to harvest than alternative sites. The rate of change in effort required at this level of harvestable abundance would be increasingly noticeable to the divers, providing enough impetus for divers to switch effort to less preferred sites. The results associated with harvesting a sub-zone containing a moderately preferred region provided management with little incentive to manage the fishery at a finer scale.

With reference to the finding in the previous paragraph, if neighbouring fishing sites hold only marginally less harvest appeal than the preferred site, finer-scale management serves no purpose since the difference in spatial scale between the associated diver-harvesting behaviour and harvest management at a sub-zone level is negligible. Furthermore, the same conclusion must hold for a sub-zone where all sites are equally preferred for harvesting, where divers would strongly consider switching harvest effort to neighbouring sites regularly, in the expectation of higher catch rates.

This study may contain relevance to other species (particularly abalone) with sedentary characteristics. Firstly, the preference index used in this study, reflecting the strong aggregative behaviour of blacklip abalone, provides scope for serial population depletion within a management zone. For species that are less densely aggregated, a preference index tending closer to linearity would be required to characterise such species. This would result in less stock depletion because harvest switching would occur more regularly than for blacklip abalone due to an increased change in effort rates. Secondly, the preferential behaviour of divers in this study is extreme, since the preferred site is always the highest priority for harvesting. If this extreme behaviour
does not hold for other species, then again, there will be less scope for stock depletion within a management area.
4. Bioeconomic modelling

4.1 Introduction

There are various motives that drive the behaviour of abalone divers within a fishery. Literature focusing on the length of working days by abalone divers in New Zealand found divers were likely to persevere on days where catch rates were poor because the daily time spent on fishing is primarily based on hours targets of the divers (Eggert & Kahui, 2013). By contrast Gorfine & Dixon, (2001) studied the spatial choices of abalone divers and discovered that divers who are driven by catch quotas were inclined to switch locations without hesitation if the catch rates did not meet their expectations. It was further noted, however, that such diver behaviour did not necessarily extend to the more remote fishing grounds where stocks were abundant.

The effects that divers can have on a fishery was borne out during an intense period of harvesting with high catch rates at a location containing blacklip abalone (Chick, Mayfield, Burch, Turich, & McGarvey, 2012). Measurements taken before and after the event showed that the habitat space left vacant during the harvest event was mainly reclaimed by members of the sub-legal population. This suggests that premature emergence of blacklip abalone from cryptic space leaves the species vulnerable to serial depletion. The analysis of the spatial movement of humans as resource predators by Ling & Milner-Gulland, (2008) is an example of similar work to this study. With the development of a bioeconomic model the authors consider how increased travel costs
that apply to harvesting in remote areas of a spatially continuous wildlife resource system can significantly impact on the system with a reduction in overall productivity.

The simulated behaviour of divers in Chapter 3 involves the migration of the entire diver population to another reef when triggered by a low biomass threshold at the reef being fished. In some commercial fisheries the instantaneous movement of total fishing effort from one location to another is more likely to be an involuntary response to the implementation of management policy such as seasonal closures of specific fishery areas (Demestre, de Juan, Sartor, & Ligas, 2008).

In this chapter a more rational account of diver behaviour is presented by focusing on the motivations behind divers moving to different fishing grounds. This requires more details pertaining to the spatial behaviour of divers with less emphasis placed on the population structure. The adopted management method is a simplified process whereby a constant equilibrium rate of effort is employed to control harvest. The system behaviour of divers is determined by the following assumptions:

- divers move to the most profitable location;
- there are delays relating to the time taken by divers to respond to changes in profit.

These assumptions have further implications for divers in a fishery where the harvesting cost differential between fishing grounds is substantial. Under this circumstance the search for high catch rates is tempered by the corresponding harvesting costs.
The questions to be addressed in this chapter are: How does diver migration behaviour affect the key variables relating to management objectives? Are some fishing locations more vulnerable to stock depletion than others? The latter question stems from the result given in Chapter 3 demonstrating that a strongly preferred fishing location was unable to sustain equilibrium harvest due to increased attention from divers.

The remainder of this chapter includes a detailed description and analysis of a bioeconomic fishery system. This is followed by the presentation and explanation of model results that measure system effects of diver migration behaviour following a system disturbance. Finally, a discussion of the results is provided in the context of the research questions.
4.2 Methods

In this study, divers move to different areas of a system in search of more profitable harvesting grounds. The simplest way to gain initial understanding of this system is by considering two harvesting areas of equal carrying capacity \((K_1 = K_2 = K/2)\). The bioeconomic system model presented by Clark, (2010) is used as a platform to develop a model that separates a fishery system into two sub-systems. Similar to Clark, (2010), the intended design of this system is not to replicate a real world fishery. Instead, a simplified model is developed to facilitate various measurements of dynamic fishery system behaviour. Two views of this system are possible (Figure 4.12). One view gives an aggregated system perspective (AGGP) while the other represents the sub-system perspective (SUBP) where the sub-systems identified as Location 1 and Location 2 provide finer system detail. Importantly the system behaviour of the model by Clark, (2010) can differ from that exhibited by the AGGP of this model.

4.2.1 Sub-system characteristics and assumptions

Biomass, harvest and growth under the AGGP can all be derived by summing corresponding biomass amounts of the two sub-systems. In addition, total effort and daily profit can also be ascertained by summing sub-system amounts. Without any loss of generality it is assumed that Location 1 is designated as the area with the lowest daily harvesting cost per unit of effort. A difference in daily harvesting costs between the two sub-systems can be partially explained by location-specific daily travel costs.
In this system harvest control is achieved by maintaining a constant rate of diver effort. In a limited access fishery there is a cap on total effort with restriction on the number of commercial harvesting licences. This system interprets a fishery licence as a unit of effort that is measured as a complete diving day by a single diver at a given location.

In the following sub-sections the evaluation of the sub-system model is performed with identification of equilibrium system biomass points and analysis of their stability, robustness and dynamic behaviour. Dynamic diver behaviour in response to system profits is then incorporated into the model.

Figure 4.13: System variables from the aggregated system view (Table 4.3) derived by summing corresponding sub-system amounts.
4.2.2 Sub-system bioeconomic dynamics.

All of the model variations that are employed in this study stem from the following system of two differential equations and two profit equations

\[
\dot{B}_1 = G(B_1) - h_1(t) \tag{4.1}
\]

\[
\dot{B}_2 = G(B_2) - h_2(t) \tag{4.2}
\]

\[
\pi_1 = ph_1(t) - c_1E_1(t) \tag{4.3}
\]

\[
\pi_2 = ph_2(t) - c_2E_2(t) \tag{4.4}
\]

where \( p \geq 0 \) is the price per unit of harvest and for each sub-system \( i \), \( c_i \) is the daily harvesting cost, \( G(B_i) \) is the growth function and \( B_i, G(B_i), h_i, E_i, c_i \geq 0 \).

