Final Report

Costs-benefit Analysis of Implementing Alternative Techniques for Rehabilitating Reefs Severely Depleted by the Abalone Viral Ganglioneuritis Epidemic.

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Biospherics P/L

Western Abalone Divers Association

FRDC Project 2008/076

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WADA
Western Abalone Divers Association

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OBJECTIVES:

1. Review of current and past research related to abalone reseeding to determine whether the approach is initially feasible.
2. Construct a quantitative economic and population dynamics model of reseeding, translocation and natural rebuilding for blacklip abalone reefs and use it to analyze the likely costs and benefits of alternative techniques of rehabilitating a reef code in the Victorian western zone.
3. Prepare and present a written and audio-visual report to WADA at their October 2008 and February 2009 reef assessment workshops detailing modelling methods, underlying assumptions and results of evaluating alternative rehabilitation techniques.

NON TECHNICAL SUMMARY:

OUTCOMES ACHIEVED TO DATE

The outcome of this project was that members of the Western Abalone Divers Association (WADA) were provided with cutting edge analysis of the relative cost-benefits of using reseeding of juveniles, translocation of adults or natural rebuilding processes to recover reefs in the aftermath of infection by Abalone Viral Ganglioneuritis (AVG). The analysis was presented to WADA through meetings in August 2009 and January 2010 and seems to have enabled WADA to overcome previously polarized views on the topic. Together with the evolving economic climate for the Australian abalone fishery this analysis seems to have consolidated a general consensus amongst members that managing the natural rebuilding of stocks is likely to be the most cost effective approach to recovering Western Zone Reefs following AVG.

Following the epidemic in 2006 and 2007 caused by Abalone Viral Ganglioneuritis (AVG) through the Victorian western zone, the Western Abalone Divers Association (WADA) began discussing the relative merits of alternative methods for rehabilitating the most heavily impacted reefs. A workshop of experts convened by WADA in December 2007 recommended that prior to initiating costly field programs quantitatively rigorous analyses should be conducted comparing likely costs and benefits of pursuing alternative strategies involving reseeding, translocation or natural recovery processes. Subsequently the National Abalone Health Workplan Priority Setting Workshop held in June 2008 also gave high priority to conducting an economic cost benefit analysis of rebuilding intervention. This Tactical Research project was initiated by WADA in response to these recommendations and priorities with the aim of inform WADA members of the expected relative merits of the different means of rehabilitating abalone reefs.

This project began by surveying past and current research related to abalone reseeding, translocation and mortality rates. Values of important assumed parameters for the cost-benefit model were developed on the basis of this literature. The literature shows that, at least, in Japan the augmentation of abalone stocks by reseeding juveniles and translocating adult is technically feasible, although the literature from other parts of the world is more equivocal. It is not possible to determine from the literature whether the difference between the Japanese experience, and that of other countries, is due to lower predator levels in Japan, as claimed by some or the inherent biases associated with the differing experimental designs employed. Only in Japan have long term, large scale augmentation programs been attempted and fishery wide returns monitored until the augmented year classes have been fished out so that a total count of that year class is possible. Outside Japan studies have been small scale, short-term experiments, so that recapture rates have been depressed by the cryptic nature of juvenile abalone, and movement out of research areas.

On the basis of the literature a quantitative model of the population dynamics and economic processes underlying reseeding, translocation and natural rebuilding of a blacklip abalone reef in the Western Zone of Victoria was constructed, and used to analyze the likely costs and benefits of the alternative techniques of rehabilitating the Kilarney reef code in western Victoria. This appears to be the first time a fully specified population model has been used to analyse the biological and economic processes underlying the rehabilitation of abalone reefs.

A preliminary analysis, using place-holder values was presented to a WADA workshop 13 August 2009 which suggested that when the cost of capital is included none of the selected enhancement scenarios were as cost-effective as the predicted natural rebuilding process. This workshop then chose the range of parameters which were analysed for the final analysis. As requested by WADA stakeholders the final analysis incorporated higher seed costs, local length weight data, and the cost of outplanting. Its results were even less favorable for any form of intervention, than the preliminary analysis using place-holder balues for key assumptions. Refitting the population model with the local data produced a modeled stock which was smaller but more productive making it even harder for any strategy to significantly improve on the rate at which natural rebuilding processes are expected to work for no cost. Taking into account the cost of capital, the natural rebuild scenario wins out in all the scenarios considered. Although the results suggest that augmentation of breeding biomass immediately after a die-off will come closest to covering the cost of the capital involved.

Within that context the two techniques, translocation of adults, or reseeding were similar in effectiveness. Across the scenarios modeled in the final analysis, translocation was estimated to never fare too badly, always at least paying for direct costs, but not covering the cost of capital, while reseeding only covered direct costs when the price of abalone exceeded $40/kg and seed is cheap. However, none of the intervention scenarios produced any return when the cost of capital was considered.

The results of this analysis are conditioned by the assumption that even low at low densities the population remains productive, albeit from a low biomass base. The assumed form of the stock recruitment relationship used in this analysis is the accepted form in Australian abalone assessment science, however, it precludes effects that some abalone ecologists fear overwhelm abalone productivity at very low densities. The actual shape of typical abalone stock-recruitment curves has never been directly studied and this is an area of unresolved uncertainty when it comes to the interpretation of these results.

KEYWORDS: **Cost-benefit, abalone enhancement, population modeling**

Acknowledgments

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Background

In May 2006 an epidemic caused by Abalone Viral Ganglioneuritis (AVG) began infecting wild stocks of abalone in the Victorian western zone, rapidly spreading through the zone causing mortality rates of 40-95%. It was the observation of WADA's divers that, after being closed to fishing, the reefs that had suffered relatively lower mortality rates (<75%) visibly started to recover. Supported by an FRDC TRF project 'Indicators and strategies for resumption of harvesting following catastrophic loss of abalone' WADA members agreed that the 'lightly' impacted reefs should be closed for at least threes closure before being re-opened (some in 2009) with conservative size limits and catches. In response many reefs remain closed at the time of writing and the zonal TAC is >75% lower than before the infection.

In contrast to reefs that suffered relatively low mortality rates WADA's divers observed that after a years closure a number of formerly important reefs which suffered very high mortality rates (>90%) did not appear to be recovering. Worryingly Californian studies (Miner et al. 2006) suggested disease related reductions of abalone grazing pressure might lead to overgrowth of suitable settlement surfaces retarding natural rates of re-colonisation. WADA estimates one 'high mortality' reef, Kilarney lost 100-200t of biomass and its members were concerned that without intervention there would be a semi-permanent loss of productivity from some important reefs.

To prevent this WADA's members began discussing the relative merits of alternative methods for rehabilitating the most heavily impacted reefs by alternatively:

- Re-seeding of hatchery produced juveniles,
- Translocation of adults from other reefs in the western zone
- Long term closure to allow natural recruitment processes.

On 10-11 December 2007 the Western Abalone Divers Association (WADA) convened a workshop on Translocation and Reseeding, chaired by Prof. Gee Chapman, attended by experts in the fields of abalone culture, reseeding, translocation and ecology, and observed by Mr Jason Fromm (FRDC). The workshop reviewed the western zone situation post infection with the AVG virus, and knowledge of reseeding and translocation techniques. It developed proposals for R&D to support rehabilitation of western zone reefs heavily depleted by the virus. It recommended that prior to initiating costly field programs quantitatively rigorous analyses should be conducted comparing likely costs and benefits of pursuing alternative strategies involving reseeding, translocation or natural recovery processes.

Subsequently the National Abalone Health Workplan Priority Setting Workshop held in Melbourne 18 June 2008 considered four priority work areas:

1. Epidemiology
2. Stock sustainability
3. Biosecurity
4. Communication

Conducting an economic cost benefit analysis of rebuilding intervention options was given the highest possible priority under the Stock Sustainability work area; scoring 4 out of 4 for both urgency and priority.

This Tactical Research project was initiated by WADA in response to these recommendations and priorities to inform WADA members of the expected relative merits of the different means of rehabilitating abalone reefs.

Need

A cost-benefit analysis of alternative rehabilitation techniques was needed to inform members of the relative merits of alternative rehabilitation techniques and the likely magnitude of cost for rehabilitating at the scale of entire reefs.

Results of this project will enable WADA members to evaluate the likely costs and benefits of alternative methods for rehabilitating abalone reefs heavily impacted by the AVG virus. The output of this project will enable WADA to make considered and informed plans for how best to rebuild productivity in their zone that might otherwise be lost to them semi-permanently. The primary beneficiaries of this outcome will be WADA members who will be empowered to carefully consider both the costs and benefits of alternative approaches, and who will stand to benefit directly if the recovery time of these reefs can be reduced.

The State of Victoria and in particular Victorian recreational divers stand to benefit directly from the rapid recovery of reefs heavily depleted by the AVG virus. More broadly, and in the longer term, because the AVG virus is likely to spread more widely through the Australian abalone industry, and because there are areas of abalone reef throughout Australia that have been heavily depleted through localized overfishing, the outcomes of the project will benefit the commercial and recreational sectors of the abalone fishery Australia wide, who may be interested in intervening to recover abalone stocks in their own regions. Less directly, the broader Australian community will also benefit from cost-effectively and rapidly rebuilding the productivity of a rural industry which had a capitalized value of approximately $4 billion prior to the impact of the AVG virus.

Objectives

1. Review current and past research related to abalone reseeding to determine whether the approach is initially feasible.
2. Construct a quantitative economic and population dynamics model of reseeding, translocation and natural rebuilding for blacklip abalone reefs and use it to analyze the likely costs and benefits of alternative techniques of rehabilitating a reef code in the Victorian western zone.
3. Prepare and present a written and audio-visual report to WADA at their October 2008 and February 2009 reef assessment workshops detailing modelling methods, underlying assumptions and results of evaluating alternative rehabilitation techniques.

Methods

This project was a collaboration between Dr Jeremy Prince (Biospherics P/L), Dr Sarah Jennings and Satoshi Yamazaki (University Tasmania) and Mr Mark Gervis (Southern Ocean Mariculture) and Harry Peeters (Western Abalone Divers Association). Dr Prince undertook the programming, analysis and writing. Drs Jennings and Satoshi Yamazaki provided their extensive experience with economic and cost-benefit analyses and specified how to make the economic side of the analysis quantitative, Mr Mark Gervis provided estimates of the costs likely to be associated with reseeding and scrutinized assumptions utilized by the analysis. Harry Peeters the Executive Officer of WADA initiated this project, scrutinized assumptions utilized by the analysis and was the liaison between the project and WADA members.

Synthesis

A review of current and past research related to abalone reseeding and the study of abalone mortality rates. A primary aim of the literature review was to initially determine whether the techniques for rehabilitating abalones stocks using reseeding or translocation techniques are feasible. The other aim was to provide sound estimates for incorporation into the model used in this analysis. Estimates of natural and juvenile mortality rates, along with expected cost frameworks for both reseeding and translocation provide some of the most important data inputs to this analysis. To ensure the most defensible assumptions were used, the published literature and results of previous and ongoing FRDC studies were gathered and analysed. The synthesis has been presented as a literature review in Appendix 3. In Appendix 3 the literature review is followed by a simple re-analysis of data gathered from the Japanese literature which derives estimated mortality rates used in the analysis.

Modelling

As a part of the FRDC TRF project (Project 2007/066) entitled ‘Rapid response to abalone virus depletion in western Victoria: information acquisition and reefcode assessment’ an age-based spreadsheet population model was developed for three western zone reef codes including Kilarney; which was one of the large heavily depleted reef codes WADA members were discussing with regard to the potential for proactively rehabilitation. That spreadsheet model was fitted to the available quantitative and qualitative data gathered during WADA’s reef assessment process and simulates past and future population dynamics in this reef code. As a part of the previous TRF project the spreadsheet model was validated against a more sophisticated and computationally complex adaption of the Vic. DPI’s size based zonal model for the reef code 'The Crags'. That comparison showed the age-based spreadsheet model produced similar results to the DPI model which itself is an adaption of the Australian Abalone Assessment Model. The age-based spreadsheet model has since been used to model scenarios for the natural rebuilding of the three reef codes after varying levels of viral impact.

This project adapted the age-based spreadsheet model developed for the Kilarney reef code to model the biological and economic processes involved with both reseeding and translocation processes and compared these results with the models prediction for a Natural Rebuild of the stock without any rehabilitation program. The existing population dynamics component of the model required some modification because like the current DPI zonal model an underlying assumption is that adult mortality rates apply to all age classes and the underlying stock recruitment relation is parameterized around this assumption. Thus the current stock assessment model assumes unrealistically low juvenile mortality rates but it also estimates correspondingly lower levels of 0^+ recruitment. In the context of the zonal stock assessment these two assumptions cancel each other out, so that by the age of recruitment to the fishery, estimated year class abundance is unbiased. So within the context of assessing emergent recruited adult stocks the unrealistic assumption of low juvenile mortality has no impact, however when analyzing the dynamics or reseeding and translocation it is necessary to accurately model juvenile mortality rates and this meant re-estimating the stock recruitment parameters currently used by the model. This was achieved through the synthesis of the available literature (Appendix 3) to develop defensible assumed mortality rates for the juvenile year classes and then setting these values into the base case model. The model was then refitted to produce the closest fit to observed and inferred historic trends with the minimizing routines fitting to the recruitment parameters.

The structure of the model and the parameters used, in development and in final form, is documented in Appendices 4. For brevity this detail is not represented here.

Comparative Cost-Benefit Analysis

I am indebted to Drs Sarah Jennings and Satoshi Yamazaki (UTAS) for suggesting the methodology used to quantitatively compare differing outcomes of the various scenarios.

The conceptual framework for this approach is to consider the value of the Kilarney resource prior the impact of AVG. In the fitted version of the model used for this analysis the Maximum Sustainable Yield (MSY) is 33t and this is produced by the stock when its Biomass (B_{MSY}) is 65t. By convention in the abalone fishery catch entitlements are worth some multiple of the expected annual income from the sale of the catch. Assuming the multiplier to be seven (7), which is similar to its medium term value in the fishery, and valuing abalone at \$25/kg, the capitalized value of the Kilarney resource at B_{MSY} can be estimated to be \$5,775,00. However, the resource is currently below this level and the value of having it at B_{MSY} must be deferred until it can be rebuilt to that level. This Deferred Value can be estimated by applying discount rates to the value that would be placed on the resource if it were currently at B_{MSY}.

Thus establishing B_{MSY} and its value as the benchmark, the time taken for each scenario to rebuild to B_{MSY} was estimated, and a range of assumed discount rates (3%, 5%, 7%) were applied over this time frame, to the estimate the value of having the resource at B_{MSY} ($5,775,00). This is referred to as the Deferred Value of the resource. In this way the Deferred Value from Naturally Rebuilding the stock, which has no direct costs attached, was calculated as simply the value of the stock at B_{MSY} discounted for the time period taken after 2010 to rebuild the stock to B_{MSY}. For the alternative rehabilitation scenarios the cost of each rehabilitation strategy were compounded by the discount rates over the time period taken to rebuild to B_{MSY} and then subtracted from the Deferred Value.

The detail of the equations used to implement this conceptual framework can be found in Appendices 4.

Consultation and Development of the Analysis

The contract for this project was signed in December 2008 and the project finally initiated in early 2009. The development of the model and analysis proceeded in phases.

Initial Round of Synthesis and Model Development

During March and April 2009 the literature on reseeding and translocation was surveyed and feasibility of reseeding and translocation initially assessed. The literature on the study of abalone mortality rates was also surveyed and a schedule for mortality rates developed for use in the model. During May 2009 the age-structure model developed through project 2007/066 was adapted to estimate population trends and simple costs of alternative rehabilitation strategies. This version of the model was used to conduct a simple illustrative analysis of various rehabilitation strategies based on placeholder values gleaned from the literature to fill all major assumptions. An unpolished report based on this analysis was provided to all project participants in June 2009. On 23 June 2009 Drs Sarah Jennings and Satoshi Yamazaki (UTAS) met with Dr Prince in Hobart, Tasmania to specify the structure of the economic module of the model. This was incorporated into the model structure and a preliminary cost and benefit analysis was completed using the same placeholder values. In this initial analysis a wide range of alternative rehabilitation strategies were explored relatively naively in order to explore and describe the model's behavior. A report of this preliminary analysis was prepared for WADA and project collaborators during July 2009.

WADA Workshop

On 13 August 2009 a workshop on the project was held with WADA members in Port Fairy, Victoria and the results of the preliminary analysis were presented and discussed. Mr Mark Gervis attended and provided an outline of a reseeding strategy that would be logistically feasible for Southern Ocean Mariculture involving seeding 500,000 1 year old (20-30mm) abalone each year for three years, with the seed costing approximately @ 2c/mm of length. The workshop agreed the values to be used for key parameter estimates and the range around key parameters they wished to be analysed.

Final Analysis and Reporting

During the final phase of the project (February-March 2010) the parameter estimates provided by the WADA were used in a final model development phase. After incorporating a local length-weight relationship provided at the WADA workshop by Mr Glen Plummer the model was refitted to historic data a final time and the final analysis completed. In comparison to the preliminary report, which analysed a wide range of strategies unconstrained by logistical considerations, the final analysis is focused on comparing alternatives of the three year strategy proposed by Mr Gervis. The detail of the final model and analysis are contained in Appendix 4.

Results/Discussion

Literature Synthesis

Approximately 65 scientific articles were collected and surveyed during the course of the literature synthesis and are reviewed in Appendix 3. Only a brief overview of the most relevant issues is provided here.

Initial Evaluation of Feasibility

The body of international literature clearly indicates the technical feasibility of both reseeding and translocating abalone although the underlying cost-effectiveness of both approaches remains debatable.

The vast bulk of the world's experience and research on abalone reseeding comes from Japan where the government underwrites the annual costs of mass producing up to 40 million juvenile abalone of five species for planting out by fishing cooperatives (Kojima 1995; McCormick 2000). Seki and Taniguchi (2000) suggest that annual reseeding rates since the 1980s has been has high as an average 150 million per annum since 1980. The Japanese Government started the marine stock enhancement program in 1963 and abalone were one of the first targets in the program (Inoue 1976). The primary aim in Japan until recently has been to augment catches directly by growing out hatchery produced seed and then recapturing them in the fishery, rather than an aim of enhancing recruitment to natural populations (Seki and Taniguchi 2000). Hamasaki and Kitada (2008) summarize and analyze Japan's experience with reseeding, estimating mean cumulative recapture rates 4-5 years after release from 0.014 to 0.238. Rates thought to be influenced by fishing method, size of release, seed quality, release site and season, predators, prey and competitors and interactions between these factors (Howell et al 1999; Leber et al. 2004; Bell et al. 2005, Hamasaki and Kitada 2008). The essential technical feasibility of reseeding is demonstrated by the fact that across the various Prefectures seeded abalone comprise 7-84% of the abalone landed (Hamasaki and Kitada 2008).

In Japan reseeding had been expected to sustain and/or augment fishery production (Seki & Sano 1998), however, despite recent releases of 25-30 million juveniles per annum the annual catch of abalone declined from around 6,500t in 1970 to about 2,000t in recent years (Masuda & Tsukamoto 1998; Hamasaki & Kitada 2008). The declines have been greatest in the north of Japan where the reseeding has been most intense (Tegner 2000). This has lead to increasing discussion as to the cost effectiveness of Japanese stock enhancement programs for abalone (Masuda & Tsukamoto 1998; Seki & Sano 1998; Seki & Taniguchi 2000; Hamasaki & Kitada 2008). More recently the Japanese literature has also begun discussing the use of lower fishing pressure, habitat rehabilitation, translocation and reseeding as means of rebuilding breeding stocks, rather than the permanent outgrowing of hatchery production.

Outside Japan there has also been considerable research. In southern California, a wide range of stock enhancement studies have been conducted with the view to rehabilitating depleted stocks; involving releases of both hatchery produced larvae and juveniles, and the translocation of wild adults (Tegner 2000; Bell et al. 2005). In New Zealand, Australia and South Africa larval and juvenile releases have occurred at experimental scales with the aim of studying optimal release conditions, indicative survival rates and potential costs and benefits (Bell et al. 2005).

Recovery rates of seeded abalone in non-Japanese studies have generally been far lower than those observed in Japan. Some researchers (e.g. Tegner and Butler 1985; McCormick 1994) attribute this difference to lower predator numbers on Japanese fishing grounds but it seems likely that fundamental differences in the scientific designs of the studies also contributes to this result. Most non-Japanese studies have been conducted on small experimental scales, within limited research areas and over short time periods 1-2 years, which makes them prone to underestimating survival rates due to the potential of seeded abalone to move out of the area being searched, and the reseeded abalone still being cryptic and hard to find at the conclusion of the study, even if they remain in the study area. In contrast, many of the Japanese studies have been large scale releases onto heavily exploited fishing grounds and commercial catches have been monitored over many years until released cohorts have completely emerged from their cryptic habitat and been fished out.

Far fewer published studies on the translocation of adults have been published. The most substantial study is that by Saito (1979) who reported on a translocation program run from the 1950s through to the mid-1970s in 6 Prefectures around Funka Bay on the south east coast of Hokkaido, the northern most major island of Japan, just outside the natural range of abalone. The translocations involved a total of 4.72t abalone collected from crowded low growth potential areas (i.e. stunted stocks) on the south west coast of Hokkaido, and transplanted into an area of Funka Bay that was recognized as having good growth potential but no natural abalone stocks. As with the Japanese reseeding program, this translocation program seems to have been conceived primarily as a sea-ranching exercise rather than to establish a self-sustaining population. On average harvesting occurred two years after the transplantations began and recovery rates were estimated for the 5 Prefectures at 30.5, 22.8, 22.6, 25.8 and 2.3%. The lower recovery rates were attributed to the extreme eastern location of the newer grounds where conditions were not thought to be as favorable for the abalone. Recovery rates of 25-30% were considered uneconomic in terms of the on-growing exercise. However, Saito (1979) noted that the estimated rates of return took no account for the breeding that took place before the translocated abalone were re-harvested. Saito argued on the basis of length-frequency histograms showing juvenile size classes below the size of the translocated stock, and the magnitude of continuing catches, that the natural recruitment created by the translocated abalone had been considerable. Tegner (2000) concluded that Funka Bay is an example where transplantation of wild juveniles was successful in establishing a self-sustaining populations of *H. discus hannai* in an area that did not naturally support abalone.

Synthesis on Mortality Rates

The literature synthesized (Appendix 3) clearly shows that seeding and translocation incur some level of additional mortality above normal natural levels. However, using the assembled literature to estimate that additional level or mortality, or even a comparative difference in mortality rates between wild and reseeded juveniles is far from straight forward making the selection of an appropriate age schedule for the mortality rates of juvenile abalone under each of the three potential rehabilitation techniques challenging. This synopsis shows that the 'normal' rate of juvenile natural mortality in wild abalone populations has been rarely studied. Shepherd & Breen (1992) noted that there are many more studies of the mortality of hatchery seed transplanted into the wild than of the natural mortality rates of wild populations. However most of the studies that have been conducted worked with blacklip abalone, and together with some international studies they create a consistent if not concise picture. This body of information has been well summarized and discussed by Shepherd & Breen (1992) and Shepherd et al. (2000) and consequently the age related natural mortality curves outlined by those authors must hold some authority, and have provides some sort of baseline for the assumptions used in this study.

As discussed above rates of mortality amongst reseeded juveniles have been documented in two fundamentally different ways. In Japan recovery rates of released cohorts have been documented some years after release as the seeded cohort has moved through the fishery over several years. This provides an accurate estimate of overall recovery rate for each cohort, because the recovery processed continued until the entire cohort had emerged from the cryptic juvenile habitat and been captured. However this approach obscures the annual rate of mortality in each year because the individuals in each cohort are captured after and over several years. It must also be remembered that the relatively high recovery rates observed from the Japanese fishery are widely attributed in some part to the intense removal of abalone predators and competitors that occurs (Tegner and Butler 1985; McCormick et al 1994).

By way of contrast non-Japanese studies tend to have been much more limited in extent and time span, normally involving experimental scale releases, involving just 100s to a few thousand seeded individuals, and recovered by research divers after just one or two years. Theoretically the data from these studies should be easier to use in this exercise, as the time at liberty over which mortality has occurred is more easily determined and the rate of mortality more easily estimated. Unfortunately, it is often unclear as to what extent the low recovery rates recorded by the non-Japanese studies reflect actual mortality, or loss of individuals to view, or from study sites. Amongst the non-Japanese studies the most useful results are those that built in to their experimental design the estimation of the confounding factors (e.g. Dixon et al. 2006). Unfortunately these are rather few. Another approach has been to set up experimental habitats either within enclosures to contain the seed, or isolated and designed to be dismantled at the end of the study so that something approaching 100% of surviving juveniles could be recovered. The weakness of this third approach is the confounding influence of the experimental structure itself on the observed mortality rate (e.g. Schiel 1992).

From the perspective of this study it is unfortunate that there have been very few parallel comparative studies of both wild and reseeded mortality rates. The only parallel comparative study of wild and reseeded abalone survival rates Schiel (1992) observed the mortality rate of reseeded abalone during the first year of release to be about 40% higher than similarly sized wild *H. iris* placed in the same experimental enclosures.

Re-Analysis of Mortality Rates

The state of the literature and the needs of this modeling exercise led to a re-analysis of the extensive Japanese experience with the aim of converting those results into annualized rates that could be used in the model and compared to the non-Japanese results for wild stock mortality rates. Plots of the cumulative recovery rates of 49 cohorts of abalone released by 10 differing Prefectures presented by Hamasaki and Kitada (2008) were re-digitized to derive the recovery rate of each cohort. From this it was possible to calculate an arithmetic mean recovery rate of 0.128 and a geometric mean recovery rate of 0.124, the range being 0.002 to 0.5 (Appendix 3 - Figure 2). To make this useful in this analysis it was also necessary to estimate the correct length of time over which to apportion these survival rates. Hamasaki and Kitada (2008) observe that in Japan abalone have generally been released at about 30mm shell length and approximately 1 year of age and have mostly been recovered by the fishery from 1-4 years after release, and that cumulative captures have tended to reach a plateau at 4-5 years after release. Consequently they treated cumulative recapture rates at 4-5 years after release as the overall recapture rate. Kojima (1995) conveniently tabulated recovery profiles by years at liberty for 10 released cohorts starting with the 1980 cohort. By Hamasaki and Kitada's (2008) criteria six of these cohorts were fully recovered by the time Kojima compiled his figures. For this study the numbers in table 3 of Kojima (1995) were converted in to the proportion of each cohort recovered by age of the cohort (Appendix 3 - Figure 3) and used to estimate a mean time at liberty for these cohorts, which was calculated as 3.16 years. From this the best estimate of survival of reseeded abalone in these Japanese programs is estimated to be approximately 12.5% over about 3 years at liberty. This estimate of a mean recovery rate for reseeded abalone is placed into context by Seki and Taniguchi (2000) who use the assumption of 10% survival to estimate the number of seed needed in Miyagi Prefecture in northern Japan to achieve and eventual stock density of 1 abalone/m^2.

Model Assumptions about Mortality Rates

On the basis of the synthesis and re-analysis presented in Appendix 3, the assumptions used for this modeling exercise were:

1. That in the Western Zone blacklip abalone would be seeded at around 20-30mm and at about 1 year of age, similar to the Japanese practice.
2. That in the first year after release the reseeded juveniles will have approximately 40% higher mortality rates that wild abalone in their second year.
3. That in subsequent years both wild and reseeded juveniles experience the same mortality / survival rates.
4. That juvenile and adult survival rates should be consistent with the literature.

On this basis the base case model incorporated the mortality schedules shown in Table 1. The schedule reproduces the average survival rates observed across the Japanese studies (about 12.5% at 4 years of age) and the 40% survival differential observed by Schiel (1992). This schedule does imply slightly higher survival rates for wild stock abalone than has been observed across the various studies reviewed by Shepherd and Breen (1992).

Year	Wild Mortality Rate	Wild Survival from age 1	Seed Mortality Rate	Seed Survival from age 1
0	0.9			

1	0.6		0.70	
2	0.4	0.5	0.40	0.3
3	0.3	0.3	0.30	0.18
4	0.2	0.21	0.20	0.126
5	0.15	0.168	0.15	0.1008

Table 1. The mortality and survival schedule used in the Basic Population model.

Cost-Benefit Analysis

Detailed results of the analysis conducted during the final phases of analysis can be found in Appendices 4. Only a precise of the main results are represented here.

A clear result that comes from both the preliminary and final analysis (Appendix 4) is the importance of matching the scale of rehabilitation to the scale and productivity of the stock. These analyses clearly show that early interventions scaled to move the stock up the steep part of the SRR curve toward B_{MSY} are likely to be more profitable than larger, later interventions. In the final analysis the base case reseeding strategy of reseeding 500,000 seed per annum for three years and the comparative translocation strategy involving 27,000 adult abalone per annum for 3 years were both estimated to be relatively efficient strategies given the scale of the resource being rehabilitated. With abalone worth $25/kg and seed 40c/piece these strategies were estimated to cost $958,929 and $931,500 respectively.

Interestingly both analyses also suggest that there is very little difference in cost effectiveness between the two rehabilitation techniques. The descriptive analysis which ignored the cost of capital and simply estimated costs of rehabilitation and comparative expected income through the period 2011-2025 suggested that across the range of pricing scenarios used the translocation strategies were more likely to produce a level of income in excess to that expected under a Natural Rebuild, and unlikely to incur the large losses or incomes, predicted under some reseeding and pricing scenarios. Ignoring the cost of capital and simply accounting for stock productivity and costs, figures 3 and 6 of Appendix 4 suggest that unadjusted returns from reseeding should exceed costs if the price of abalone approaches $40/kg. While most translocation scenarios should return incomes in excess of costs at any of the abalone prices considered.

The main difference between reseeding and translocation strategies in this respect results from differentials between the price of seed and abalone. Thus translocation scenarios incorporated the price of abalone as a basis for costing the rehabilitation and estimating expected incomes, buffering the eventual result. In contrast the range of reseeding scenarios estimated costs on the basis of the cost of seed, and income on the value of abalone, thus scenarios incorporating a low price of seed and high price of abalone were likely to produce high estimated incomes, and high seed cost / low abalone price scenarios could produce high estimated losses.

That being said the major result from the final analysis is that taking into account the cost of capital none of the rehabilitation strategies analysed appear to be cost effective (Appendix 4 Table 2-6). Taking into consideration the actual cost of capital, all the scenarios analysed in detail produce lower returns than the Natural Rebuild scenario. This is a similar, although more extreme, result to that produced by the preliminary analysis which found that only if the highest discount rate was assumed (0.07) were some reseeding and translocation scenarios likely to be profitable. However, while the estimated time taken to recover to B_{MSY} was similar for both models and analyses, in the preliminary analysis the estimated costing of the rehabilitation strategies only accounted for the cost of the translocated abalone or seed which was assumed to be low, and unlike the final analysis did not account for the cost of outplanting.

In the context of analyses suggesting that measured interventions that moved the stock up the SRR towards BMSY were more profitable, the model was also used to examine its sensitivity to the timing of the rehabilitation. Given the context of WADA's discussion, previous rehabilitation scenarios has assumed they commenced in 2010 three years into the third year of Natural Rebuilding, and 2010 was assumed as the zero point for calculating the effect of economic discount rates. In this final sensitivity analysis (Appendix 4 - Table 7), the rehabilitation strategy was assumed to have commenced in 2008 the year following the modeled viral impact. In these final scenarios 2008 was also assumed to be the zero point for applying discount rates. The results of this last analysis suggest rehabilitation will be more cost-effective if commenced early (2008) so that the stock is more quickly moved towards a productive level of breeding biomass. This is particularly true for the translocation strategy which adds directly to the breeding biomass. However, the model still estimated that compared to the Natural Rebuild strategy, both forms of rehabilitation were not cost-effective enough to cover the cost of the capital involved.

Depensatory Effects

The model used in the final analysis suggests that natural rebuild scenario recovers the stock to B_{MSY} by 2017 while only the translocation scenario with 1% mortality was capable of recovering the stock to this level before 2015. The estimated difference in timing between Natural Rebuild and rehabilitation is too small to pay for the real cost of the rehabilitation strategies. This outcome is driven entirely by the resilience of the assumed Stock Recruitment Relationship (SRR) at low biomass levels.

The standard form of the Stock Recruitment Relationship (SRR) used in Australian stock assessment was used in this analysis. The two most important features of this standard parameterization of the SRR for this analysis are:

A. The density dependence mechanism acts between adult stock and the recruiting year class, rather than as competition for crevice spaces by juvenile year classes.
B. The spawning biomass becomes increasingly effective per unit of remaining biomass as it approaches zero levels.

Thus in incorporating this form of the SRR the model is built to predict that the resource is rebuilding rapidly after the virus attack and consequently that investing in either seed or translocation will produce relatively little extra benefit. The same accepted SRR was applied to all scenarios so that within the terms of this study no treatment is unfairly advantaged because uniform assumptions were used. However, while consistent with the stock dynamics modeled more formally with the Australian Abalone model structure, the explanatory form of analysis performed here does not exclude other underlying and as yet unobserved dynamics. Almost nothing is known about the actual SRR for abalone anywhere in the world (Campbell 2000) because only a few experimental SRR studies have been performed by Shepherd and Partington (1995) in South Australia with greenlip abalone, and the non commercial species *H. scalaris*, and no studies have occurred anywhere in the world on any species, at the scale of a reef code like Kilarney.

In this analysis the assumed resilience at low biomass becomes problematic in driving the results of the analysis so strongly, because it is in stark contrast with the belief of many abalone ecologists who believe that at low biomass levels abalone populations become subject to depensatory effects which reduce the effectiveness of spawning causing lower than expected recruitment from the remaining biomass (Liermann & Hilborn 1997). Tegner et al. (1989) was the first to touch on "reduced fertilization efficiency" as one of a number of possible explanations for the observed decline in Californian landings. Although several Australian studies have become the mainstay of this concern (Shepherd & Brown 1993, McShane 1995, Shepherd & Partington 1995, Babcock and Keesing 1999). Two primary mechanisms are proposed for depensation at low biomass levels.