The harvest in sub-system \( i \) is derived as

\[
h_i = E_i k f(B_i/K_i) \tag{4.5}
\]

where \( k \geq 0 \) is the catch per dive unit at maximum abundance and \( f(B_i/K_i) \geq 0 \) represents the fishing encounter density function for sub-system \( i \).

The revenue component of daily fishery profit (Equations 4.3 and 4.4) is the product of the price per harvest unit and the number of daily harvest units per sub-system whilst the daily expense is accounted for by the location-specific daily cost per unit of effort multiplied by the corresponding level of effort. Substitution of the \( h_i \) term in Equations 4.3 and 4.4 with the RHS of Equation 4.5 produces
\[ \pi_i = p E_i k f(B_i / K_i) - c_i E_i \]

\[ = E_i (p k f(B_i / K_i) - c_i) \]

resulting in the daily unit of effort profit equation

\[ T_i = p k f(B_i / K_i) - c_i \]  \hspace{1cm} (4.6)
Table 4.3: Key system parameters/variables of the with description and units

<table>
<thead>
<tr>
<th>Symbol</th>
<th>Description</th>
<th>Unit</th>
</tr>
</thead>
<tbody>
<tr>
<td>$B(t)$</td>
<td>Biomass size at time $t$</td>
<td>tonne</td>
</tr>
<tr>
<td>$H(t)$</td>
<td>Harvest rate at time $t$</td>
<td>tonne/day</td>
</tr>
<tr>
<td>$K$</td>
<td>Ecological carrying capacity</td>
<td>tonne</td>
</tr>
<tr>
<td>$G(B)$</td>
<td>Net natural growth in biomass at size $B$</td>
<td>tonne/day</td>
</tr>
<tr>
<td>$r$</td>
<td>Instantaneous rate of net natural growth</td>
<td>dimensionless/day</td>
</tr>
<tr>
<td>$E(t)$</td>
<td>Rate of diver unit effort at time $t$</td>
<td>du/day</td>
</tr>
<tr>
<td>$k$</td>
<td>Maximum daily harvest per unit of effort</td>
<td>tonne/du. day</td>
</tr>
<tr>
<td>$f(B/K)$</td>
<td>Fishing encounter density determining the</td>
<td>dimensionless</td>
</tr>
<tr>
<td></td>
<td>proportion of $k$ harvested at size $B$</td>
<td></td>
</tr>
<tr>
<td>$p$</td>
<td>Unit harvest price</td>
<td>$/tonne</td>
</tr>
<tr>
<td>$c$</td>
<td>Daily harvesting cost per unit of effort (diver day)</td>
<td>$/du. day</td>
</tr>
<tr>
<td>$\pi$</td>
<td>Daily system profit</td>
<td>$/day</td>
</tr>
<tr>
<td>$\Upsilon$</td>
<td>Daily profit per unit of effort</td>
<td>$/du. day</td>
</tr>
</tbody>
</table>
4.2.3 Equilibrium points of system.

At equilibrium, profits remain equal in the absence of any diver movement between locations. The daily profit equality between locations is represented as

\[ pkf(B_1/K_1) - c_1 = pkf(B_2/K_2) - c_1(1 + \Omega) \]

where \( \Omega \geq 0 \) is the proportionate difference in daily harvesting costs between locations. To determine the equilibrium points of the system we require

\[ f(B_1/K_1) = f(B_2/K_2) - \Delta \quad \text{(4.7)} \]

\[ \dot{B}_1 = \dot{B}_2 = 0 \quad \text{(4.8)} \]

The term \( \Delta = \frac{\Omega c_1}{pk} \) is the daily cost difference in harvesting costs with \( \Omega c_1 \) standardised by the attainable revenue per unit of effort achievable at maximum system abundance (pk). A further condition of an equilibrium system point requires the corresponding rates of net natural growth and harvest to be equal at both locations (Equation 4.8).

Equilibrium point representation of the system by Clark, (2010) is formed by the equation \( B_1 = B_2 \) with homogenous harvesting costs (\( \Delta = 0 \)). Alternatively any cost difference (\( \Delta > 0 \)) means that \( B_1 < B_2 \) because Location 1 contains the more profitable harvesting ground. A further consequence of \( \Delta > 0 \) is that each location will have a different biomass density threshold at which point harvesting becomes unprofitable. This unique biomass point known as bionomic equilibrium can be found by rearranging

\[ pkf(B_i/K_i) - c_i > 0 \]

meaning that fishing only occurs at Location 1 or 2 if
and the bionomic equilibrium biomass sizes are $B_{1, BE} = K_1 f^{-1} \left( \frac{c_1}{pk} \right)$ and $B_{2, BE} = K_2 f^{-1} \left( \frac{c_1 (1+\Omega)}{pk} \right)$.

The expressions for the bionomic equilibrium equations above where commercial harvesting is economically viable at both locations, is feasible if $pkf(B_2/K_2) > c_1 (1 + \Omega)$. Figure 4.13 highlights the contrast in the equilibrium biomass density structure of a fishery with homogenous harvesting costs opposed to a fishery with substantially different costs. At equilibrium simultaneous harvesting occurs in the corresponding ranges $B_{1, BE} < B_1 \leq K_1 f^{-1} (1 - \Delta)$ and $B_{2, BE} < B_2 \leq K_2$. The difference between the equilibrium biomass densities at both locations grows with increasing $\Delta$ and the difference is an increasing function of biomass.
Figure 4.14: Two sets of system equilibrium points are shown. Where harvesting costs are equal ($\Delta = 0$) the equation for the line of equilibrium points (dotted line) is $B_1 = B_2$. Alternatively for different harvesting costs at each location ($\Delta > 0$) the equation (solid line) is $B_1 = K_1 f^{-1} (f(B_2/K_2) - \Delta)$. At any equilibrium point both locations are equally profitable to divers providing that the harvest rates equate to the corresponding growth rates at each location. If $\frac{B_1}{K_1} > f^{-1}(1 - \Delta)$ then harvesting of $B_2$ does not take place (system is unfeasible). Harvesting does not occur in either location if biomass falls below the bionic equilibrium point \left( \frac{c_1 f^{-1}(\frac{c_1}{p_k})}{p_k}, \frac{c_2 f^{-1}(\frac{c_2(1+\Omega)}{p_k})}{p_k} \right).