1. Some abalone ecologists (Shepherd & Brown 1993, McShane 1995, Babcock and Keesing 1999) emphasize the potential for low fertilization success in sparse fished populations where aggregations are fished before they can spawn reducing the density at which eggs and sperm are mixed in the water column during spawning reducing fertilization success.

2. The other school of concern (Miner et al. 2006) involves the conditioning of clean coralline algal (CCA) substrates for the successful settlement of abalone larvae, through the grazing pressure of adult abalone and other demersal herbivores. When abalone populations are rapidly depleted much of the coralline algal surfaces previously maintained through grazing pressure maybe overgrown over month and years by fleshier algal and sponge biota, reducing the availability of settlement sites for juvenile abalone.

The extent to which these factors result in depensation and alter the assumed shape of the abalone SRR curve remains uncertain. Some studies (Shepherd 1986; Prince 1992; Shepherd & Partington 1995) emphasize the mobility of individual abalone in and out of aggregation and the ability to actively reform breeding aggregations if left undisturbed, or fished lightly enough, suggesting that unless heavy fishing is allowed through the spawning period, abalone should be able to aggregate sufficiently to ensure optimal fertilization. While the CCA settlement surfaces are known to be created and maintained by the grazing of many benthic herbivores including abalone (Morse et al. 1979) so that growing aggregations of abalone might be expected to reestablish settlement surfaces around themselves as their aggregations re-grow and need more settlement surfaces. In fact, Shepherd & Partington (1995) argue that it is the interplay of these effects that creates the standard form of the Ricker SRR that they described and which at low biomass is similar in shape to that used in this study.

To the extent depensation changes the shape of the abalone SRR it might be expected to have a flatter approach to the origin than suggested by the SRR used in this study. If this were the case the SRR curve would dip down low at some critical level of breeding biomass above zero. Below that threshold level of Breeding Biomass the productivity of each unit of biomass would be depressed and the recovery of the stock would be very much slower than suggested by the SRR curve used here. Incorporating a depensatory effect into the SRR curve would disproportionately reward any strategy that enhances the stock's own natural productivity by rapidly rebuilding the breeding biomass to above that threshold level. So compared to the analysis presented here the Natural Rebuild strategy would be penalized if it left a stock below the threshold at unproductive levels.

The difficulty for this study is that while depensatory effects are surmised, there has been so little research on this aspect of abalone fisheries ecology that there is no basis for deciding what level of biomass would constitute the threshold, nor of the reduced level of productivity that might occur below that threshold. Consequently attempting to capture this aspect of abalone ecology in this modeling exercise would necessitate incorporating a wide range of assumptions that have no defendable basis. In that case it is likely that built in assumptions would completely overwhelm the 'scientific data' incorporated into the model. As unsatisfactory as this maybe; the only possible choice at this junction is to acknowledge that this factor is in no way incorporated into the analysis. This is a necessary weakness of the analysis and as a consequence caveats need to be placed upon its results. To the extent that depensatory effects exist for abalone at Kilarney this analysis will tend to favor the cost effectiveness of naturally rebuilding the stock, over the alternative interventions. But without understanding the phenomena better we cannot robustly opine whether this effect is significant or not.

In this context comments made by divers who have been observing the virus impacted areas recover at the 13 August, 2009 WADA workshop were of great interest. Several of the divers had originally in 2006 been supportive of rehabilitation for the most heavily impacted areas because they like most involved with the issue could not imagine these areas of stock recovering naturally. At that time most people were implicitly expecting some form of depensatory effect. So it was of great interest that at the WADA workshop on 13 August, 2009 those same divers reported that in their opinion even the most depleted areas were now visibly recovering, and that because of that they were no longer convinced intervention was needed. This may be anecdotal evidence that the assumed form of the SRR may be more applicable than feared.

Comparative Cost-Benefit Analyses

Finally, we consider where these results fall within the context of similar studies.

On the basis of his trials of reseeding *H. iris* in New Zealand, Schiel (1992) estimated that rates of return would have varied from -100% to 43.7%. Only 3 sites were estimated to have been likely to produce a positive economic yield while five sites would have produced a negative return. Averaging across his sites he calculated the financial yield would have been likely to be about 9.6% greater than the expected yield on the same investment at 5% interest.

Using Japanese estimates of growth and survival for *H. discus hannai* Seki & Sano (1998) modeled the cost of re-establishing a 12t breeding stock comprised of 100,000 x 96mm adults. They estimated that depending on whether 1, 2, 3 or 4 year old seed were used this would take 1.9, 1.1, 0.6 or 0.35 million individuals cost, respectively, 67, 211, 304 or 385 million yen. A crude conversion of these estimated costs at 100 yen/$AUD converts these estimated costs respectively to $AUD 0.67, 2.11, 3.04 and 3.85 million. Perhaps unsurprisingly, as Japanese estimates of survival have been used in this study, these costings are similar in magnitude to those derived through this study. On the basis of their analysis, Seki & Sano (1998) argued that using sea ranching abalone was uneconomic and that reseeding should not be used to support catches. Rather they argued that a broader ecological perspective is needed and that Japan should consider using reseeding to re-establish breeding stocks, which should then be protected within the fishery by fisheries management.

In contrast to the results of the two above studies, which are generally comparable to the results of this study, Roberts et al. (2007) estimated returns from reseeding 10mm *H. iris* ranging from 7% to 41% per annum. Which they considered compared favorably with opportunity costs of about 10% per annum. However their estimated rates of return assumed 3 years at liberty beyond the 1-2 years they studied seed survival over and assumed M=0.1 for 3+ and older animals during those years.

Legislative constraints, translocation protocols, management advice and timeliness in the BCA assessment

Any proposals for translocation, restocking or reseeding of abalone would involve a comprehensive ecological risk assessment and extensive consultation in accordance with Victorian legislation and the *Guidelines for Assessing Translocations of Live Aquatic Organisms in Victoria.*

All applications for the translocation of live organisms are required to be submitted to the Translocation Evaluation Panel (TEP). The TEP requires a full environmental impact assessment be done for any new type of stocking activity that does not fall into any already established protocol for translocation. Translocation, reseeding or restocking will therefore require a full scale risk assessment. If trials are cost effective and can be conducted with acceptably low ecological risks, then Fisheries Victoria will investigate the development of specific translocation protocols for these resource enhancement programs.

Benefits

The results of the preliminary analysis has already been conveyed through a Preliminary Report that was presented to the 13 August, 2003 workshop for WADA members, as will this final report which will also be spoken to during WADA meetings during 2010. The benefit derived from this work is that WADA's ongoing discussion of the options for rehabilitating reefs severely impacted by the AVG infection can been informed by the application of this best available science and analysis. This should allow WADA to better inform their discussions of the issues and manage their decision-making processes. Beyond the immediate benefit to WADA this analysis can also inform VADA, in the Victorian Central Zone which is beginning to confront similar issues on their westernmost reefs which have also suffered AVG infections. These are the benefits identified in the original application.

This project is the first time a cost-benefit analysis of this type for abalone has been adapted to incorporate normal stock building processes into the comparison. This maybe the first time it has been done for any marine species. The novelty of this approach may produce a wider unforeseen benefit of stimulating parallel analyses of this type.

Further Development

From the synthesis of the literature this analysis appears to be the first time fully specified population models have been applied to the analysis of the costs and benefits of rehabilitating abalone stocks, consequently wider dissemination of the results could usefully inform the international community of researchers, managers, fishers and mariculturists working with abalone. This could be achieved within Australia through presentation of the results at an Australian Abalone Conference and internationally by presentation to the 4th International Symposium on Stock Enhancement and Sea Ranching planned for Shanghai Ocean University, China 21-25 April 2011.

This novel approach of incorporating a fully specified population model into the cost benefit analysis also creates further research potential. Drs Sarah Jennings and Satoshi Yamazaki (UTAS) have expressed an interest in working further with the model as a generic tool to study the optimization of rehabilitation strategies.

The results of this analysis also highlight the critical gap in our knowledge concerning the relationship between adult stock levels and future recruitment into the population. As discussed in several places through this document, the assumed shape of the stock-recruitment relationship (SRR) basically precludes the analysis from finding that any form of pro-active rehabilitation can be cost effective. However, there is considerable discussion in the literature as to whether this assumed form accurately describes recruitment dynamics at low stock levels. There is probably little benefit to be gained from elaborating on the approach taken here until this fundamental uncertainty has been resolved. Should depensation prove to be a significant factor in determining the effectiveness of abalone recruitment at low biomass levels, these results may well prove to be flawed. Likewise, with the move towards reef-scale management strategies there is growing interest in knowing at higher adult densities, what level of stock optimizes the long-term productivity of reefs. Based on this gap in our knowledge long term studies should be established on a series of representative reefs to study the nature of the relationship between adult stocks and future recruitment. In this respect, the AVG infection might be considered to be presenting an unparalleled opportunity for investigating the nature of the SRR at densities lower than normally seen within the fishery.

In terms of further development it should also be noted that comparative studies of the survival of seeded and natural juveniles is a virtual missing link in this field. This study could only locate the single study which released wild raised and seed abalone into the same experiment and compared their rate of survival. Further studies of this type would enable the existing body of literature to be more accurately interpreted, and illuminate future studies of this type.

Planned outcomes

This project's primary outputs are the synthesis of the relevant scientific literature (Appendix 3), and the modeled analysis documented in Appendix 4 along with audio-visual presentations of the analyses' results that have already, and will be, made to WADA meetings, and other meetings as the opportunities arise. These outputs have already informed WADA members of the scientific evidence available to evaluate the issue and considerable comment was received at the 13 August 2009 workshop as to how useful WADA members had found that material. Likewise comment from WADA members indicated that even the preliminary results of the analyses were useful in putting their discussions about future options for pro-active rehabilitation on to a shared, sound factual basis.

Conclusion

All the projects objectives were achieved.

Current and past research related to abalone reseeding, translocation and mortality rates has been surveyed, synthesized and re-analyzed (Appendix 3). The scientific literature shows that in Japan at least augmenting abalone stocks both by seeding juveniles and translocating adult is technically feasible. The literature from other parts of the world is more equivocal, but it is not possible to determine whether this difference is due to lower predator levels in Japan as claimed by some, or biases associated with the experimental designs employed in other countries. Only in Japan have long term, large scale augmentation programs been attempted and fishery wide returns monitored until the augmented year classes have been fished out. Outside Japan studies have been small scale, short-term experiments, so that recapture rates have been depressed by the cryptic nature of juvenile abalone, and movement out of research areas.

A quantitative model was constructed of the population dynamics and economic processes underlying reseeding, translocation and natural rebuilding of a blacklip abalone reef in the Western Zone of Victoria, and used to analyze the likely costs and benefits of the alternative techniques of rehabilitating the Kilarney reef code in western Victoria (Appendices 4). This appears to be the first time a fully specified population model has been used to analyse the biological and economic processes underlying the rehabilitation of abalone reefs. To achieve this estimates of age related juvenile survival rates had to be developed based on the literature synthesis and a re-analysis of the Japanese data, and used to recalculate the stock recruitment relationship used in the Australian Abalone Model.

Written (Appendices 4) and audio-visual reports, detailing modelling methods, underlying assumptions results, were prepared and presented to WADA members at their August 2009 workshop and further presentations will be made during 2010.

The depleted stock was modeled by default as being highly productive at low level levels and currently rebuilding back towards BMSY levels which it is predicted to achieve without catches by 2017. The translocations and reseeding scenarios starting in 2010 were estimated to achieve this level of rebuild by 2015. Taking into account the cost of capital, the natural rebuild scenario wins out in all the scenarios considered because there are no costs associated with it. In that context the two techniques, translocation of adults or reseeding were similar in effectiveness. Across the scenarios modeled in the final analysis, translocation was estimated to never fare too badly, always at least paying for direct costs, but not covering the cost of capital, while reseeding only covered direct costs when the price of abalone exceeded $40/kg and seed is cheap. However, none of the intervention scenarios produced any return when the cost of capital was considered.

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Appendix 1: Intellectual Property

The intellectual property developed through this project is for general publication.

Appendix 2: Staff

Dr Jeremy Prince was the principal researcher funded by this project. Mr Mark Gervis (Southern Ocean Mariculture) and Drs Sarah Jennings and Satoshi Yamazaki of University of Tasmania consulted to the project providing detailed costings and useful advice.

Appendix 3: Literature Synthesis

Introduction

This study proceeds from the premise of using a simple age-structured population model to analyze the likely comparative costs and benefits of letting a reef code like Kilarney that has been heavily impacted by the AVG recover be either rebuilding through natural processes or be reseeding with juveniles, or the translocation of adults into the area. Since the 1970s a broad body of scientific literature has been published which is relevant to this subject and which can be used to inform this analysis. However, while establishing the technical feasibility of the techniques involved the literature synthesized below, in most cases leaves the essential question of cost effectiveness under active discussion creating the need for a synthesis and analysis such as this.

From the perspective of this analysis a critical aspect of abalone ecology concerns the rates of mortality that can be expected in wild, reseeded and translocated abalone. It is immediately obvious that assumptions about these mortality rates, and in particular any difference between hatchery produced and wild juvenile abalone will have a major influence on the estimated cost of each treatment. Perhaps less obviously the rate of ongoing natural mortality will also determine the length of benefit obtained from translocating adults, and from growing reseeded juveniles through to maturity. It is for this reason that the major focus of this literature synopsis is the estimation of mortality rates in reseeded and wild stocks of abalone, although, this narrow focus does not preclude notes being made on a range of lesser but still important issues for the cost benefit analysis; i.e. comparative costings.

The synthesis begins with an overview of the broader topic before working systematically through the literature on studies of natural mortality in wild abalone stocks, the mortality rates of seeded abalone, before providing brief overviews of larval reseeding and translocation. In the final section of this synthesis the body of literature discussed is used to develop and justify the base case mortality rates used in the cost benefit analysis (Appendices 4).

Overview of Studies of Survival and Mortality in Abalone

In their review of rates of mortality in abalone Shepherd and Breen (1992) observed this is very poorly known aspect of abalone population biology. A fact they attributed to it being one of the most difficult areas of abalone study. This is because of the turbulent marine environment abalone live in and the fact that the basic techniques available are imprecise, subject to a range of known but difficult to quantify biases, and require long time frames.

Mortality rates are studied by monitoring a range of proxy measurements.

Early studies of wild stocks were generally based on the disappearance of tagged individuals from a population, or the analysis of length data using an assumed growth curves. Reseeding studies have tended to either release marked juveniles and then conduct diver searches for them over the following 1-2 years, or alternatively monitor returns from a fishery over 6-8 years. Mortality rates can also be estimated from the rate at which empty shells accumulate, and sometimes on the basis of growth rings.

Each technique has its advantages from ease of application to greater accuracy, but all also have weaknesses. Previous reviewers of this topic (i.e. Shepherd and Breen 1992) make much of the variability observed between differing studies on this topic concluding that mortality rates are generally higher in warmer waters but that predator abundance, which they claim tends to be higher in warmer waters, is the causal factor. They may be correct in this analysis, but to some extent this variability they describe is explained by the biases of the different techniques used in each area. In other words the data we have contain a lot of observational error. Some of the variation is undoubtedly a reflection of our 'noisy' estimates rather than reality varying.

Juvenile abalone are known to be cryptic and avoid direct sunlight, emerging during the process of maturation (Prince et al. 1988; Heasman et al. 2004; Dixon et al. 2006). Released juveniles move into the substrate upon release (Shepherd et al. 2000) and survivors can be initially almost impossible to find but slowly emerge over time (Dixon et al. 2006). Dixon et al. (2006) estimated that this cryptic behavior meant that 25% of the abalone present in their studies were not sighted during surveys that pulled apart habitats constructed purposefully to facilitate searching. Many studies observe that recovery rates of juveniles increase with time over 1-3 years. Estimates of diver efficiency in detecting released abalone in cryptic habitat range from 13% to 80% (Tanaka et al. 1991; Preece et al. 1997; Shepherd 1998; Seki & Taniaguchi 2000; Heasman et al. 2004; Dixon et al. 2006).

This behavior biases in differing ways a range of techniques for estimating abalone survival. Short term studies of the survival of reseeded abalone based on searching once or twice for survivors 0-3 years after release inevitably under-estimate survival because the researchers fail to find all the survivors. However any techniques based on tracking the abundance of a cohort over time is likely to over estimate survival because the abalone become more easily found over time as they become less cryptic. Depending on their size abalone that are released as seed, translocated breeding stock or in tagging studies are capable of moving 1 - 100s of meters over days, months and years (Shepherd 1986, Ault 1987, Prince 1992, Tarr 1995). This means that depending on the scale of the experimental field of search abalone may move out of the research area and be lost to a study without dying, and reduced counts of recovered abalone (Shepherd 1998; Heasman et al. 2004). Thus monitoring survival within a limited field of search can only provide a minimum estimate of survival. Both experimental reseeding studies, and tagging studies can be influenced by movement of abalone out of the research area reducing recapture rates and causing survival to be under estimated.

In a similar way dead shell can be transported out of a field of search by predators and wave action, or buried and lost to view. Over time they break down and are permanently lost to a searching diver. So shell based estimates can only provide an estimate of maximum survival.

Likewise any type of unmeasured loss of tags from a tagging study adds to the estimate of mortality causing natural survival rates to be under estimated. Common unmeasured sources of tag loss can include an initial loss of tags associated with the tag and release process, mortality resulting from tagging, ongoing loss of tags over time as abalone repeatedly wedge themselves back into crevices after grazing, and the poaching, or legal fishers not reporting or not seeing tags in abalone they have collected.

The logistical need to limit the field of search in these studies, the limited effectiveness of searching complex bottom, undetected tag loss, and the ecology of abalone conspire to reduce many experimental estimates of survival and mortality to being relatively crude minimum or maximum estimates. The best of these experimental studies (Shepherd 1990,1998; Seki and Taniguchi 2000; Dixon et al. 2006) have used an overlaying of techniques for estimating abundance, sighting probability and movement providing means of taking some account for these various factors, but the majority of the published work have not.

In terms of avoiding the sources of observational error discussed above the best studies are the large-scale Japanese reseeding studies, where recovery rates of reseeded cohorts have been estimated by monitoring the quantity of reseeded abalone in successive years of the commercial catches over the following 2-8 years. In these studies the entire reef area is fished and cohorts are fished to exhaustion over several years as they progressively emerge and become vulnerable to collection. Thus these estimates are not influenced by movement from the field of search, the cryptic behavior of abalone, or unquantified tag loss. This is not to say interpreting the relevance of Japanese results for western Victoria is entirely straight forward. These Japanese estimates tend to be multi-year estimates, while the rest of the body of work in this field discusses, and the model will require, annual or instantaneous mortality /survival rates.

Probably a more difficult factor to take account of in this context is the widely held belief that the relatively high recovery rates observed from Japanese fishery are in some part due to the active removal of abalone predators and competitors that also occurs on Japanese fishing grounds (Tegner and Butler 1985; McCormick et al 1994). This observation goes to the validity of basing assumptions about survival of abalone in western Victoria on those experienced in Japan.

A final note of context is required with regard to how rates of mortality and survival are cited in the literature. Some of the following is somewhat technical, and some readers may prefer to skip this section and simply read the concluding clarification about the two differing ways mortality and survival rates get referred to in the literature.

Technically speaking (Gulland 1977) and using the subscript ‘t’ and ‘t+1’ to denote the passage of one unit of time:

Eq. 1: $\text{Survival} = N_{t+1}/N_t$

In plain language survival is the number (N) remaining at ‘t+1’ (N_{t+1}) divided by the original number (N) at time ‘t’ (N_t). Mortality is the inverse of survival so

Eq. 2: $\text{Mortality} = 1 - \text{Survival}.$

By definition this calculation has the time step t+1 and so the estimated proportion is a rate over time. Many studies cite this proportion as an annual proportion of abalone expected to die or survive and this is probably the easiest way to conceptualize the topic. The model being used is a simple spreadsheet model based on annual steps and so it uses an estimate in this form; the proportion surviving each year of life. In this report I will extensively use this annual proportion of abalone either surviving or dying and refer to it as the ‘annual rate’ of survival or mortality. In cited studies this simple rate of mortality may also occur over the differing time periods of their studies.

However, given the phenomena being described is a continuous function and may be applied over any period of time, much of the literature use the more technically correct form and cite the instantaneous mortality rates. Where the instantaneous rate of total mortality (Z) is estimated as:

$Z = -\log_e(N_{t+1}/N_t)$

Using instantaneous rates of mortality

Survival = e^{-Z}

In fisheries ecology the instantaneous rate of total mortality (Z) is commonly conceptualized as being comprised of F, the instantaneous rate of fishing mortality, and M the instantaneous rate of natural mortality. So that $Z = F+M$. However in many cases the studies dealt with here are dealing with unfished populations, or parts of populations and so for all intents and purpose $Z = M$.

The take home message from this discussion is that this report will refer extensively to the ‘annual rate’ of survival or mortality and this will refer to the simple annual proportion of abalone which either survive or die, and will vary between 0 and 1.0. To make the language even plainer I will tend to cite these values as the annual percentage (0-100%) surviving or dying.

It should also be noted that some cited studies use this simple proportional rate of mortality and survival calculated over the length of their experiment (days – years). Within the context of those studies those rates may be reported over their specified time period, however in the body of the report the aim will be to convert the estimates derived from those studies back to simple proportions surviving or dying over an annual time span.

This report will also cite or quote ‘instantaneous rates’ of mortality which may be denoted with the symbol ‘M’. This is a more mathematically correct reference to the annual rate but calculated as a logarithmic function. At low levels (0.1-0.2) this logarithmic rate is superficially similar to the simple annual rate, however the logarithmic reaches values >2 as the simple mortality rate approaches 90%. While citing and discussing these logarithmic rates this report will also provide conversions across to the simple annual proportional rates that will eventually be used in the model framework.

The Literature on Natural Rates of Mortality in Wild Stock of Abalone

A dogma became established early in abalone science that natural juvenile survival rates were relatively high, similar to adult survival rates (e.g. Tegner, Shepherd). This view can now be seen to have been based on insufficient research as the body of evidence shows mortality rates decline with age and size amongst the cryptic juvenile age classes before stabilizing at lower levels amongst the emergent adult population age classes (Shepherd & Breen 1992).

To analyze the population dynamics of reseeding, translocation and natural rebuilding it is necessary to model juvenile mortality rates as accurately as possible and so in this synopsis differing age categories (post settlement, juveniles, adults) will be dealt with separately. While strictly outside the scope of this study, some information on post-settlement survival has been collected because it provides context for qualitatively evaluating the potential for larval reseeding which is also sometimes discussed by WADA members.

Post-Settlement Mortality Rates

Haliotis rubra

McShane (1991) described monitoring the numbers of newly settled and juvenile (<1 year old) on Crustose Coraline Algal (CCA) settlement surfaces in three populations of *H. rubra* off north-eastern Victoria with a venturi powered underwater vacuum cleaner with a brush on its nozzle. He observed rapid decreases in the density of newly settled abalone and found a linear relationship between the survival of 5 month old juveniles, and the log of the density (range 9 to 2000/m^2) of newly settled abalone (approx. 1 week old). He concluded that survival over the first 5 mo. was density dependent, and likely to be less than 10% annual instantaneous rates (M) of 6.6 – 10.2 were estimated.

While clearly indicative of the low survival of abalone in their first months of life McShane's study may be over estimating M for these young animals because his search focused just on the CCA surfaces. More recently researchers have observed movement away from the CCA surfaces as early as 1 month and <1 mm (Heasman et al. 2004) and that this movement is density dependent. So it is not obvious that McShane can totally exclude the effect of movement of the small juveniles off the coralline surfaces into poorly sampled cryptic habitat, so these estimates may be something of a maximum estimate.

Other Species

Through direct observation searching CCA surfaces with the aid of underwater magnifying glasses Shepherd (1987) cited in Shepherd and Breen (1992) observed instantaneous mortality annual rates (M) of 3.5 – 4.6 for greenlip (*H. laevigata*) abalone at West Island during their first 6 months of life. Expressed as the simple annual proportion this would represent just 1-3% survival after a full year. These could be biased high by the same potential bias of juveniles moving away from CCA surfaces into crevice habitat.

Juvenile Mortality Rates

Shepherd & Breen (1992) noted that studies of wild juvenile mortality rates were rare. A decade later Shepherd et al. (2000) could find only 4 published studies on the natural mortality rates for juveniles 6 months to 3 years old.

Haliotis rubra

Day and Leorke (1986) cited in Shepherd and Breen (1992) used tag and release methods to estimate M for 0.5 to 1.5 year old blacklip abalone in Port Philip Bay at 0.91, and for 0.81 for 2 year olds, equivalent to annual survival rates of 40% and 50%.

Prince et al. (1988) tracked the progression of juvenile modal length in length frequency histograms over 18 months to derive estimates of M for the various modes which declined from around 1.4 (25% annual survival) for the cohort with a mean age <1 year old, 0.7 – 0.9 for the 1-3 year old cohorts (40-50% annual survival), and around 0.2 for the older cohorts (82% annual survival).

Shepherd and Breen (1992) derive an estimate of 0.42 (65% annual survival) for 2-4 year old blacklip abalone in Waterloo Bay from a subset of some tagging data Shepherd had gathered earlier.

Other Species

After converting length composition to age composition data Tutschulte (1976) used both a life-table method and a comparison of the same population between two years to calculate an M of 0.70 for juvenile (1-7 year old) *H. corrugata* and 0.53 for 2-7 year old *H. fulgens* (50% and 59% annual survival).

Olsen (1984) cited in Sloan and Breen (1988) with few other details, estimated M for 1-3 year old *H. kamtschatkana* at 0.36 (70% annual survival).

Shepherd and Godoy (1989) estimated M for juveniles (1-3 year old) *H. scalaris* at 1.1 (33% annual survival) by tagging and movement studies to account for movement out of the study site.

Hines and Pearse (1982) estimated mortality rates of 2+ and older *H. rufescens* under heavy otter predation in Monterey Bay by mapping their positions and recording their disappearance and from the rate at which they collected empty shells to derive estimates of instantaneous rates of juvenile mortality (M) that ranged between 0.3 and 1.0 (37-74% annual survival).

Clavier and Richard (1985) used tagging studies and the analysis of length to estimate juvenile instantaneous mortality rates for *H. tuberculata* falling from 1.7 (18% annual survival) for the 0+ age class, to 0.26 (77% annual survival) in 1-2 year old age classes and 0.14 (87% annual survival) for the 4+ age class.

Shepherd (1987) cited with little detail, in Shepherd and Breen (1992) observed an average instantaneous mortality rate 0.64 (53% annual survival) for greenlip (*H. laevigata*) abalone 6 months to 2.5 years of age.

Shepherd (1998) describes monitoring cohort strengths and rates at which tagged abalone are lost from a research area from 1983 to 1997 tracking rates of mortality for juvenile (2 months to 4 years old) cohorts of greenlip (*H. laevigata*) and scalaris (*H. scalaris*) over that time period. The authors followed a similar design to that of Beinssen and Powell (1979) which released the tagged abalone within a grid and then tracked movements within the grid with the aim of estimating the loss of tagged abalone from the field of search. They also analysed changes in cohort strength over time to provide an alternative measure of mortality.

Instantaneous mortality rate M from 8 months to 4 years ranged from 0.2 to 3.2 per year (82-4% annual survival). M was consistently higher for greenlip, which the author attributes to the difference in the sculpting of the shells. Proposing that the ridges on the back of Scalaris provide protection against crabs and borers. Large changes in mortality rates were correlated for both species, which Shepherd suggested was due to changes in the abundance of predators common to both species, mainly wrasse and stingrays. With the tagging studies the study estimated instantaneous juvenile (1-3 years old) mortality rates (M) to be as high as 0.56 to 0.86 per year (57-42% annual survival). These may still be something of an overestimate as movement from the area may still have been underestimated. Monitoring cohort strength produced estimates of M around 0.40 – 0.45 (64-67% annual survival) these may provide the other bound because studies of cohort strength are normally biased to some extent by the emergence over time from cryptic habitats increasing estimates of survival.

Adult Mortality Rates

Haliotis rubra

Beinssen and Powell (1979) conducted an erudite study in which they released three batches of about 625 tagged adult blacklip abalone (*H. rubra*) into a grid within an unfished area and then over 790 days observed movement through and out of the grid along with the decline in numbers of tagged abalone. Using estimates of movement out of the grid to accounting for loss of tags due to movement they estimated the rate of natural mortality at 0.2 (82% annual survival).

Shepherd et al. (1982) derived estimates from tagging 2-6 year old blacklip abalone at Tiparra reef of M=0.21 and at West Island of 0.36 (70% annual survival) for 2-5 year old blacklip abalone. These early tagging studies focused on the emergent population and while nominally including juvenile year classes (2-4 year olds) would have been dominated by the more adult 5 and 6 year old abalone.

Shepherd and Breen (1992) cite unpublished tagging data collected for several populations of adult blacklip abalone in NSW which produced estimates of M=0.7 (50% annual survival). The studies referred to were conducted by Mr Gary Hamer of NSW Fisheries he was of the view that his recapture rates were influenced by a degree of unrecorded fishing.

Prince (1989) conducted a long term multiple tag and release program on the emergent adults (principally 7-9 years of age) in a reportedly unfished population of blacklip abalone on George III Rock in south-eastern Tasmania. Some 7,537 abalone were released over 24 months into a 1km^2 area isolated by large areas of sand. Four local divers monitored by researchers fished down the area 28 months after the first releases and 2,503 abalone were recovered. Rates of tag loss were estimated and taken into account. The rate of instantaneous mortality (M) was 0.28, equivalent to about 75% survival. A single tag and release study nested within the multiple release study produced an estimate of M=0.46 (64% survival per annum). The difference in estimates might be due to variations in rates of initial tag losses, imprecisely measured rates of long term tag loss, or just the fact these estimates are by nature imprecise and variable.

Other Species

Sloan and Breen (1988) report using estimated growth rates to decompose length frequency distributions to derive estimates of M=0.12-0.20 (89-82% survival per annum) for unfished adult populations of *H. kamtschatkana* in British Columbia.

After converting length composition to age composition data Tutschulte (1976) used both a life-table method and a comparison of the same population between two years to calculate an M of 0.20 for adult (>7 year old) *H. corrugata* in southern California.

Olsen (1971) cited in Sainsbury (1982) estimated M for adult Californian green abalone *H. fulgens* at 0.07 (93% survival per annum) on the basis of physiological data.

Tegner et al. (1989) estimated M at 0.15 (86% survival per annum) for emergent adult *H. rufescens* in southern California.

Clavier and Richard (1985) estimated adult (10-12 year olds) mortality rates for *H. tuberculata* in Britanny, France with tagging at 0.26 (77% survival per annum).

Sainsbury (1982) applied three differing methods based on the use of estimated growth rates to analyse size composition data and derived estimates of M for an emergent population of paua *H. iris* on Banks Peninsula, New Zealand which ranged from 0.02 – 0.25 (98-78% survival per annum). He concluded that the true value lay close to 0.1 (90% survival per annum).

McShane & Naylor (1997) created three 5 x 5m 60cm high mesh enclosures on hard bottom off the Wellington N.Z. coastline and placed about 1,000 tagged *H. iris* into the three enclosures. Where possible fortnightly censuses were conducted within the enclosures and within 20m of the enclosures. The outside searches but not the inside searches involved turning boulders. Tagged abalone found outside the enclosures were returned to the enclosures. A complete destructive search of the enclosures occurred after 12 months. All dead and live shells were collected. They also selected 6 sites off D'Urville Island and distributed another 1,000 tagged abalone between these sites. These sites were revisited after 9 months and searched for live and dead shells. Estimates of natural mortality were based on the ratio of empty shells found compared to the number of live abalone found. The estimates of M produced in this way off the Wellington shoreline ranged from 0.02 – 0.08 (98-92% survival per annum) and from the D'Urville Island site in this way ranged from 0.00 – 0.125 (100-88% survival per annum).

The weakness of their approach is its reliance on the assumption that dead shells are equally as likely to be found as live shells. The authors argue that shells were unlikely to be buried on the sites selected as they were comprised of solid hard bottom and would have persisted for at least 9 months at the D'Urville Island site. Also that predatory fish, rays and octopus likely to carry shells away were not prevalent at the sites, and that starfish were the main source of predation. This is in contrast with other studies which implicate the burying of abalone and predation by larger swimming fish, rays and octopus as major sources of abalone mortality. The authors concede that the estimates should be considered minima but argue their estimates are comparative to other estimates for paua and that the lower mortality estimates reflects lower predator densities and that paua is less productive than other species.

Putting Naylor & McShane's claims into context. Shepherd (1998) monitored the decomposition of shells of juvenile greenlip and scalaris and calculated a mean half-life for empty shells of 48 days. Also Dixon et al. (2006) who estimated mortality rates on experimental reefs using both the sighting probability of individually tagged hatchery produced shells, survey counts and counts of empty shells, found that while shell collections accounted for 10% of the total population, they only accounted for 18% of the total disappearances. Naylor and McShane (1997) captured and released both juveniles and adults, of which adult shells may have been more resilient and less easily lost than the juveniles studied in the two studies cited. But never the less it seems obvious that Naylor and McShane's estimates are very much minimum estimates of M even for *H. iris*.

Shepherd (1990) presented estimates of adult (>3.5 years of age) greenlip abalone (*H. laevigata*) mortality made over 18 years on the same isolated unfished population at West Island on the Fleurieu Peninsula in South Australia by decomposing annual length frequency data for emergent abalone using estimates of growth rates. Over that period the population cycled up and down twice in response to pulses of recruitment and density dependent mortality rates which ranged from M = 0.02 to 0.86 (98-42% survival per annum) around a mean of 0.38 (68% survival per annum). Analysis of dead shells suggested stingrays were responsible for 70-94% of mortality while octopuses, crabs and unknown causes resulted in the remainder. Shepherd (1990) attributed periods of extremely high mortality rates to the aggregation of stingrays in response to pulses of recruitment into the emergent abalone population.