4.2.4 Stability and robustness of equilibrium system points.

Alternative views of system growth are illustrated in Figure 4.14. The determination of the stability of equilibrium points from the single biomass system is well understood. There normally exists one unique system equilibrium point that produces maximum sustainable yield (MSY). For any specified fixed harvest amount $H^*$ that is less than MSY there are two system points that maintain system equilibrium at zero net growth. The equilibrium system points consisting of low and high biomass density are identified as $B_{td}^*$ and $B_{hd}^*$ respectively (Figure 4.14A). Any biomass size within the range $(B_{td}^*, B_{hd}^*)$ produces positive net system growth.

By contrast the SUBP of system growth occurs in three-dimensional space (Figure 4.14B). The first point to make about this system view is that MSY (central system point) can only be achieved if there are homogenous system costs ($\Delta = 0$). Where harvesting costs differ between the two locations the maximum achievable sustainable yield (MASY) of the system diminishes in size for increasing values of $\Delta$ reducing the possible range of choice for $H^*$. From the SUBP the two equilibrium biomass points in Figure
4.14B lie on a circle of system points that produce identical growth. This means that any choice of $H^*$ corresponds with a circle of system points that represent equilibrium system growth. Equilibrium is only attainable if the chosen circle intersects with at least one system equilibrium point.

![Diagram](image_url)

A) System growth

Domain of attraction of $B_{hd}^*$

Rate of biomass change (% MSY)

Biomass density (%)
Figure 4.15: Production model of net growth under A) aggregated system and B) sub-system perspectives. In both examples there are two equilibrium system points for fixed harvest amount \( H^* \). Point \( B_{hd}^* \) in (A) is a stable system point where the robustness of its stability is a function of the domain of attraction. In (B) \( H^* \) consists of fixed equilibrium harvest quantities \( h_{1,hd}^* \) and \( h_{2,hd}^* \) for biomass size \( B_{hd}^* \) and \( h_{1,ld}^* \) and \( h_{2,ld}^* \) for biomass size \( B_{ld}^* \). For this example a cost difference between Locations 1 and 2 means that different harvest rates occur (\( h_{1,hd}^* \neq h_{2,hd}^* \)). The stability and robustness measure at \( B_{hd}^* \) from the coupled sub-system perspective has more complexity than the aggregated system view.

From the SUBP system points \( B_{ld}^* \) and \( B_{hd}^* \) both comprise Location 1 and 2 biomass densities that yield the same daily profit per unit of effort and produce a combined net growth that equals \( H^* \). If \( \Delta > 0 \) the difference in biomass densities means that different harvest rates occur at both locations. Furthermore there is a loss of system symmetry as
the sub-system harvest rates also differ between equilibrium system points \( h_{s,td}^* \neq h_{s,td}^* \). While Location 1 is the most vulnerable area of the fishery with less biomass density the robustness of an equilibrium point is not just a matter of measuring its domain of attraction in three-dimensional space. The profit constraint also comes under consideration because any loss of biomass in Location 1 means that Location 2 becomes more attractive to harvesters due to higher profitability.

4.2.5 Dynamic behaviour at equilibrium points

Following deviation from equilibrium the divers respond to the changes in the distribution of profit. This will lead to changes in the harvest rate at both locations while maintaining the fixed rate of total harvest \( H^* \). If each diver unit of effort has perfect knowledge of the fishery by moving instantaneously to the most profitable location the following conditional dynamic feedback harvest strategy holds

\[
h_1(t) = \begin{cases} 
H^* & \text{if } T_1 > T_2 \\
0 & \text{if } T_1 < T_2 \\
G(B_1) & \text{if } T_1 = T_2
\end{cases} \quad (4.11)
\]

\[
h_2(t) = \begin{cases} 
0 & \text{if } T_1 > T_2 \\
H^* & \text{if } T_1 < T_2 \\
G(B_2) & \text{if } T_1 = T_2
\end{cases} \quad (4.12)
\]

Under these fixed harvest rate assumptions dynamic system behaviour can be explored. The system shown in Figure 4.15 with a cost difference between the locations \((\Delta > 0)\) illustrates two examples of system recovery following a shift in biomass. In both
cases the biomass shift to the base of each arrow occurs on line $B_1 + B_2 = B^*$. Under this scenario only the daily diver profit in each location is affected since $B^*$ is unchanged. We note that in single population models (e.g. Clark, (2010)) only $B^*$ is observed and from this perspective the system is unchanged.

It can be interpreted from Figure 4.14B that the shift that results in $T_1 < T_2$ will realise a loss of equilibrium system growth because system points produce less growth with increasing separation from the central system point. This explains the recovery trajectory in Figure 4.15 where the harvest rate exceeds the growth rate during system recovery resulting in a loss of biomass at settlement. Alternatively there are two cases where $T_1 > T_2$ follows a deviation from equilibrium. Firstly, if the difference in biomass densities at both locations reduces (small deviation) then surplus system growth is realised. In this case the growth rate exceeds the harvest rate and the excess biomass is absorbed into the system during system recovery. The second case involves a deviation large enough for the difference in biomass densities at Locations 1 and 2 to increase in which case a decrease in system growth occurs initially.
Figure 4.16: Two examples of system behaviour following a biomass shift to the base of each arrow that involves no change to aggregated biomass. The location with the larger daily profit following a shift in biomass is harvested at the rate of $H^*$ while the other location is unfished until profits are equalised. Assuming there is a difference in cost between both locations ($\Delta > 0$) in the case that $T_1 < T_2$ following a system disturbance, the difference between biomass densities of the two locations has increased resulting in less equilibrium system growth.

4.2.6 Stability analysis of system.

It is necessary to define some system parameters to explore the system more rigorously. The logistic growth equation $G(B_i) = rB_i(1 - \frac{B_i}{K_i})$ is used to model net natural growth at sub-system $i$ with a specific rate of growth $r \geq 0$, biomass and a biomass density multiplier. The solution of $G(B) = 0$ under the AGGP gives the MSY rate which occurs at biomass size $B = K/2$ under logistic growth. The determination of the MASY rate for $\Delta > 0$ under the SUBP is derived from $G(\dot{B}^*) = G(\dot{B}_1^*) + G(\dot{B}_2^*) = 0$ and is achieved at equilibrium system points that satisfy $B_1^* + B_2^* = K/2$. 
It is also necessary to give definition to the fishing encounter density function. The nature of this function in abalone fisheries continues to be scrutinised (Coates, Hovel, Butler, Klimley, & Morgan, 2013). A study of the catch per unit of effort (CPUE) that is required to harvest abalone describes the tendency of abalone to form aggregations in discrete habitat patches (McShane, 1996). This behaviour leads to the fishing encounter density changing at a proportionately slower rate than the corresponding change in biomass density. Thus in a system with carrying capacity $K$ and biomass $x$ the following relationship holds

$$f(x/K) = \sqrt{x/K} \quad (4.13)$$

With substitutions of Equation 4.13 into Equation 4.7 and the logistic growth equation into 4.8 the following general equilibrium biomass equations can be derived

$$B_1 = (K_1[\Delta^2 K_2 [-K_1 + K_2] + K_1 B + K_2 B] - 2\sqrt{\Delta^2 K_1^2 K_2^2 [-\Delta^2 K_1 K_2 + K_1 B + K_2 B]} ) / (K_1 + K_2)^2$$

$$B_2 = (K_2[\Delta^2 K_1 [-K_2 + K_1] + K_1 B + K_2 B] + 2\sqrt{\Delta^2 K_1^2 K_2^2 [-\Delta^2 K_1 K_2 + K_1 B + K_2 B]} ) / (K_1 + K_2)^2$$