Shepherd et al. (1992) presented estimates of natural mortality derived using tagging techniques on two adult populations of greenlip abalone (*H. laevigata*) they estimated instantaneous natural mortality at West Island to be M=0.22 – 0.38 (80-68% survival per annum) and at Tipara Reef M=0.22.

Overview of Literature on Abalone Reseeding

The vast bulk of the world's experience and research on abalone reseeding comes from Japan where the government underwrites the annual costs of mass producing up to 40 million juvenile abalone of five species for planting out by fishing cooperatives (Kojima 1995; McCormick 2000). Seki and Taniguchi (2000) suggest that annual reseeding rates since the 1980s has been has high as an average 150 million per annum since 1980. The Japanese Government started the marine stock enhancement program in 1963 and abalone were one of the first targets in the program (Inoue 1976). The primary aim in Japan until recently has been to augment catches directly by growing out hatchery produced seed and then recapturing them in the fishery, rather than an aim of enhancing recruitment to natural populations (Seki and Taniguchi 2000). During the 1970s and 1980s the Japanese conducted and published research on the optimal size of release, release techniques and survival rates (Masuda & Tsukamoto 1998).

Many of the Japanese papers are only in Japanese and in relatively obscure local Prefecture Station technical report. Some carry abstracts in English making their figures a little more accessible. More recently Japanese researchers have published a series of good reviews and the overall body of Japanese work is much discussed in the English literature. A review like this by necessity inevitably places a lot of emphasis on the secondary and tertiary literature. The Japanese studies tend to have been of two types; academic studies reporting on shorter term (<2 year) experimental releases into a research area, and recapture by research divers, targeted at studying some facet of reseeding, or longer term commercial scale releases into the fishery to be caught 3-6 years after release as they recruit into the fishery and are fished up. In these studies recapture rates over time are estimated by sampling the catch and looking for abalone with the distinctive green spire of hatchery raised abalone.

Reseeding had been expected to sustain and/or augment fishery production (Seki & Sano 1998). However, despite recent releases of 25-30 million juveniles per annum the annual catch of abalone declined from around 6,500t in 1970 to about 2,000t in recent years (Masuda & Tsukamoto 1998; Hamasaki & Kitada 2008). The declines have been greatest in the north of Japan where the reseeding has been most intense (Tegner 2000). This has lead to increasing discussion as to the cost effectiveness of Japanese stock enhancement programs for abalone (Masuda & Tsukamoto 1998; Seki & Sano 1998; Seki & Taniguchi 2000; Hamasaki & Kitada 2008). More recently the literature is also discussing the use of lower fishing pressure, habitat rehabilitation, translocation and reseeding as means of rebuilding breeding stocks.

In southern California, a wide range of stock enhancement studies have been conducted with the view to rehabilitating depleted stocks; involving releases of both hatchery produced larvae and juveniles, and the translocation of wild adults (Tegner

2000; Bell et al. 2005). In New Zealand, Australia and South Africa larval and juvenile releases have occurred at experimental scales with the aim of studying optimal release conditions, indicative survival rates and potential costs and benefits (Bell et al. 2005). Survival rates in virtually all the non-Japanese experimental trials, and in most of the Japanese experimental work have been estimated from the recovery rates of searching research divers 1-2 years after release. The recovery rates have in general been very low but it is by no means clear that this is an accurate indication of survival rates. This is because only two methods have been used to monitor survival; the collection of dead shells and the number of survivors found over time by searching divers, and together they only provide estimates of minimum and maximum survival (Huchette et al. 2000; Shepherd et al. 2000; Heasman et al. 2004).

The Japanese Experience

Momma (1972) Observed what happened to seeded hatchery abalone dropped from the boat. Most fell on their back and under attack by predators righted themselves before moving several meters over the following hour to more protected sites. Most injured animals were predated. He developed baskets of oyster shell in which to protect the seeded animals during the release process.

A seminal Japanese paper, which I have been unable to find a copy of, and which in any case is in Japanese, is that by Inoue (1976). According to many references to this paper Inoue (1976) studied the survival rate of seed released at different sizes and recaptured one year later to describe a logistic curve-like relationship between the shell length of juveniles at release and their survival rates. The relationship leveled off at about 70% survival at 70 mm and larger seed (Hamasaki and Kitada 2008) and the survival of 20 mm, 30 mm and 40 mm seed were 10%, 30-60% and 70-80% survival respectively (Masuda & Tsukamoto 1998). It was on the basis of this study that Japanese releases have generally occurred at about 30mm shell length and approximately 1 year of age (Hamasaki and Kitada 2008).

Saito (1984) as cited in McCormick et al (1994) placed 20,000 hatchery raised Ezo abalone (*H. discus hannai*) in Oshoro Bay, Hokkaido and found optimal survival rates over 16 months was 29 to 36% for abalone that were 22-38 mm when released. Survival of smaller seed was considerably less. Between 1977 and 1979 an additional 62,754 hatchery seed were planted out at other sites around Hokkaido and the recapture rates (over 3-4 years) was 5 to 10%.

Zhao et al. (1991) reported survival rates of hatchery produced seed of *H. discus hannai* released in four experimental reseedings in Akita Prefecture and modeled size related survival rates and seed costs to estimate the most cost-effective size for releasing seed. The annual survival rates observed varied between 15-22% for 8 mm shells up to 17 to 40% for 30 mm shells. Prices ranged from about 20 yen per 10 mm individual up to 50 – 100 yen per individual for 30 mm individuals. According to their model the most cost effective size of seed was about 20 mm.

Kojima (1995) reported estimates of the recapture rates of hatchery-reared abalone during their whole lives range from 8% to 38% of *H. d. discus* (Yanagisawa et al. 1988; Kanamaru et al. 1993; Tachiyama et al 1993) and from 17% to 22% for *H. d. hannai* (Takeichi 1988; Kanamaru et al. 1993). The estimated recapture rates of planted *H. d. discus* off Abu varied between 12% and 51% in the 1980-85 year classes in this study. Although hatchery-reared abalone ranging from 8 to 45mm in shell length were planted in the waters of Abu, abalone grown from shells less than 13 mm at planting were not found in samples. This seed size appears too small to result in an enhancement of the abalone population off Abu. The results of many release-recapture experiments on Japanese abalone have shown that survival rates of seeds increase as the size at release increases (e.g. Inoue 1976). Seeded animals released towards the end of the first year of life and mainly recovered 3 years later. So survival rates basically reflect three years at liberty.

Seki & Sano (1998) argue that rather than using reseeding to support catches a broader ecological perspective is needed and that Japan should consider using reseeding to re-establish breeding stocks, which should then be protected within the fishery. Using Japanese estimates of growth and survival for *H. discus hannai* they modeled the cost of re-establishing a 12 t breeding stock comprised of 100,000 96mm adults. They estimated this would take 1.9, 1.1, 0.6 or 0.35 million individuals that would be 1, 2, 3 or 4 year old, which would respectively cost 67, 211, 304 or 385 million yen. A crude conversion of these estimated costs would converts these respective yen figures at 100 yen/$AUD to give estimates respectively of $AUD 0.67, 2.11, 3.04 and 3.85 million.

Seki and Taniguchi (2000) describe an erudite series of transplant experiments undertaken 1996-1998 in test fishing ground of Ooya, Miyagi Prefecture in northern Japan. The researchers mapped the habitat of the 1 ha research area (200 x 220 m) that had 'been depleted by recent overfishing' and over several years released 166,000 *H. discus hannai* at the shoreward end of grid of survey transects that ran out across the shallow cobblestones habitat where the releases occurred out through, Elsenia forest and Coralline flats to the edge of the hard substrate. Interestingly for this study, in their introduction and methods the authors discuss determining the correct amount of seed for the 1 ha research area; by assuming that the survival rate of transplanted juveniles over three years would be 10%, so that 100,000 seeds would be required to achieve a final density of $1m^{-2}$. Adding the note that "the highest survival of past transplants at Iwate Prefecture was a recapture rate of 25-30% for 30mm seeded abalone."

Released juveniles were individually tagged with binary coded wire tags. Juveniles were released attached to oyster shell. Transects were surveyed by divers every two months over 1-2 years. All juveniles seen during surveys were collected and shells cleaned to check for characteristic green spire coloration of hatchery produced seed and tag. Sighted densities along the transects, and mapped habitat areas were used to estimate population numbers by habitat and estimate survival rates. Within two hours of the oyster shells being hand placed most abalone had moved off the oyster shells, but remained aggregated around the point of release either staying on the upper surfaces of boulders or moving into narrow crevices between them. Few seed remained on the oyster shells 1 day after release. Over the course of the surveys seed was detected dispersing 100+m along the two longest across shore transects out into the deeper coralline flat habitat. The authors claim their data show that over time the abalone preferentially moved out of cobble stone habitat into the kelp forest habitat where growth rates were higher, and that smaller releases in the coralline flats showed the same pattern. They estimated survival within their research area to be 26.7% after 853 days at liberty, which the authors note is high compared to the 10% normally assumed for the area and amongst the highest rates (25-30%) observed in that prefecture. The authors attributed this relative success to selecting the shallower cobble bottom with its abundant crevices for reseeding.

Hamasaki and Kitada (2008) summarize and analyze Japan's experience with reseeding. They synthesize the results of monitored release programs conducted at experimental and large scales (mean number released varies from 3000 to 78 300 and duration of annual releases ranged from 1-19 years). Shell length at release was around 25-30 mm (about 1 year old) and the abalone were mainly recaptured at 90-115 mm. The release season is mainly from spring to summer for *H. D. hannai* when algal forests grow thickly, and from winter to spring for the other three species when predators are less abundant.' Released animals began to be fished 1-4 years after release and cumulative captures tended to reach a plateau at 4-5 years after. So Hamasaki and Kitada. (2008) treated cumulative recapture rates at 4-5 years after release as the overall recapture rate. Mean recapture rates (with SDs) varied from 0.014 (0.09) to 0.238 (0.11). Rates of recovery thought to be influenced by fishing method, body size at release, quality of seed, release site and season, predators, prey and competitors and interactions between these factors (Howell et al 1999; Leber et al. 2004; Bell et al. 2005, Hamasaki and Kitada 2008).

In their analysis of the data Hamasaki and Kitada. (2008) found some clear cases of density dependence where large numbers of seed were released in limited areas and resulted in higher mortality rates. They also record a release of 50,000 *H. gigantean* & *H. madaka* resulting in a recapture rate of 6.4 – 16% but give no further supporting detail.

Tegner and Butler (1985) and McCormick (1994) suggest that the high survival rates of hatchery-reared abalone in Japan are caused by intense fishing pressure on abalone predators. The coastal waters of Japan are controlled by fishery cooperatives that are able to control these predators, which include octopuses, crabs, carnivorous fishes and spiny lobsters.

The Californian Experience

Californian efforts at enhancement began after abalone landings began declining in the late 1960s with the view to rehabilitating depleted stocks. Experimental releases of both hatchery produced larvae and juveniles have occurred in southern, central and northern California, and the translocation of wild adults have all been trialed in southern California (Tegner 2000; Bell et al. 2005). Modeled on the Japanese programs the California Department of Fish and Game and the Californian Sea Grant Program conducted a variety of seeding experiments in southern California using all three major species; reds *H. rufescens*, pinks *H. corrugata*, and greens *H. fulgens* (Tegner 2000). A range of juvenile seed sizes, planting densities, and techniques were conducted in both island and mainland habitats in southern, central and northern California. Species choice was based on depth, habitat type and what was previously native to each study area. Seeding techniques included hand planting of individuals, the use of oyster shell as seed substrate, cinder block habitats and concrete modules (Burton and Tegner 2000; Tegner 2000).

Many experiments were conducted on the Palos Verdes Peninsula in Los Angeles County (Tegner 2000). Tegner and Butler (1985) describe the experimental releases of 10,000 *H. rufescens* seed of 20 to 35mm in average shell length at Palos Verdes. Predators were not manipulated for the first release and 173 empty shells were collected after the first month. Predators, principally octopus were harvested before, during and after the next release and only 36 shells were recovered from the release site one month after release. Counts by searching divers were used to estimate survival rates after 1 to 2 years. When preliminary data suggested long-term survival was very low, experimental releases of 40-80 mm abalone were trialed along with ongoing predator removal. There was no significant difference between size classes of seed or predator control treatments and the average recovery rate was about 1% overall

(Tegner & Butler 1985; Tegner 2000). However, empty shells collected during the searches only accounted for 44% of the loss of released abalone from the sites, less than half the larger shells were accounted for.

Six larger scale seeding experiments were also conducted with red and green abalone at Palos Verdes, and Santa Cruz and Santa Catalina Island. Despite testing size ranges from 10 to 80mm there was no evidence of better survival rates with increasing seed size. In all cases the estimated 1-2 year survival rates based on the recovery of live seed was 2.8% or less (Burton and Tegner 2000; Tegner 2000).

A discussed above a range of factors probably influence the results of these studies based on research searches. The relatively small scale of these, and many other, release and search sites make it impossible to use these estimates as anything other than lower bounds on minimum survival rates likely.

Davis (1995) seeded 7200 juvenile red abalone into artificial habitats made up of concrete blocks inside a wire mesh frame, structures designed to sample abundance of juvenile abalone for fisheries management purposes, at three sites on Santa Rosa and Santa Cruz Islands in 1989. At the end of the first and second years 3.9 and 0.8% respectively of the hatchery-reared abalone were still living in the artificial habitats, and wild juveniles had also moved into the experimental habitats. Recovery rates of marked shell scattered around the habitats at each site were used to estimate possible maximum mortality rates based on the assumption that all animals not accounted for were dead. Averaging the minimum and maximum estimates the authors derived likely survival rates of 32% after the first year and 24% after the second year. The collection of 2977 empty shells provided no evidence of size related survival across a range of 15-91 mm at the size of release.

Shepherd et al. (2000) considers the Davis (1995) estimate of M at 0.70 as ‘ambiguous’ and excludes it from their analysis.

Rogers-Bennett and Pearse (1998) describe seeding 50,000 *H. rufescens* averaging 8mm in length over six sites in northern California. Juveniles were seeded into six sites during an 11-day period. Two seeding methods were used: hand planting and release from abalone modules. Hand planting red algae that the juvenile abalone had crawled onto into selected habitats like sea urchin canopy and crevices took 6 hours per site. Abalone were seeded at approximately 150-170 seed /m^2. Recoveries of juveniles at 6 months, 1 year and 2 years totaled less than 1% of the number seeded, however seed accounted for one third of the 1995 cohort recovered during the surveys. Recovery rates were higher in areas with a canopy of sea urchins to protect the juveniles. The authors noted that recovery rates did not decrease over time and that this along with the generally the low recovery rates, were in part due to the cryptic nature of the abalone and the short time frame of the study.

Californian Genetic Studies

Gaffney et al. (1996) purported to present genetic evidence showing that one of the Californian Department of Fish and Game reseeding trials had actually been highly successful. The original release of 42,432 juvenile red abalone seed (mean size 30 mm) had taken place at a restricted location Tyler Bight on the southern side of San Miquel Island in 1979. Gaffney et al.’s claim was based on an allozyme analysis of abalone sampled from a single commercial catch made along the southern side of San Miquel in 1992. The area was originally selected for a study of size-specific survivorship because it supported (and continues to support) a healthy abalone population (Burton and Tegner 2000). The site was searched in 1983 but the results were apparently never written up (Gaffney et al. 1996). Gaffney et al. (1996) took genetic samples from adult populations in northern and southern California, and from the Tyler Bight area in particular, as well as from several hatcheries. They found the Tyler Bight and hatchery populations were the only samples out of Hardy-Weinberg equilibrium, and both were 100% heterozygous (Burton and Tegner 2000). On this basis Gaffney et al. (1996) claimed the Tyler Bight sample was dominated by the genome of the abalone out planted in 1979. It should be noted that no samples of the original 1979 broodstock or seed were ever tested, all the samples analysed were taken in 1992.

Burton and Tegner (2000) repeated the approach of Gaffney et al. (1996) sampling the same set of wild abalone populations with the aim of testing Gaffney et al.'s results. Burton and Tegner (2000) found no support for the earlier finding and concluded Gaffney et al.'s findings were highly questionable on several grounds. All the wild populations they sampled had similar levels of heterozygosity around calculated Hardy-Weinberger equilibrium. They pointed out that the diver who collected the 1992 samples had taken them from along 10km of shoreline many times larger that the 100-200m long release site. This made it likely that the sample would contain a mixture of wild and hatchery animals and that this would be expected to be reflected in the heterozygosity of the samples. Burton and Tegner (2000) also noted that the samples Gaffney et al. claimed to be of hatchery origin had 100% heterozygosity at the loci of interest and explained that this was a highly improbable result ($P<0.001$) as it would represent the first generation of uniformily homozygotes parents. They suggested instead that sample degradation explained Gaffney et al.'s (1996) result.

Californian Experimental Studies of Seed Survival

In a series of laboratory experiments Schiel & Welden (1987) found that cultivated and wild abalone (8 to 50 mm) were slower to seek hiding places for the first 7 h following release on rocky substrates in aquaria. Laboratory experiments with hatchery reared and native abalone of the same sizes show subtle but significant differences in behavior of red (Scheil & Welden 1987) and green (Tegner & Butler 1989) abalones and in both cases significantly more cultured animals were consumed when predators were added to the experiments. Scheil & Welden (1987) were able to demonstrate reduced predation on acclimated seed relative to animals fresh from hatcheries.