At equal carrying capacity ($K_1 = K_2 = K/2$) these biomass equations yield

$$B_1 = B/2 - \Delta/2\sqrt{BK - \Delta^2 K^2/4} \quad (4.14)$$

$$B_2 = B/2 + \Delta/2\sqrt{BK - \Delta^2 K^2/4} \quad (4.15)$$

where $B = B_1 + B_2$. 
To determine system stability of the equilibrium points the usual approach of inspecting the eigenvalues of the relevant Jacobian matrix formed by Equations 4.7 and 4.8 was undertaken. This reveals that there are no stable equilibrium points for the case where harvest is fixed and diver behaviour is driven by profit. Equilibrium points with size $B_1 + B_2 > K/2$ have one zero eigenvalue combined with one negative eigenvalue. This means that these points are neutrally stable equilibrium points that will settle at a different neutrally stable point following a small change to system equilibrium. The sub-system equilibrium points with size $B_1 + B_2 < K/2$ have a combination of a positive and a zero eigenvalue, indicating divergent behaviour following a disturbance to equilibrium.

System stability can also be measured in terms of fixed equilibrium effort from the rate of change in system biomass $\dot{B} = G(B) - Ek\sqrt{B/K}$. By taking the derivative of

$$E = \left(\frac{r}{k}\right)\sqrt{BK} (1 - B/K)$$

it is easily shown that maximum effort $E_{MAX}$ occurs at $B = K/3$.

With repetition of the stability analysis above, a Jacobian matrix is obtained from Equation 4.7 and equating the following two system equations to zero;

$$\dot{B}_1 = G(B_1) - E_1 k\sqrt{B_1/K_1}$$  \hspace{1cm} (4.16)

$$\dot{B}_2 = G(B_2) - E_2 k\sqrt{B_2/K_2}$$  \hspace{1cm} (4.17)

it is found that the unstable region of the system under a constant rate of effort occurs in the range $[0, K/3]$ while equilibrium points with biomass size $B_1 + B_2 > K/3$ are
neutrally stable. These findings have relevance to the system model used later in this chapter where management control fishery stock with a fixed rate of total diver effort.

4.2.7 Dynamic diver behaviour

Where the number of licensed divers $E$ is a constant rate the migratory behaviour of divers can be measured by the rate of change in effort at Location 1 by

$$
\begin{align*}
  \dot{E}_1 &= \frac{E - E_1}{\omega} \left[ \frac{T_1 - T_2}{T_1 + T_2} \right], \quad \text{if } T_1 \geq T_2, \quad \text{(a)} \\
  \dot{E}_1 &= -\frac{E_1}{\omega} \left[ \frac{T_2 - T_1}{T_1 + T_2} \right], \quad \text{if } T_1 < T_2, \quad \text{(b)}
\end{align*}
$$

(4.18)

where $\omega$ is a measure of time taken for divers to adjust to changes in profit.

To investigate the effects of diver movement within an abalone fishery it is important to establish a realistic estimation of $\omega$. A simple estimation method is to consider a substantial reduction to diver profit in one location while maintaining the other location at equilibrium. This will enable estimation of $\omega$ because (a) a decrease in profit in a particular location stimulates more rapid migration of divers from that location than does an increase of a similar amount and (b) only the behaviour of divers in one location needs to be predicted.

A method that can be employed to establish a value for $\omega$ involves judging the length of time it would take for diving effort in the affected location to halve its initial value.

Whilst an abrupt profit loss of $0.1T_1$ may be immediately noticeable to fishermen it may be difficult to discern against normal economic fluctuations. Therefore an instantaneous
loss of 0.2\( T_1 \) is proposed as a substantial trigger for diver migration. Determination of the length of time taken by a half of the divers at a location to move in response to a profit loss of 20% is problematic due to unknown factors such as the time scale at which divers monitor their profits. As such, the assumption is made that the delayed response by divers to a 20% loss of profit in a location would cause diving effort to halve within one month (four weeks).

The solution to differential equation (4.18b) can be written as

\[
E_1(t) = E_1(t_0)e^{-\beta t}, \quad \text{where} \quad \beta = \frac{1}{\omega}\left[\frac{t_2 - T_1}{T_1 + T_2}\right]
\]  

(4.19)

Using the assumption above to solve for \( \omega \) the following half-life equation is calculated as

\[
E_1(t_{0.5}) = 0.5E_1(t_0) = E_1(t_0)e^{-\beta t_{0.5}}.
\]

This yields \( t_{0.5} = \frac{1}{\beta} \ln(2) \)

(4.20)

Assume a loss of profit at Location 1 so that \( T_1 = 0.8T_2 \) and that in response half the divers vacate Location 1 in four weeks. Substituting the appropriate values into (4.19) and (4.20) yields \( \omega \approx 0.64 \).
4.2.8 System investigation

To further understand the system effects from diver migration behaviour three scenarios are investigated (Table 4.4). Key features of each scenario and details of system changes are as follows:

- **Scenario 1**
  
  At 50% biomass density the system is neutrally stable. The cost of harvesting either location is the same. A high-mortality event (≈ 27% stock loss) occurs at Location 2 reducing the daily catch from 354kg to 303kg at this location.

- **Scenario 2**
  
  The system is unstable at 20% biomass density. Initially the cost of harvesting Location 2 is twice the cost of harvesting Location 1. Daily harvesting costs are increased by $400 (6.7%) at Location 2 only. In this instance there is no physical effect on the fishery from this change.

- **Scenario 3**
  
  This system state is closely related to Scenario 2. In this case the two locations have an unequal carrying capacity. The system is less profitable than Scenario 2 (with the majority of carrying capacity in the high-cost Location 2). Location 1 has low biomass density and units of effort.