A variety of seeding devices (Ebert and Ebert 1988; McCormick et al. 1994; Davis 1995; Swejd et al. 1998) have been proposed to efficiently collect, transport and seed relatively large numbers of abalone while also reducing handling stress and facilitating adaption to the bottom and ultimately increasing survival. These often show short-term advantages but have rarely been tested for longer than a few weeks against hand planted controls (Tegner 2000). Tegner (2000) provides a brief description of a longer term comparison conducted by California Fish & Game which concluded that the release modules made the release process more efficient, but that long term recovery rates were determined by longer term heavy predation in the substrate, not the short term impact of initial adaption. Thus they concluded that modules had no significant long term impact because they only delayed the onset of the heavy mortality rate by some days.

The New Zealand Experience

Schiel (1992) conducted small scale replicated experimental out-plantings of paua from December 1985 to 1986 using eight enclosures made of plastic mesh stretched around steel posts hammered into the reef to form 1 x 1m enclosures. Two sizes of abalone were used 1 year olds (19mm) and 2.5 year olds (51.8 mm). They found site, predator exclusion, release size of the paua and whether or not the pauas were hatchery produced or taken from the wild all had significant effects on the results. Notably they observed a 40% difference in survival rates between wild and hatchery produced abalone after 1.2 months at liberty. Shepherd et al. (2000) record this as “The survival of hatchery produced seed was 25% lower after 5 weeks than the wild abalone. However, ‘minor shifts in habitat’ was the most significant factor over all. Replicates most affected by sedimentation had the lowest survival rates.

In larger scale releases survival after one year at liberty was 9-32% and as high as 24% after 3 years, this last figure was based on only one of the three larger scale releases (2100 paua in each release). The other sites had considerable human interference over that time and the results were discounted. On this basis Schiel went on to model cost effectiveness of reseeding using 19.3% as the survival for the first year after release, which was the average survival of all transplanted abalone in the larger scale releases.

In 1990 and 1991 Schiel (1993) released 80,000 seed at 8 sites on Chatham Island. The seed were from 3 to 30 mm and growth and survival was monitored for between 5 and 23 months. The annual mortality rates observed varied between 27.6 and 98.8% and annual growth increments were from 13.5 to 23.8 mm. As in his previous small-scale experiment mortality rates were apparently highest on sites which were prone to being covered by sand. Anticipating this, prior to commencing the reseeding project, he conducted an extensive search for areas he expected to be conducive to producing good survival and the areas he selected were relatively small varying from 208 – 900 m^{-2}. This aspect of the study raises questions about the ability to simply scale up these results to a broad scale program which would need to be less selective for bottom type and so might incur lower survival levels than observed by Schiel. The densities he reseeded at were also relatively high varying from 9.9 – 35.4 ind. m^{-2} reflecting the New Zealand view that paua naturally occur at relatively high density. He records that releasing 10,000 seed by hand required about 4 hours bottom time.

In the longer term releases recovery rates often improved over time suggesting that in the short term the cryptic behaviour of the small seed were confounding estimates of survival.

Roberts et al. (2007) constructed replicate boulder habitats and conducted small scale short term experiments to examine size related survival rates and density related impacts on growth and survival. They then selected 5 natural boulder shorelines to release 2,600 to 20,000 abalone at about 10mm shell length. These were then recaptured 17-20 months later at 50-60 mm length. In the short term experiments relative seed survival increased progressively with seed size, but the greatest gain was between 5 and 10mm. Absolute seed survival over several months at liberty averaged 21± 6% for 5mm shell and 49±13% for 20mm shell. Growth and survival declined with increasing density but there was so much variability around this trend that it was not significant. In the 5 longer term larger scale releases survival rates of 1.7, 7.3, 25.1, 18.7 and 16.2% were observed. The worst survival rate was observed at a site exposed to two large swell events. The average survival rate across all sites was 13.8%

or 16.8% excluding the worst site. Assuming a subsequent natural mortality rate of only 0.1 they then estimated a survival rate to legal size and 300g of 12.5%.

Roberts (2000) contains this summary. "New Zealand reseeding studies have highlighted the importance of site selection choosing habitat with abundant complex crevice space. Movement of rocks and sand during storms caused major mortality in four studies. Lower survival of hatchery stock [than wild stock] was documented in all 3 studies where it was examined. Predator exclusion had only a minor impact in 1 study."

The South African Experience

Sweijd et al. (1998) reported experimental releases of *H. midae* on the northwest coast of South Africa in an area to the north of that species natural range. A total of about 2,600 4-6 month old abalone (10-20 mm) were released at three sites during two releases. Releases were made in batches of either 500 or 800 within a single release module at each site. The area within 15m radius of the three release sites were searched several times over the 12 months following the releases. They argued that recovery rates under-estimated survival because of movement out of the search area and cryptic behaviour. They estimated a minimum survival rate of 30% through to three months post-release. Growth rates were similar to, but slightly lower than rates measured for some wild stocks. Using an assumed cost of R1.00 to 1.50 per seed ($AUD 0.70 - 1.00) they estimated that survival through to 100 mm shell length needed to be at least 10-15%.

The Australian Experience

Shepherd et al. (2000) report on a range of small reseeding and grow-out trials conducted around southeastern Australia during the 1980s and 1990s. Two small scale reseedings of 7 month post settlement blacklip abalone (mean length 12.1mm) into a marked grid over a boulder field at Blubber Head in south east Tasmania by the PhD student Peter White. The first study was a short term study (1 month) looking at 4 release densities 10-40 seed.m^2 and three release methods (attached to abalone shells, inside oval cavities in cement blocks, or by hand under boulders, it was conducted with 3 replicates. Searches were conducted at days 2, 6, 19 and 27, 100% searching efficiency was assumed. Short-term recovery rates indicated no difference due to release technique so this aspect was excluded from the analysis. The second study initiated at the same time with the same structure was sampled at 4, 8 and 12 month intervals.

At the end of the first 28 day experiment all except 5 of the 1666 recaptures were found within their original 2 x 2 m release grid indicating negligible horizontal movement over that time. At 2 days post release the seeds had dispersed almost equally through the three layers of the boulder field, but the majority were in the upper layer by the end of the trial. After two days 46% of the seed were still in the open on the top or sides of boulders suggesting naivity to predation. A plot of presumed % mortality over 28 days showed three phases of mortality. During the first release phase the mean mortality was 21%. In the second phase 2-14 days coinciding with migration back to the surface of boulders mean mortality was 12% and in the final phase 12-28 days mean M was 4.6%, equivalent to an instant rate M of 1.2 and possibly representing the long-term mortality rate.

In the long-term experiment proportional mortality was 84-92% after 4 months and numbers recovered at 8 and 12 months did not differ significantly from the 4 month counts. Maximum movement was 8.9m after 12 months but the vast majority remained in their square of release. The mean size of seeded abalone increased from 12.6mm to 42.9mm in 13 months, similar to the growth at the same site measured by monitoring the seasonal progression of wild stock length frequency histograms (Prince et al. 1988).

Shepherd et al. (2000) also reported on one of what they claimed were many commercial trials of placing 50 tagged three year old hatchery produced greenlip seed in an open cement module in the middle of a large *Posidonia* beds at Flinders Island, Tas. The modules provided a hollow triangular shelter for the abalone but were open so that abalone could leave and but could also enter, but being placed in the middle of a large seagrass bed they were removed from both predators and other abalone habitat. They reported that after 6 years 17 tagged adults with a mean size of 149 mm and total weight of 427g were recovered from the modules. Producing a single estimate of an annual mortality rate of 0.18.

Shepherd et al. (2000) concluded that that the very high rates of mortality over the first few weeks after release were due mainly to handling stress. Although their own data shows that the seeded abalone took some days to finally achieve a stable distribution in the surface layers of the boulder, so beyond the immediate effect of stress, this can be assumed to have resulted in some higher level of risk for seeded abalone. Shepherd et al. (2000) also concluded that high mortality up to one year after release was due to persistent density dependent mortality factors and predator naivety. This later conclusion they supported with the observations of low mortality when hatchery seed are placed onto isolated artificial reef structures with lower levels of natural predators.

In Backstairs Passage, S.A. Dixon et al. (2006) constructed small experimental scale reefs from 12 boulders, 2 x 3 in 2 layers so that they could be effectively searched in their entirety during each survey. The reefs were laid out as two rows of three boulders each about 6m^2 and separated from each other by 3m gaps of seagrass and sand. No natural reef was within 10m of the experimental reefs. The area in which the experiment was conducted had historically produced abalone but at the time of the experiment did not contain significant natural stocks. Dixon et al first tested reefs made either one or two boulders deep and found that double layered boulders produced significantly better short term recovery rates. They then conducted the experiment with double layered artificial reefs so that they could be replicated and also effectively dismantled when searching for the surviving seed. Individually tagged juvenile *H. laevigata* were seeded at about 18 months of age and around 28mm, onto the artificial reefs at 8 sites.

Because they individually tagged the seed they were able to estimate that at the three month survey during the divers failed to find at least 25% of the cryptic seeded abalone that was later estimated to have been present. Six months after seeding almost all juveniles were still in cryptic positions between boulders, but by 9 months after seeding 17% were emergent (mean size = 56mm). They estimated minimum survival after 9 months was poor (0 and 23%) at two sites affected by the starfish *Coscinasterias maricata*, but at the other six sites ranged from 47 to 57%. Empty shells accounted for 10% of seeded juveniles but only 18% of abalone estimated to have disappeared. A second seeding a year later at one site resulted in similar survival after just 3 months. They concluded that their results compared favorably with previous seeding experiments, probably due to careful site selection, careful handling of the seed and the provision of complex cryptic habitat to the seed.

Heasman et al. (2004) and Goodsell et al. (2006) describe an experiment conducted 10 larval releases and 47 releases of juvenile (1-45mm) *H. rubra* along the NSW coast. Having studied the comparative effectiveness of the range of searching techniques deployed for juvenile abalone, wherever possible they selected release sites with habitat conducive to being dismantled and searched destructively. Sites were monitored for 1-3 years. Goodsell et al. (2006) describe the results of a subset of six site selected around Fingal Head and Fingal Island near Port Stephens, NSW to test:

1. The importance of spatial and temporal configuration of seeding (where you placed seed in relation to ledges and sea urchins),
2. Whether placing seed under sea urchins enhanced survival, and
3. Whether or not the release devices enhanced survival rates.

Sites were chosen to be at least 100m apart. Seeding densities were about 120 abalone/m^2. Replicated releases occurred in batches of up to 200 abalone, mainly released from 50 of the cylindrical release devices described by McCormick et al. (1994). After a few days total recovery amongst releases ranged from 3-31%. After approximately 2 weeks the rate of recovery had fallen to 0.6-7% and after 1-2 months it had fallen further to 0.2-2%. Known mortality as evidenced by the collection of empty shells was never greater than 0.4%. They argued that recovery rates were lowered by movement out of the release areas, and the cryptic nature of abalone, but that the results also reflected actual mortality rates. They did not find the placement of seed relative to crevices or sea urchins influenced long term survival rates. Survival in release devices may have been greater than release with urchins, but that result varied between sites. The concluded that while survival rates were low the multiple small releases had been effective at matching the natural densities of the wild stock and on this basis argued that low density releases would be more cost effective than concentrated releases of large numbers of seed. Comparing their estimates of survival with estimates of natural mortality in juvenile wild stock blacklip, as estimated (M=2.5 at 6 months declining to M=0.3 after several years) by Shepherd and Breen (1992) they concluded that most of their estimates of survival were below these levels and some suggested survival rates half the level estimated by Shepherd and Breen (1992).

Reseeding Growth Rates

Kojima (1995) found that the growth curve of planted *H. d. discus* corresponded closely to that of wild abalone: the faster growing seeds reached the legal size of 90 mm in shell length at 2+ years old, and most of the planted abalone exceeded the 90mm legal size at 4+ years in the same way as the wild abalone.” Other workers reported seeded animals being taken as 2-6 year olds (Yanagisawa et al. 1988), 3-7 year olds (Kanamaru et al. 1993) and 4-9 year olds (Tachiyama et al 1993) but at Abu the seeded abalone were all caught as 2-6 year olds, although wild abalone as old as 8 year olds are caught. The difference at Abu between the age composition of planted and wild is because the reseeding only occurred at about 1m depth and fishing pressure is inversely related to depth.

Roberts et al. (2007) assumed wild growth rates for seeded abalone.

Density of Reseeding

Reviewing the literature on reseeding Huchette et al. (2000) note that it seems that each reef can only sustain a certain density of abalone and that above that density mortality escalates rapidly. Probably the limiting factor for juvenile abalone is the availability of appropriately sized crevice space into which they can narrowly wedge themselves to avoid predation by crabs and octopus etc. Universally complex boulder habitat has been found to produce the best survival rates and most workers emphasize the careful selection of this type of habitat to maximize seed survival Analysing data reported by Scheil (1993) for paua in New Zealand, Huchette et al. (2000) reported that Scheil's data showed that he achieved the best survival when seeded density did not exceed 4 x the density of wild stock in the reseeded area. They also estimated and index of intra-specific competition (K) for the trials:

$K = log(seeding) - log(final\ density)$

They observed that when the density of wild abalone was low, implying a limited habitat, a greater proportion of the seed were not recovered and the level of intra-specific competition was estimated to be high.

Heaseman et al. (2004) argued on the basis of their own experiments on the carrying capacity of CCA surfaces conducted in the laboratory, along with the observational results from the field for *H. rubra* (Prince et al. 1988; McShane 1991; Shepherd & Daume 1996; Shepherd 2000) that seeding densities of 1 year abalone should be limited to about 1-10/m^2 because this is the natural carrying capacity of bouldery juvenile habitat. They observed a 3.3 times increase in survivorship when releases of seed occurred in batches of 100 rather than 1,000 individuals. Heasmen et al. (2004) focused on the carrying capacity of the CCA surfaces they studied in determining the carrying capacity of a boulder field, however by the time the juveniles have reached 1 year of age the availability of crevice space, as highlighted by other authors, has probably contributed to determining that carrying capacity.

Roberts et al. (2007) observed good survival rates of *H. iris* (paua) in New Zealand on bouldery bottom of up to 300/sq.m

Costs of Reseeding

Roberts et al. (2007) used a base cost of paua of NZ$0.25 per 5mm seed and $0.0225 per mm SL thereafter in their analysis of the cost effectiveness of differing sized seed. In that analysis they concluded 10mm was the optimal size for release and claim that this result is similar to Heaseman et al. (2004) who found that 5-15 mm abalone are most cost effective.

Cost Benefit Analysis

Schiel (1992) conducted an economic analysis on reseeding paua (*H. iris*) based on a range of assumptions based on his reseeding trials:

- The observed survival rate at each site for the first 3 years of life
- An assumed annual 80% survival for ages 4-6 year old
- Harvest at legal size upon attaining 7 years of age
- The average meat weight (250g) at legal size (125 mm) observed in the fishery.
- 5% interest rate
- Seed cost of $0.20
- Price of meat $75/kg

Schiel (1992) estimated rates of return varied from -100% to 43.7%, 3 sites were estimated to have been likely to produce a positive economic yield while five sites would have produced a negative return. Overall the sites he calculated the financial yield would have been likely to have been about 9.6% greater than the expected yield on the investment at 5% interest.

In their cost-benefit analysis of reseeding 10mm *H. iris* Roberts et al. (2007) assumed (in NZ$):

1. The price of 10mm seed in the experiment – $0.40
2. Current price possible for large and long-term seed supply contracts $0.32
3. A possible future seed price if large scale hatcheries are established in NZ $0.20, and
4. A possible future price if costs rise substantially - $0.60

They estimated returns ranging from 7% to 41% per annum which they considered compared favorably with opportunity costs of about 10% per annum. However these estimated rates of return assumed 3 years at liberty beyond the time scale they examined during which they assumed M=0.1 for 3+ and older animals. They attribute this estimate to Sainsbury (1982) and McShane & Naylor (1997).

Concluding Discussion on Reseeding

As noted by Shepherd & Breen (1992) there are many more studies of the mortality of hatchery seed transplanted into the wild than of the natural mortality rates of wild populations. However, these studies must be interpreted cautiously, because as with all species used for stock enhancement there is evidence for reduced fitness of hatchery reared seed (Tegner 2000; Bell et al. 2005). Cultured seed and wild abalone behave differently in the presence of predators, with initial mortality rates about 25% higher in seed abalone (Schiel and Weldon 1987). Laboratory experiments with hatchery reared and native abalone of the same sizes show subtle but significant differences in behavior of red (Scheil & Welden 1987) and green (Tegner & Butler 1989) abalones and in both cases significantly more cultured animals were consumed when predators were added to the experiments. Scheil & Welden (1987) were able to demonstrate reduced predation on acclimated seed relative to animals fresh from hatcheries.

It is widely assumed that the relatively high recovery rates observed in the Japanese fishery are in some part due to the active removal of abalone predators and competitors that also occurs and that the intensity of predation pressure apparent in Southern California puts a premium on seed behavior that minimizes susceptibility to predation (Tegner and Butler 1985; McCormick et al (1994).

A variety of seeding devices have been proposed to efficiently collect, transport and seed relatively large numbers of abalone while also reducing handling stress and facilitating adaption to the bottom and ultimately increasing survival. These often show short-term advantages but have rarely been tested for longer than a few weeks against hand planted controls (Tegner 2000). Shepherd et al. (2000) concurs with the view based on their experience that the early mortality occurs after the seed enter the bottom so that release mechanisms provide only short term shelter from this affect.

Tegner (2000) concluded that California has learnt “that seeding will not be the hoped for panacea for depleted stocks.”

Shepherd et al. (2000) also concluded that re-seeding poses formidable barriers. Handling stress can be serious but could be managed with insulated containers, lower densities in containers and the use of release modules. However, they considered the persistent elevated mortality suffered by hatchery reared juvenile seed for many months after seeding, as exemplified by their Blubber head trials in southern Tasmania, to be a less tractable problem. The behavioral deficits of hatchery reared animals that are naïve to predators have been better described for fish (Bell et al. 2005). However, several studies on abalone demonstrate that the same issues exist for abalone. Schiel and Weldon (1987) and Tegner & Butler (1989) found that hatchery raised juveniles moved about more, rested in the one more, and were more susceptible to predation than wild abalone. Shepherd et al. (2000) also cite a NZ MSc study by Osumi (1998) showing a 16% to 66% survival over 33 days, respectively, for seed and wild *H. iris* around 35mm in length due to the hatchery raised abalone being slow to move into shelter after reseeding. While Osumi (1998) apparently observed smaller seed (about 20mm) to have similar behavior to wild abalone of the same size, they still had lower survival rates (47% to 71%). Shepherd et al. (2000) concluded that their observations made at Blubber Head supported the results of these other studies results in showing that naïve seed suffering a short period of heavy mortality.

In particular Shepherd et al. (2000) warned against the ability of large mobile abalone predators such as myliobatid rays and wrasse to aggregate in response to the scents released by concentrated releases of abalone, citing various examples of extremely low survival rates observed whenever this occurs. Shepherd et al. (2000) concluded pessimistically about the cost effectiveness of seeding abalone into the wild as a stock enhancement tool, "except possibly, where abalone stocks have disappeared or where enhancement and habitat improvements are combined with marine farming operations."

In contrast, Roberts (2000) reasons that "if naïve behaviour is a function of the hatchery environment, then opportunity exists to "train" abalone to exhibit appropriate behavior. Farmers could provide shelters to maintain cryptic habitat of light-sensitive juveniles. Strong light: dark cycles and night time feeding may encourage the maintenance of normal behavior. Periodic exposure to predators or their odours may help develop and maintain appropriate predator avoidance behavior. Anti-predator training should target the major threats expected at the site and will probably be most effective against the slower moving predators such as starfish."

Relevant to the context of this study would be the additional cost of such 'abalone training' and the scale at which it is possible.

Based on their collective experience Shepherd et al.'s (2000) recommendations for any future reseeding in Australia were:

- Release abalone at optimal sizes determined by experiment to minimize density dependent and behavioral deficits.
- Optimal release densities are probably quite low and rapid release techniques using release devices to minimize handling stress and disperse seed over wide areas by boat have promise.
- Releases should be timed to minimize predation and handling stress. This is likely to be late afternoon as benthic fish predators (wrasse) become less active and in cool weather.
- The quality of seed in a genetic compatibility sense is also of paramount importance but little emphasized or studied.

Overview of Literature Larval Reseeding

Tegner (2000) notes that the high cost, behavioral problems, and low survival rates associated with reseeding juveniles creates interest in whether reseeding larvae may be more cost effective. This approach is based on the premise that very high mortality rates of larvae may be offset by even lower costs of production. Preece et al. (1997) reviews the history of larval seeding from the earliest efforts of Tong et al. (1987) and Schiel (1992) who conducted short term experiments in New Zealand.

Two trials of larval reseeding have been conducted in New Zealand. Tong et al. (1987) released 300,000 12 day old larvae into a 50m^2 gulch. Three months later minimum survival was estimated at 0.4%. At the current price of larvae (US$500 per million larvae the survivors had cost US$0.14 each at <4mm SL (excluding any cost of release). This is 70% of the cost of NZ hatchery seed at 10mm SL suggesting Roberts (2000) that the approach had some promise. Scheil (1992) released batches of 20,000 competent larvae within mesh tents over 1m^2 of seafloor. Despite enclosing the larvae for 24 hours only about 10% of larvae settled in the experimental area. Minimum survival after 5 months was 0.06% of larvae giving a current cost of survivors of US$0.80 per juvenile produced. Scheil (1992) concluded that on this basis the transplantation of larvae was not economically viable as it was much less cost effective than juvenile reseeding.

McShane & Naylor (1995) also conducted experimental releases of competent larvae of *H. iris* onto experimental boulder habitats and observed survival rates of about 10% over 4 months which they considered comparable to that observed for wild *H. rubra* post-larvae by McShane (1991) in Victoria.

Preece et al. (1997) conducted a larval seeding study near Port Lincoln, S.A. They seeded competent larvae into shallow boulder habitat at densities of 1,000 to 80,000.m^{-2}. They used larval tents to contain larvae for 1 to 6 days and tracked growth and survival of post-larval and juvenile *H. rubra* and *H. laevigata* for more than a year after seeding. Preece et al. observed survival rates of 0.4% to 6.5% after 6 days and a mean survival rate of 0.5% after the first 49 days. The authors estimated that if seeded at densities of 1,000-4,000/m^2 which they deemed as low, and if protected from being dispersed by wave action and tidal currents, survival rates after one year would be in the order 0.03%. They went onto estimate yields of legal sized greenlip after six years of 0.0074%.

Roberts (2000) cautions that when evaluating survival rates, that in the protected hatchery environment settlement rates are typically below 10%. Tegner (2000) briefly describe what they describe as encouraging results from trials conducted off Santa Barbara settling larvae into tented crates of cobble.

Shepherd et al. (2000) reported on 6 experiments conducted in 1994 and 5 further experiments conducted in 1995 conducted near Port Lincoln with both *H. rubra* and *H. laevigata*. Seeding sites were within 2-3h transport of the hatchery, long-shore boulder fields 500-1000m^2 in extent with boulders amenable to turning and searching. Larvae competent to sample were transported to sites on damp mesh screens in insulated containers and released over an area of 500m^2 by a plastic hose controlled by a diver. All areas were seeded at larval densities of 4000/m^2. The sites were monitored by diver searches for 1-2 years. In the blacklip experiments a mean of 0.6% (range 0.02 – 1.2%) of the larvae settled in the experimental areas. But only one estimate of the minimum post-larval mortality was possible because natural settlement also occurred during the subsequent period of surveying, confounding counts over time of survivors.

In the greenlip experiments there was also high variability of larval settlement (0.1 – 7.8%, mean 2.1%) and because negligible natural settlement occurred during the subsequent survey period estimates of M of 1.8 to 2.8/month could be made. For older juveniles the mean M per annum for all sites were 2.9 (SE 0.3) for the first year and 1.4 (SE 0.3) per annum for the second year. With the greenlip mortality data a significant correlation existed between M and the logarithm of post-larval densities, so that the mean density of one year old juveniles converged at around 0.7 – 1.0/m^2. But there was no relation between mortality rate of the 1+ age class during its second year and the initial density of that year class.

While Shepherd et al. (2000) notes that larval behavior probably plays a major role and ”little is known about the limiting factors except in a general sense. Observing that Sasaki & Shepherd (1995), McShane (1991) and Shepherd et al. (2000) all found found that post-larval survival was strongly density dependent. In Shepherd’s (2000) paper, five out of nine larval seeding enhancements produced no enhancement after two years, although some enhancement persisted through the first year. On their empirical evidence they proposed the carrying capacity of CCA crusts in their studies were <100 post-larvae/m^2 and that most of their releases far exceeded this level to no further benefit. Shepherd et al. (2000) concluded that “the fragility of abalone larvae imposes a serious liability on the widespread use of larval seeding for enhancement. The other main weakness is the unpredictability compared with seeding juveniles.”

Heaseman et al. (2004) experimentally tested a range of handling and habitat characteristics experimentally on the settlement rate of larvae of *H. rubra*. Factors tested were; crustose coralline algae (CCA) settlement surface or settlement plate, presence or absence of a chemical (GABA) known to attract settlement to CCA surfaces, temperature shocks, refrigeration, water flow and elapsed time. The type of settlement surface was found to be the predominant factor, with CCA being strongly preferred, low water flow and longer time periods were the other most important influences. Heasman et al, (2004) also conducted laboratory experiments to determine the optimum density to settle larvae onto CCA covered rocks. They tested densities in the range 1,000 to 32,000/L and observed densities on the rock 2, 14, 28 and 56 days later. They found that while higher densities produced higher settlement densities over time the density of larvae on the CCA rock converged at the same level. Average yields at day 56 were 3-12% of the original number seeded, about 100 to 250 juveniles per litre of CCA rock. Mean growth rates were also significantly lower for the denser settlements. Density dependent effects apparently drove abalone off the CCA surfaces onto the walls of the tanks faster and at smaller sizes (<1mm), and those that left the CCA surface had higher growth rates. Converting the volume of CCA boulders they used into estimates of surface area of CCA they estimated that the carrying capacity for 6 month old 8-10 mm juveniles in typical boulder habitats (characterized by surface areas of 2m^2/m^2 of planar bottom areas as used by Preece et al. 2000) is likely to be only 8-16/m^2.

Heaseman et al. also released 24 million competent *H. rubra* larvae in 10 releases along the NSW coast between Port Stephen and Eden. At the time or writing they could only evaluate survival at three sites on Fingal Island, Port Stephens. Each of the three 500m^2 boulder zones were seeded with 1.1 million (2,200.m^2) competent larvae, followed 8 months later with a second batch of 2 million competent larvae (4000/m^2). The three sites yielded an average of 125 juveniles of hatchery origin when sampled 316 days after the second seeding. The weighted mean age of the surviving seeded stock at the time of sampling was 400 days and the average survival was 42 juveniles per million larvae seeded (0.0042%).

Heaseman et al. (2004) concluded that is was "clear from the experiments that several factors have the potential to influence the ability of *H. rubra* larvae to settle and metamorphose in the field. Perhaps most importantly, the remarkably low rates of settlement over short time periods in the presence of relatively low water flow suggest settlement in the field will be limited unless larvae can be positioned in areas that maximize the duration of their exposure to appropriate habitat. Appropriate habitats in the field are discrete and restricted, being interspersed with areas where survival and growth is likely to be low. Unless larvae are released in areas of low water flow settlement will be low. This emphasizes the importance of release method. On the other hand within the context of several days of transport the larvae proved relatively robust to handling conditions."

Overview of Literature on Translocation

Saito (1979) reported on a translocation program run from the 1950s through to the mid-1970s in 6 Prefectures around Funka Bay on the south east coast of Hokkaido, the northern most major island of Japan, just outside the natural range of abalone. The translocations involved a total of 4.72t abalone collected from crowded low growth potential areas on the south west coast of Hokkaido, and transplanted into an area of Funka Bay that was recognized as having good growth potential but no natural abalone stocks. One of the Prefectures had only recently commenced the program so had no results. Basically all emergent abalone (30-110 mm and 0-9 years of age) were collected from the southwest coast of Hokkaida and translocated to the opposite coast. On average harvesting occurred two years after the transplantations began and recovery rates were estimated for the 5 Prefectures which had been fishing the translocated abalone for some time at 30.5, 22.8, 22.6, 25.8 and 2.3%. The lower recovery rates were attributed to the extreme eastern location of the newer grounds where conditions wre not thought to be as favorable for the abalone. Recovery rates of 25-30% were considered uneconomic in terms of an on-growing exercise. However, Saito (1979) noted that his estimates took no account for the breeding that took place before the translocated abalone were re-harvested. Arguing from length-frequency histograms and the magnitude of continuing catches that the natural recruitment created by the translocated abalone had been considerable. Tegner (2000) concluded that Funka Bay is an example in Japan where transplantation of wild juveniles was successful in establishing a self-sustaining populations of *H. discus hannai* in an area that did not naturally support abalone.

Inoue (1976) as cited in Shepherd & Breen (1992) observed transplanted *H. sieboldii* and *H. gigantean* decrease for about 30 days before stabilizing. As cited in Shepherd & Breen (1992), Momma (1972) reported that a high proportion of injured outplanted abalone subsequently died.

Hendersen et al. (1988) transplanted 517 adult pink abalone (*H. corrugata*) in two groups during 1983 from San Clemente Island in the Californian Bight to a reserve on Santa Catalina Island which was also a popular anchorage site for recreational divers. Short term mortality rates of the two translocations (27% and 10%) after several months were proportional to the observed injury rates caused by capture and handling (29% and 10%). A final search 12 months after the translocation found less than 3% of the transplanted abalone. The low survival rate was attributed to poaching by recreational divers.

Emmett & Jamieson (1989) investigated the biological and economic feasibility of transplanting large numbers of adult stunted ‘surf’ pinto abalone (*H. kamtschatkana*) to formerly productive high growth rate areas. Survivors were found by dive searches nine months later. They observed immediate post-handling mortality of 2-3% and estimated recovery rates 2 years later within the field of search as being about 30%. The abalone were translocated at an average size of 88mm and recaptured with an average size of 100mm.

Tegner (1992) translocated 4453 sexually mature green abalone (av. size 156 mm) abalone (*H. fulgens*) between November 1981 and September 1982, to two locations on the Palos Verdes Peninsula. By September 1983 they had observed a total mortality of 9.1%. Most deaths occurred within a few days of the transplant and sites at which predator removal occurred prior to the transplant had much lower levels of mortality. Recruitment of newly settled juveniles was observed subsequent to the transplant at scales of hundreds of meters out to 4 km away from the brood stock introduction. There was no equivalent in green abalone recruitment at two control sites removed by some distance from the translocations. However, in the long term this translocation failed because poaching apparently stripped out the translocated adult stock and their recruitment over time (Tegner 2000). This outcome highlights the fact that the effect of adult translocation lasts longest when adult mortality is lowest.

Concluding Discussion

Shepherd and Breen (1992) provide what is still the most thorough review of the literature on the estimation of mortality rates for abalone, particularly with regards to studies of wild populations. Although it is now somewhat dated with regard to the survival rates of reseeded abalone as many studies on this topic have been published since their review. However, in summarizing they make three salient overarching points which remain valid despite the passage of time and the growing body of work on the topic

The first is the difference between the mortality rates of seed transplants and wild populations associated with high initial mortality rates caused by handling during the release process, and the naive behavior hatchery produced abalone display initially (Saito 1984; Schiel & Welden 1987; Tegner & Butler 1989). Shepherd & Breen (1992) note that studies of the mortality of hatchery seed transplanted into the wild must be interpreted cautiously, because cultured seed and wild abalone behave differently making them more vulnerable to predation. It is widely assumed that the relatively high recovery rates observed in the Japanese fishery are in some part due to the active removal of abalone predators and competitors that also occurs (Tegner and Butler 1985; McCormick et al (1994).

The second point Shepherd and Breen (1992) emphasize is the great variability in mortality rates observed between species, age, habitat, density and other environmental differences. They attribute the variability between species to both latitudinal differences with colder water having lower mortality rates, and differing assemblages of predatory species. They suggest that colder water species such as *H. kamtschatkana* and *H.iris* have lower adult mortality rates (0.1 – 0.2) while warmer water species such as *H. marae*, *H. corrugata*, *H. fulgens* and *H. laevigata* have higher adult mortality rates rates and *H. rubra* being between those groups. Although they add the caveat that ‘many more studies need to be done before these trends become convincing.’

All reviews on this topic note the variability of the estimates produced and generally concur it is due to differing habitats and predator populations. However, as the underlying techniques being used in each study varies, and they are all imprecise, it becomes extremely hard to determine, how much of the measured variability reflects real variation? And how much might be attributed to observational error due to the vagaries and inherent biases of the techniques employed in differing regions at differing times.

Finally Shepherd and Breen (1992) reviewed the body of evidence that indicates that contrary to the original dogma mortality rates decline with age/size amongst the cryptic juvenile age classes before stabilizing at lower levels amongst the emergent adult population age classes. Studies of the mortality of juvenile abalone during this phase in the wild are rare (Shepherd & Breen 1992). Shepherd et al. (2000) could cite only 4 published studies on natural mortality rates for juveniles 6 months to 3 years old in wild populations, but there were many studies of the mortality of hatchery seed transplant in the wild. Shepherd and Breen (1992) combined the various estimates of M for blacklip abalone derived from work conducted in Tasmania (Prince et al. 1988) and Victoria (Day and Leorke 1986), greenlip from Shepherd's studies and values for *H. tuberculata* (Clavier and Richard 1985) and plotted M as a function of age to produce the figure below.