In each scenario the system perturbation triggers a 20% profit loss at Location 2 where $\omega = 0.64$ is the measure of time taken for divers adjust to changes in profit.
Table 4.4: Key variable values at three equilibrium system points. In Scenario 1 the
fishery has biomass density of 50% and homogenous costs. Scenario 2 has substantially
different harvest costs at each location and low density and Scenario 3 differs from
Scenario 2 in that the carrying capacities of both locations are unequal. The harvest price
is \( p = 30,000/\text{t} \) and the determination of equilibrium numbers of daily units of effort is
based on the assumption \( k = 0.5/\text{du.day}^{-1} \).

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Biomass (t)</th>
<th>Carrying capacity (t)</th>
<th>Divers du/day</th>
<th>Catch t/du.day (^{-1})</th>
<th>Cost $/du.day (^{-1})</th>
<th>Profit $/du.day (^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Location 1</td>
<td>30,000</td>
<td>60,000</td>
<td>23.25</td>
<td>0.354</td>
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<td>$7,606</td>
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<td>0.354</td>
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<td>$7,606</td>
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<tr>
<td>Location 1</td>
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<td>60,000</td>
<td>19.59</td>
<td>0.168</td>
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<td>36.69</td>
<td>0.244</td>
<td>$6,000</td>
<td>$1,331</td>
</tr>
</tbody>
</table>

As stated earlier the results given in the following section are based on fixed rates of
total effort \( E \) rather than fixed harvest targets. This provides more convenient
interpretation of the resulting effects of diver behaviour where there are 46.5, 44.7 and
45.4 units of diver effort for the respective scenarios. A plausible specific growth rate for blacklip abalone species in the state of Victoria is $r = 0.2$.

By reducing the carrying capacity of sub-system 1 in Scenario 3 the fishery is less profitable than Scenario 2. Biomass density for each sub-system is reduced because as $K_2 \rightarrow K$ equilibrium levels of $B_2/K_2$ tend towards $B/K$ in sub-system 2 and as $K_1 \rightarrow 0$ then $B_1/K_1 \rightarrow 0$ in sub-system 1.
4.3 Results

4.3.1 Scenario 1

Following the loss of biomass in Location 2 the response of divers to the subsequent change in profits is shown in Figure 4.16A. As expected, one half of the divers at Location 2 migrate to Location 1 within four weeks of the system change. Full diver migration occurs after six months and remains in place for a further twelve months at which time the profit becomes equal at both locations. During the next six months diver migration commences at an almost linear rate in response to continuing gains in Location 2 profit. At the point where effort distribution has returned to the original equilibrium state the profit difference has damped substantially by 78%.

Diver migration continues into Location 2 over the following nine months at which time profits have been equalised for the second time and the majority of total diver effort is operating in Location 2. A further seven months of diver movement into Location 1 completes the first full migration cycle with the original profit difference reduced by 86%. In Figure 4.16B the profit amounts at each location are shown over a period of three complete migration cycles. On completion of the third cycle the daily profit rates are both in excess of 90% of the original equilibrium profit rate.
Figure 4.17: Diver migration effects following a negative biomass shift at Location 2 for the Scenario 1 system state presented in Table 4.4. The diver response to the changed profits shows in A) that after 18 months the profit difference is neutralised, then an over-correction to profit difference follows over a 12 month period and B) the changes to daily diver profits as the difference is reduced (20% → 3%) during the first full
migration cycle. The oscillatory behaviour of spatial profit difference gradually damps over time due to the diver focus on attaining the best available daily profit rate.

4.3.2 Scenario 2

The increase in Location 2 daily harvesting costs to $6400 is a 2.7% increase in the relative cost difference of harvesting each location where $\Delta = 0.227$. If this same economic event occurred under the Scenario 1 system state the impact on diver profit would be much less. The Scenario 2 effects are illustrated in Figure 4.17 where the profit difference is negated with the migration of $0.9E_2^*$ in approximately six months. When diver effort at each location returns to equilibrium levels a sizeable biomass loss has occurred at Location 1. Note that management may be unaware of this change to stock as they make decisions at the aggregated system level.

A contributing factor towards the magnitude of stock loss is borne out by the different behaviour of two system indicators as a response to the increased diver pressure at Location 1. With the system at an unstable equilibrium point the local rates of harvest and net natural growth move in opposite directions adding to system fluctuation. While management policy is aimed at stabilising discrepancies between system growth and harvest at the AGGP level it does not observe the relationship between the two measurements at the SUBP scale. New equilibrium biomass sizes $B_1^*$ and $B_2^*$ can be solved analytically at constant effort ($E = 44.71$) and difference in daily harvesting costs $\Delta = 0.227$. 
Figure 4.18: Diver migration effects on Location 1 following a cost increase at Location 2 under the Scenario 2 system state presented in Table 4.4. The shift in cost is an increase to the existing cost difference $\Delta = 0.2$. At Location 1 there is a biomass loss of 11% in less than one year and a full cycle of diver migration occurs in less than eighteen months. The increased fluctuation in system behaviour is a consequence of divers operating at reducing levels of biomass density and profits.

The relationship between the harvest rate $H$ and daily system profit $\pi$ that arises from the changed economic conditions under Scenario 2 is shown in Figure 4.18. The initial loss of $0.1\pi$ reduces in the first two months as migration to the more profitable Location 1 takes place. During this time the negative correlation between harvest and profit is caused by migrating divers experiencing lower catch rates on arrival Location 1. This is followed by a longer period where the two system indicators experience a linear relationship.
The concept that the harvest rate falls by 25% while profit falls by less than 10% during the same period is difficult to grasp. The explanation is that the profit per unit of catch is always higher in Location 1 due to lower harvesting costs therefore divers that initially migrate to Location 1 earn more profit for less catch. Irrespective of the relationship between the rates of harvest and profit the instability in system profit that follows a mild increase in the cost of one location is of most importance to divers and license owners.

Figure 4.19: Diver migration effects on economic indicators following a cost increase at Location 2 under the Scenario 2 system state presented in Table 4.4. The rapid migration of divers into Location 1 initially produces a plummeting rate of harvest at low catch rates that translates to a profit recovery. At the point of minimum harvest rate there have been Location 1 losses of biomass ($0.08B_1^1$) and daily profit ($0.1T_1^1$).
4.3.3 Further analysis of Scenario 2

Since the effects from the mild economic change are considerable under Scenario 2 it is important to analyse the robustness of the result. Further testing that only varied the size of $\Delta$ did not have much influence over the cycles of diver migration or the nature of biomass depletion at Location 1 as changes in the different result sets for both were proportionate to the change in $\Delta$. We note that the effects of $\Delta$ on the fishery are also dependent on the magnitude of $c_1$ relative to maximum diver revenue (pk). Where Location 2 daily harvesting costs are double those of Location 1 ($\Omega = 1.0$), in a low-cost fishery ($c_1$ small) there is a negligible effect on divers’ choice of harvesting either location. Alternatively larger harvesting costs mean that the value for $\Omega$ will have greater economic influence on divers’ choice of location. It is also important to note that changes to biomass density will affect the result output. Clearly, biomass below $0.2K$ combined with large values for $\Delta$ result in lower biomass density at Location 1 leaving it more vulnerable to increased attention from divers.

The impact that alternative rates of $\omega$ has upon the stability of system biomass is demonstrated in Figure 4.19. For convenience we refer to the response times that correspond with $\omega = 0.16, \omega = 0.64, \omega = 8$ as quick, medium and slow respectively. The system behaviour gives the primary management view of the trend in fishery biomass following the change in economic conditions. Modest gains in biomass are initially made during the first diver migration cycle. This view belies the initial loss of biomass at Location 1 that is coupled with stronger biomass gains at Location 2. Biomass continues
to grow after the profit difference is neutralised until enough critical mass of diver effort migrates to Location 2 where the catch rate has increased above the equilibrium rate.

Under the *quick* diver response time a half of the divers in Location 2 migrate within one week. As a result the harvest rate remains below the equilibrium rate for six months at which time biomass has increased by almost 2%. This response is closest to the instantaneous migration scenario that was modelled in Chapter 3, demonstrating that small delays in diver responses to profit change will level profits in a relatively short period of time and minimise the magnitude of change to biomass.

The *slow* diver response time leads to less than a half of Location 2 diver effort migrating before profit equalisation occurs. In this case, a gradual rise in biomass peaks with a 2.5% increase over two years. This response by divers might be feasible where they have a combination of poor information and a poor economic understanding of the fishery. One completed diver migration cycle at the *slow* diver response rate indicates that the initial gain in biomass following an economic change is the beginning of a long term downward biomass trend. In the extreme case as $\omega$ approaches $\infty$ no diver migration occurs; minimising system fluctuation. In all cases the system tends towards extinction where a constant rate of diver effort is maintained.
Figure 4.20: Diver migration effects on system biomass at alternative rates of diver response time following a cost increase at Location 2 under the Scenario 2 system state presented in Table 4.4. For the range of diver responses to profit change shown, \( \omega = 0.16 \), \( \omega = 0.64 \) and \( \omega = 8 \) correspond to quick, medium and slow respectively. The vertical dashed lines denote the completion of one migration cycle. With quickening diver response rates to changing profit, the oscillatory effect upon system biomass increases with shorter periodicity and amplitude range. The long term effect of the economic change is a downward effect upon biomass, with an unchanging rate of diver effort.

### 4.3.4 Scenario 3

With application of the same economic change (positive shift in \( \Delta \) of 2.7%) to the Scenario 3 system point the system effects of diver migration are compared between Scenarios 2 and 3 in Figure 4.20A. Under Scenario 3 the system effects are more noticeable as the cost increase results in a 30% reduction in \( T_2 \). During the initial diver response the level of effort in Location 1 more than quadruples as the profit difference is swiftly reduced. During the first over-correction phase under Scenario 3, Location 1
remains unfinished for a considerable time because the equilibrium diver capacity of Location 1 is too small to redress the economic imbalance effectively. The corresponding comparison that shows the effects of diver response on Location 1 biomass is presented in Figure 4.20B. While more biomass is lost from this location under Scenario 2 the loss of biomass density for Scenario 3 is double that of Scenario 2.

A)
B)

**Figure 4.21:** System comparisons between Scenario 2 with equal carrying capacities at both locations and Scenario 3 with unequal carrying capacities, following the same cost increase at Location 2 for both scenarios. The Scenario 2 results are reproduced from Figure 4.17. The increased fluctuation in diver migration at the Scenario 3 system point occurs in A) because of the imbalance of equilibrium effort at both locations ($E_1^* = 0.19E$). In B) the low equilibrium biomass density at Location 1 ($< 0.1K_1$) is vulnerable to stock depletion.

### 4.3.5 Multiple location scenario

A different perspective of Scenario 3 is given by Scenario 4 in Table 4.5 where the same system point comprises more than two sub-systems. Scenario 4 is obtained by division of Location 2 under Scenario 3 into three locations of identical carrying capacity with the assumption that harvesting costs are homogeneous. For systems that contain more than two sub-systems the migratory behaviour of divers in response to changed profits can be measured using the following assumptions
• In a system of \( n \) locations a diver in location \( k \) will determine the most profitable location as \( T_{MAX} = \text{Maximum}\{T_1, \ldots, T_{k-1}, T_{k+1}, \ldots, T_n\} \)

• The rate of change of diver effort at location \( k \) in response to changing profits is

\[
\begin{align*}
\dot{E}_k &= \sum_{j=1}^{n} \frac{E_j}{\omega} \left[ \frac{T_k - T_j}{T_k + T_j} \right] , \text{ if } T_k \geq T_{MAX}, \quad (a) \\
\dot{E}_k &= -\frac{E_k}{\omega} \left[ \frac{T_{MAX} - T_k}{T_k + T_{MAX}} \right] , \text{ if } T_k < T_{MAX}, \quad (b)
\end{align*}
\]

If the same Location 2 cost increase (\( \Delta = 0.027 \)) is applied to Locations 2, 3 and 4 then the system behaviour of Scenario 4 will be identical to that of Scenario 3. This can be inferred by inspection of the respective equilibrium parameter variables of each scenario where the only difference is that biomass, carrying capacity and diver effort all differ by a factor of three.
Table 4.5: Key variable values of two system equilibrium points that consist of four locations. Scenario 4 gives a different perspective of the Scenario 3 system point where Location 2 is divided equally into three identical locations. Scenario 5 is based on Scenario 4 with different daily costs attached to each location. The harvest price is \( p = \$30,000/t \) and the determination of equilibrium numbers of daily units of effort is based on the assumption \( k = 0.5 t/\text{du.day}^{-1} \).

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Biomass (t)</th>
<th>Carrying capacity (t)</th>
<th>Divers du/day</th>
<th>Catch t/du.day(^{-1})</th>
<th>Cost $/du.day(^{-1})</th>
<th>Profit $/du.day(^{-1})</th>
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<td>$1,331</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>B/K = 0.2, ( \Delta = 0.2 )</td>
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</tr>
<tr>
<td>Location 1</td>
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<td>12.56</td>
<td>0.268</td>
<td>$6,750</td>
<td>$1,310</td>
</tr>
</tbody>
</table>

Scenario 5 gives a different perspective of Scenario 3 whereby the summary cost of Location 2 (Scenario 3) expands to a range of costs for Locations 2, 3 and 4. The 12.5% step in daily cost difference between locations has a large impact on the corresponding relative differences in biomass however this does not translate to large differences between the number of diver units of effort required at each location. Following the cost
increase of $400 to Locations 2, 3 and 4 the relative migration pattern from each location is almost identical. Profit recovery occurs marginally quicker at Location 4 where biomass density, equilibrium effort and catch rates are highest.

Overall there is little difference between the aggregated system behaviour of Scenarios 4 and 5. The increased range of costs in Scenario 5 leads to a marginally less profitable system leaving Location 1 more vulnerable to economic change. However this is offset by the distribution of effort under Scenario 5 where Location 1 gains some migration protection with a slightly larger proportion of diver effort than for Scenario 4. This occurs because as biomass approaches $K_i/3 = 10,000t$ the extra equilibrium effort required is decaying towards the maximum rate $E_{i,MAX}$. The lack of system sensitivity to differences between Scenarios 4 and 5 suggest the important factors to be considered by management are; i) What is the biomass density of the most vulnerable low-cost location?, and ii) How much diver effort can potentially move to the low-density location in a short period of time?
### 4.4 Discussion

The differences between system behaviours of the sub-system model and a single population model can be pronounced. In the latter model economic indicators such as bionomic equilibrium and maximum economic yield are the only system representation of economic detail. The sub-system model provides insight into the intra-system effects and net system effects of profit-driven diver behaviour. These effects are not observable in models designed at the management scale.

It was found in general that management should have no concerns about the spatial patterns of divers where high profitability exists. This was demonstrated in Scenario 1 where the system responds positively following eighteen months of divers operating exclusively in one location. Similarly, if the harvesting cost differential between locations is not substantial the difference in equilibrium densities of the locations becomes negligible. In such cases a system with balanced biomass density is more suited to withstand economic change. Where none of these conditions is true, the sub-system perspective sheds new light on why management should avoid the operation of fisheries at low biomass density. Fisheries with low biomass density combined with a large harvesting cost differential can experience strong system fluctuations from diver response to changing profit. Worse still, these fluctuations can have serious implications for low-cost locations that contain the lowest biomass densities within the fishery.
Under Scenario 2 the impact on biomass from diver migration in response to economic change is more volatile from the sub-system perspective in comparison to the aggregated system perspective. This is to be expected as the whole system is constrained by a constant rate of total effort while each location can experience levels of diver effort that are far removed from equilibrium rates. Where shorter response times of divers limit the effects upon total biomass the reverse is true from the sub-system perspective. A change to profit in one location combined with a quick diver response time generates sharp changes in biomass trend at both locations. In this circumstance the value of biomass sampling each location during this period would be weakened by the system fluctuation. In particular any delay between sampling of alternate locations will increase the likelihood of incorrect assessment of system abundance.

The difficulties associated with management interpretation of the system behaviour in Scenario 2 are not confined to the sub-system perspective. The changing trend in total biomass also poses problems for management. While the results only consider a system with a constant rate of total effort, management could use Total Allowable Effort (TAE) adjustments. A change to TAE normally involves altering the number of annual days fished per license and is complicated by lag times in management detection of biomass trends and the implementation of a corresponding adjustment. An increase to TAE near a biomass peak will only serve to hasten the biomass descent adding further instability to biomass and harvest. In this scenario it would be prudent to leave TAE at the existing
rate and monitor biomass trends with conservatism over a period of time irrespective of the value of $\omega$.

The influence of the size of $\Delta$ upon the distribution of biomass density at the subsystem level is a crucial dynamic of a fishery system. An increase in $\Delta$ also means increased difference between the respective biomass densities. A typical adaptation of Scenarios 2 and 3 (large $\Delta$) to an actual fishery would be where one location near a fishery access point with a low cost of harvesting is complemented by a location carrying greater harvesting costs. The results demonstrate that if the low-cost location becomes the most profitable due to a change in fishery economics then the migration of divers from the less profitable site can quickly deplete biomass nearest the fishery access point.

In particular the results for Scenario 3 demonstrate that where the location with the lowest harvesting cost accounts for a smaller portion of carrying capacity there is an increased vulnerability to biomass depletion. Under the deterministic modelling of this scenario the loss of biomass is smoothly recovered over a period of time. However it is important to note that in an actual fishery that recovery may be affected by other dynamic fishery behaviour. A superficial view of the result in Scenario 3 would suggest that there is only a small biomass amount is at risk in Location 1 however the real issue for the fishery is the potential threat to habitat of $0.25K$. 

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The stability analysis in the Methods section demonstrates that under fixed harvesting rates that system stability occurs where the sum of biomass parts exceeds $K/2$ from both system perspectives. This reinforces the benefits of operating a fishery at biomass densities that realise a gain in net natural growth in the event of a loss of biomass. It is important to note that even where the system is at a neutrally stable point it remains possible that $B_1$ does not exceed $K_1/2$ in which case a loss of $B_1$ is accompanied by losses in net natural growth. The fishing encounter density chosen in this study meant that the fishery could be stabilised under a constant rate of diver effort within the range $[K/3, K]$. Theoretically this means that within this range a loss of biomass corresponds with a required increase in effort to sustain the reduced biomass size. The maximum equilibrium rate of effort that occurs at $B = K/3$ can be interpreted as an indicator of the dwindling diver effort that is required below this biological reference point in a depleted abalone fishery.
5. Conclusion

Australian wild abalone fisheries have defied worldwide trends of unsustainable abalone harvesting, remaining generally viable until recently. Today adverse market conditions, disease and reduced catch quotas challenge the capacity of Australian abalone fisheries to sustain into the future. Profits are down and the resilience of these fisheries is under threat. As a consequence of the current position faced by Australian abalone fisheries, it has become critical that effective management strategies are implemented. Questions that have arisen include: Which management strategy is most likely to maintain sufficient biomass to produce consistent harvest rates into the future? Should the fishery be managed at a scale that corresponds with the divers’ spatial preferences? How does diver migration behaviour affect the attainment of management objectives?

A system model that includes key abalone population dynamics adapted from biological research was developed to determine a suitable strategy for maintaining future harvest levels. A management module was incorporated into the model to facilitate separate evaluation of management strategies following a high-mortality event. This model was then used as a platform for investigation of system performance at different spatial scales of management. The spatial model employs the same population dynamics in two reefs of equal size where the only difference is the potential for regional quota to be removed from one reef only. A different modelling strategy was undertaken to assess the system effects of diver migration behaviour. A bioeconomic model was
developed to capture the dynamic intra-system effects that stem from diver response to changing profits. This additional model detail was offset by the inclusion of a simpler population component for the abalone system dynamics.

The findings regarding the potential implementation of various proposed management strategies ranged from those that management should not consider to others that could be worth consideration. It was found that the proposed harvest threshold mechanism (HTM) rule in its current form is inadvisable. A problem with using commercial catch as feedback to a control rule can arise where target harvest is caught over a number of years. By the time a low harvest threshold is eventually reached the population may not be of a structure, suitable for harvest recovery. Similarly, making harvest decisions at finer spatial scales should not be considered by management under conditions where fisheries have either good biomass abundance or a harvesting cost structure with a low cost range.

Alternatively the mean length of commercial catch (MLCC) control rule that was proposed exhibited better system behaviour and could be given management consideration. This strategy was able to provide the detection of a large downward shift in biomass through subsequent changes to the mean length of commercial catch and then set appropriate adjustments to harvest. A downside to the management implementation of this strategy is the increased cost and effort for industry and inconsistency of yield.
While abalone fisheries are subject to analysis at fine spatial scales over various measurements, the concept of placing management boundaries at these scales requires much thought. It was found that the profit-driven behaviour of abalone divers provides further evidence to suggest the importance of management pursuing such measures. This is particularly true of abalone fisheries that operate at low biomass density and have substantial differences between the costs of harvesting different locations.

A limitation of the findings relates to the expense of policy implementation. In particular the logging of commercial catch data and managing a fishery at fine spatial scales are both technology-dependent management strategies. Implementation of either strategy would involve a considerable cost of additional infrastructure followed by the ongoing costs of fishery monitoring. If the costs of implementing these strategies were to decrease substantially in coming years then wide-scale introduction of these measures will have more appeal.

The sub-system perspective of an abalone fishery provided detail of how these complex systems can be affected by diver behaviour. It was shown that for an increasing difference in the daily cost of harvesting separate fishery locations, a growing difference between the respective biomass densities results. The consequences for low-density locations are more severe in circumstances where most of the fishery consists of areas with much higher harvesting costs. In these cases the majority of diving effort is conducted at high cost. This means that whenever the smaller, more lucrative area
becomes the most profitable area of the fishery, it is subjected to extreme levels of diver
effort causing rapid stock losses.

Whilst the methodologies used are aligned with abalone fisheries some of the
findings could have implications for fisheries in general. This is true of the evaluation of
the two management strategies that is conducted in Chapter 2. The advantages that the
mean length of commercial catch strategy has over the harvest threshold mechanism
could apply to many fish species. In short, the continuous nature of the catch adjustment
method under this regime, combined with potentially short time delays associated with
the implementation of catch adjustments would be suitable for many fisheries.

The models developed in this thesis and the findings achieved have yielded insight
into an abalone system. This has led to further understanding in the areas of
management harvest control, diver behaviour and the influence which fishery
economics can have on a system. Hopefully this will contribute towards improved
management practices and hence, sustainability of Australian abalone fisheries.
Furthermore, the findings will have relevance to the sustainable management of other
sedentary species.
References


Appendix

Fishing selectivity is applied to length classes marginally above the LML. Blacklip abalone have a strong preference for inhabiting cryptic spaces, especially during the juvenile period of their life cycle. To model this characteristic, a function of emergence from cryptic habitat is employed. Exploitable biomass includes only selected, non-cryptic, abalone that are of legal size. The planned monthly harvest is divided by exploitable biomass to obtain the fishing exploitation rate. This rate is applied uniformly to each length class, implying that fishing fully selected abalone is a random process.

\( n_l(t) \) represents population numbers of length category \( l \) at time \( t \).

Recruitment is calculated every season using the Beverton-Holt formula

\[ r(t) = \theta(t) \frac{(b^M(t-24) / b^M(-\infty))}{\alpha + \beta(b^M(t-24) / b^M(-\infty))}, \]

where \( \theta(t) \) is the spawing season weight for month \( t \) and \( \alpha, \beta \) are parameters of the stock-recruitment function that are governed by steepness coefficient \( h \), with

\[ \alpha = \frac{(1-h)}{4hR(-\infty)}, \beta = \frac{(5h-1)}{4hR(-\infty)}, \]

\( b^M(t) \) represents mature biomass during month \( t \) with \( b^M(t) = \sum_{i=1}^{\infty} w_i m_i n_i(t) \),

where \( w_i \) represents weight at length, with \( w_i = a \bar{L}^b \), where \( a \) and \( b \) are parameters of the weight-size relationship and \( \bar{L} \) is the median length of length class \( l \).
proportion of mature animals at length and $m_l = (1 + e^{-\ln(1.9)\left(\bar{l} - l_{50}^{mat}\right)/\theta^{mat}})^{-1}$, where $l_{50}^{mat}$ is the median maturity length and $\theta^{mat}$ is the ogive width (5th percentile).

$$f(t) = \frac{c(t)}{b^E(t)},$$

where scalar $C(t)$ represents commercial catch at the end of month $t$ and $b^E(t)$ represents the exploitable biomass at length during month $t$, with

$$b^E(t) = \sum_{i=1}^c w_i s_i h_i n_i(t)e^{-MAt}.$$

Fishing selectivity is a function of length class which models the selectivity behaviour of divers in the length range directly above the LML, where

$$s_l = \begin{cases} 0, & l < l_{min} \\ P + (1 - P)\frac{l - l_{min}}{l_{max} - l_{min}}, & l_{min} < l < l_{max} \\ 1, & l \geq l_{max} \end{cases},$$

where $l_{min}$ is the minimum length fished, $l_{max}$ is the first length fully fished and $P$ represents proportion of animals selected at $l_{min}$. $h_i$ represents the proportion of emergent animals at length and for length class $l$, $h_l = (1 + e^{-\ln(1.9)\left(\bar{l} - l_{50}\right)/\theta})^{-1}$, where $l_{50}$ is the median emergence length and $\theta$ is the ogive width (5th percentile).

Population initialisation occurs over a forty year period, where the fishing exploitation rate is zero and recruitment is a function of the mortality rate. The importance of establishing the pre-fished deterministic equilibrium population is to determine the mature biomass $b^M(-\infty)$ at this point, which is a key component of the Beverton-Holt recruitment function that is used in this model to calculate monthly
recruitment numbers. Five sixths of annual recruitment occurs during a fixed four month spawning season.

The parameter settings used during simulation are tabled as:

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