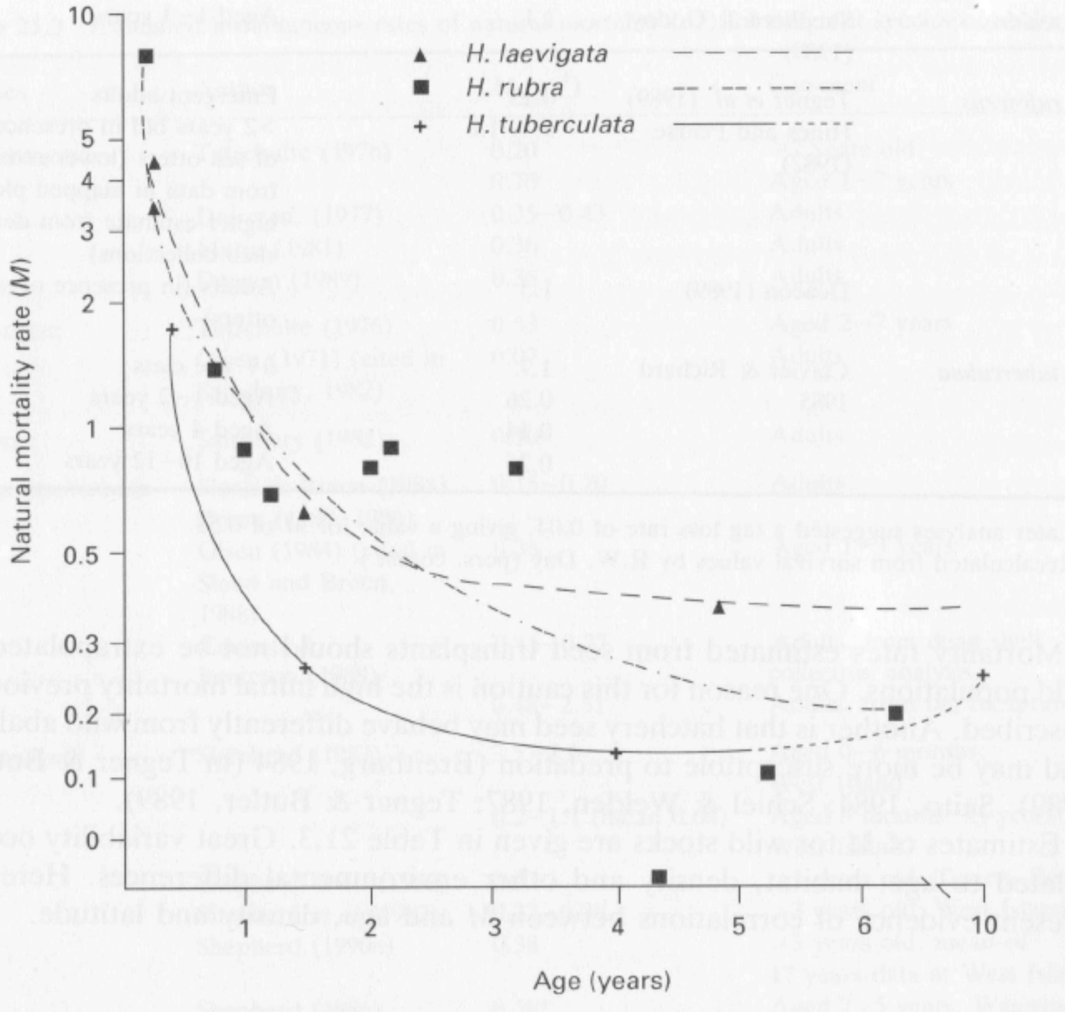

Fig. 21.2 Plots of *M* against age for *H. laevigata* in South Australia, *H. rubra* in Victoria and Tasmania and *H. tuberculata* in France. Data are given in Table 21.3.

Figure 1. Plots of M against age derived from the literature by Shepherd & Breen (1992).

Shepherd et al. (2000) noted that Prince et al. (1988) and Day & Leorke (1986) observed M in these year classes to be 0.7 – 0.9 (40-50% survival per annum), while Clavier & Richard (1985) found a value of 1.7.year^{-1} (18% survival per annum). In a long term study over 13 years Shepherd (1998) found a mean M of 1.0. year^{-1} (37% survival per annum) for H. laevigata and 0.4 (67% survival per annum) for H. scalaris. Shepherd (1998) also reviewed 12 seeding studies of at least one year's duration and found M values ranged from 0.2 to 11.9 year^{-1} (82-0% survival per annum). Excluding the latter extreme and the study of Davis (1995) whose estimate was considered unreliable, Shepherd et al. (2000) estimated a mean M for all studies of seeded abalone of 1.8. year^{-1} (16.5% survival per annum) indicating that mortality of seed juvenile transplants was thus far been higher than that for wild abalone of the equivalent age group. Heasman et al. (2004) noted that their "rates of seed survival were about half those of about 30% achieved with *H. midae* outside the normal range of abalone in South Africa by Sweijd et al. (1998), and about a third those of 40 to 50% reported for wild *H. rubra* (Prince et al. 1988; Day & Leorke 1986)."

It should however be noted that the studies of Prince et al. (1988) was based on tracking the progression of juvenile size cohorts and suffers the bias of being less likely to find the smallest size classes. As noted by Prince et al. (1988) this bias is likely to cause the estimates to over estimate the true rate of survival.

Modelling Mortality Rates

Introduction

The literature synthesized above clearly shows that seeding and translocation incur some level of additional mortality above normal natural levels. However, using the assembled literature to estimate that additional level or mortality, or even a comparative difference in mortality rates between wild and reseeded juveniles is far from straight forward making the selection of an appropriate age schedule for the mortality rates of juvenile abalone under each of the three potential rehabilitation techniques challenging. This synopsis reveals that the 'normal' rate of juvenile natural mortality in wild abalone populations has been rarely studied. Shepherd & Breen (1992) noted that there are many more studies of the mortality of hatchery seed transplanted into the wild than of the natural mortality rates of wild populations. However most of the studies that have been conducted worked with blacklip abalone, and together with some international studies they create a consistent if not concise picture. This body of information has been well summarized and discussed by Shepherd & Breen (1992) and Shepherd et al. (2000). Consequently the age related natural mortality curves outlined by those authors must hold some authority, and provide some sort of baseline assumptions for this study.

As discussed above rates of mortality amongst reseeded juveniles have been documented in two fundamentally different ways. In Japan recovery rates of released cohorts have been documented some years after release as the seeded cohort has moved through the fishery over several years. This provides an accurate estimate of overall recovery rate for each cohort, because the recovery processed continued until the entire cohort had emerged from the cryptic juvenile habitat and been captured.

However this approach obscures the annual rate of mortality in each year because the individuals in each cohort are captured after and over several years. It must also be remembered that the relatively high recovery rates observed from the Japanese fishery are widely attributed in some part to the intense removal of abalone predators and competitors that occurs (Tegner and Butler 1985; McCormick et al 1994).

By way of contrast non-Japanese studies tend to have been much more limited in extent and time span, normally involving experimental scale releases, involving just 100s to a few thousand seeded individuals, and recovered by research divers after just one or two years. Theoretically the data from these studies should be easier to use in this exercise, as the time at liberty over which mortality has occurred is more easily determined and so the rate of mortality should be more easily estimated. However, as discussed repeatedly above, it is often unclear as to what extent the low recovery rates recorded by these projects reflect actual mortality, or loss of individuals to view or from study sites. Amongst the non-Japanese studies the most useful results are those that built in to their experimental design the estimation of the confounding factors. Unfortunately these are rather few. Another approach has been to set up experimental habitats either within enclosures to contain the seed, or isolated and designed to be dismantled at the end of the study so that something approaching 100% of surviving juveniles could be recovered. The weakness of this third approach is the confounding influence of the experimental structure itself on the observed mortality rate (e.g. Schiel 1992).

From the perspective of this study it is unfortunate that there have been very few parallel comparative studies of both wild and reseeded mortality rates. The only parallel comparative study of wild and reseeded abalone survival rates Schiel (1992) observed the mortality rate of reseeded abalone during the first year of release to be about 40% higher than similarly sized wild *H. iris* placed in the same experimental enclosures.

A Re-Analysis of Mortality Rates

The state of the literature and the needs of this modeling exercise led to a re-analysis of the extensive Japanese experience with the aim of converting those results into annualized rates that could be used in the model and compared to the non-Japanese results for wild stock mortality rates.

Hamasaki and Kitada (2008) present in figure 2 of their paper plots of the cumulative recovery rates of 49 cohorts of abalone released by 10 differing Prefectures. These plots were re-digitized to derive the recovery rate of each cohort. From this it was possible to calculate an arithmetic mean recovery rate of 0.128 and a geometric mean recovery rate of 0.124, the range being 0.002 to 0.5 (Figure 2).

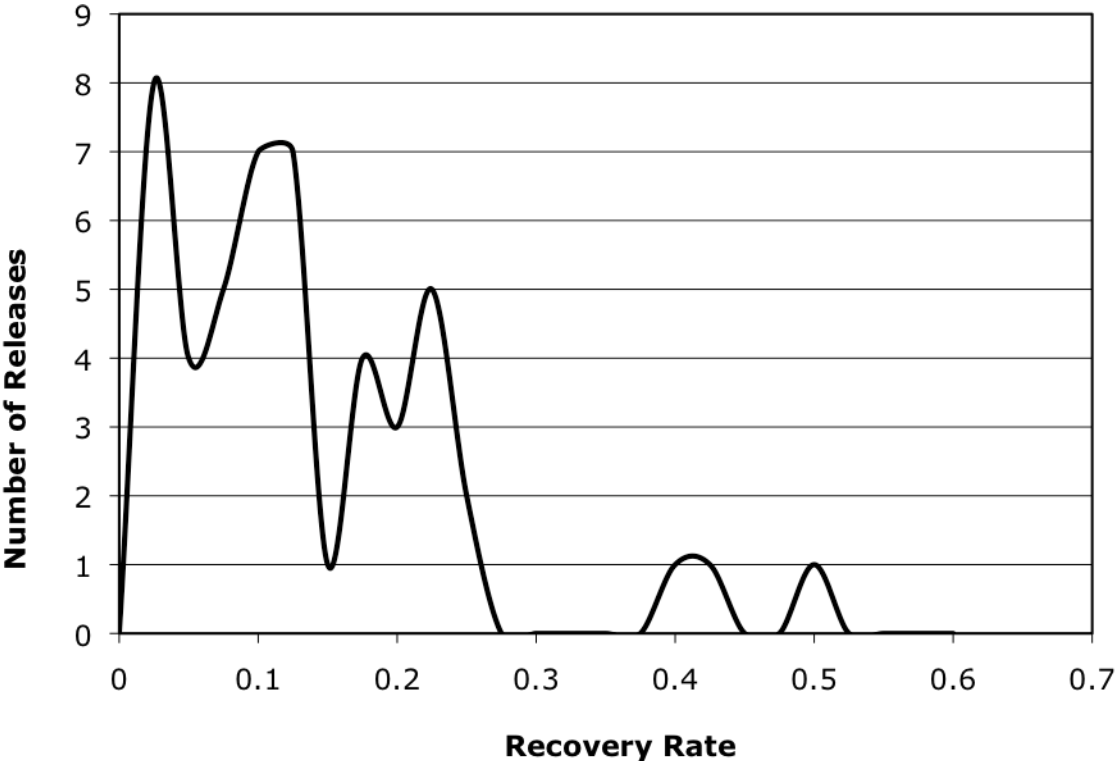

Figure 2. Re-digitized and re-plotted data drawn from figure 2 of Hamasaki and Kitada (2008) which provides the cumulative recovery rates of 49 cohorts of abalone released by 10 differing Japanese prefectures. This figure plots the recovery rates as a frequency histogram of the recovery rate recorded for each cohort across all studies.

To make this useful in this analysis it was then necessary to estimate the correct length of time over which to apportion the survival. Hamasaki and Kitada (2008) observe that in Japan abalone have generally been released at about 30mm shell length and approximately 1 year of age and have mostly been recovered by the fishery from 1-4 years after release and that cumulative captures have tended to reach a plateau at 4-5 years after release. Consequently they treated cumulative recapture rates at 4-5 years after release as the overall recapture rate.

Kojima (1995) conveniently tabulates in his table 3 the recovery profiles by year for 10 released cohorts starting with the 1980 cohort. By Hamasaki and Kitada's (2008) criteria six of these cohorts were fully recovered by the time Kojima compiled his figures. The numbers in the Kojima's (1995) table 3 were converted in to the proportion of each cohort recovered by age of the cohort (Figure 3) and used to estimate a mean time at liberty for these cohorts, which was calculated as 3.16 years.

From this the best estimate of survival of reseeded abalone in these Japanese programs is assumed to be approximately 12.5% over about 3 years at liberty and that very few studies recorded recovery rates >28%. However to put this estimate of a mean recovery rate of 12.5% into context; in Miyagi Prefecture in northern Japan, Seki and Taniguchi (2000) took as standard the assumption of 10% survival to estimate the number of seed needed to achieve eventual stock densities of 1 abalone/m^2.

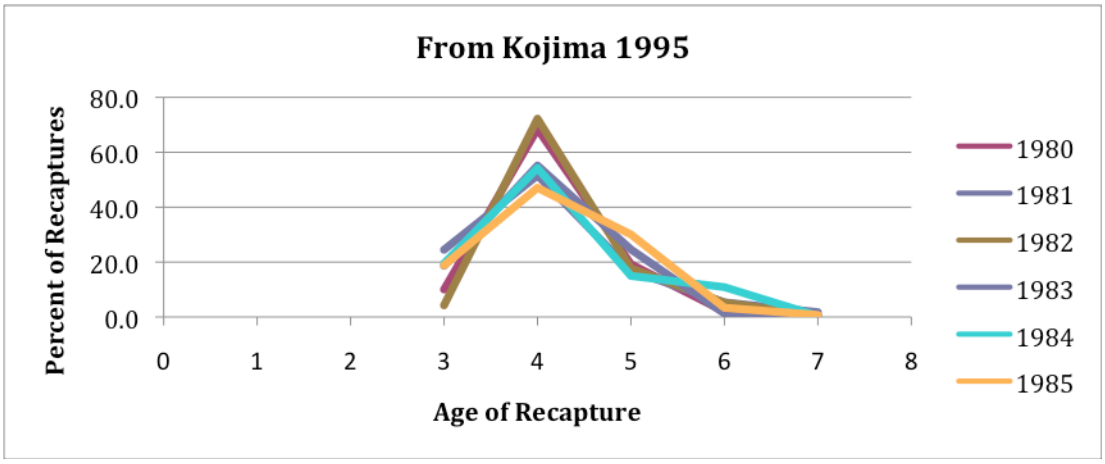

Figure 3. Data on the percent of seeded abalone recapture by age. The data have been plotted from tabular data provided by Kojima (1995).

Assumptions about Mortality Rates

On the basis of all the material contained in this synthesis the assumptions that this modeling analysis commenced with were:

5. That in the Western Zone blacklip abalone would be seeded at around 20-30mm and at about 1 year of age, similar to the Japanese practice.
6. That in the first year after release the reseeded juveniles will have approximately 40% higher mortality rates that wild abalone in their second year.
7. That in subsequent years both wild and reseeded juveniles experience the same mortality / survival rates.
8. That juvenile and adult survival rates should be consistent with the literature.

On this basis the base case model incorporated the mortality schedules shown in Table 1. The schedule reproduces the average survival rates observed across the Japanese studies (about 12% at 4 years of age) and the 40% survival differential observed by Schiel (1992), while implying slightly higher survival rates for wild stock abalone than has been observed across the various studies (Shepherd and Breen's 1992).

	Wild	**Wild**	**Seed**	**Seed**
Year	**Mortality Rate**	**Survival from age 1**	**Mortality Rate**	**Survival from age 1**
0	0.9			
1	0.6		0.70	
2	0.4	0.5	0.40	0.3
3	0.3	0.3	0.30	0.18
4	0.2	0.21	0.20	0.126
5	0.15	0.168	0.15	0.1008

Table 1. The mortality and survival schedule used in the Basic Population model.

Appendix 4: Final Cost Benefit Analysis report.

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Table 4 Comparison of reseeding 500,000 adults per year for three years with an assumed abalone price of \$20, 25 and 40/kg and an assumed price of seed varying between 40, 50 and 60c and various discount rates. For each scenario, using a range of discount rates (0.03, 0.05, 0.07), estimates of discounted deferred value of the stock when it recovers to B_{MSY} reduced by the compounded cost of rehabilitation are shown, above the estimated difference in deferred value to that estimated for the Natural Rebuild strategy.

Table 5 Comparison of translocating 27,000 adults each year for three years assuming the mortality of translocated abalone is 1%, 5% or 10%. The top half of the table shows the estimated discounted deferred value of the stock when it recovers to B_{MSY} reduced by the compounded cost of rehabilitation for each scenario and assuming a range of discount rates (0.03, 0.05, 0.07). The bottom half shows the difference in deferred value to that estimated for the Natural Rebuild strategy.

Table 6 Comparison of reseeding 500,000 juveniles each year for three years assuming the mortality of reseeded abalone is 1%, 5% or 10%. The top half of the table shows the estimated discounted deferred value of the stock when it recovers to B_{MSY} reduced by the compounded cost of rehabilitation for each scenario and assuming a range of discount rates (0.03, 0.05, 0.07). The bottom half shows the difference in deferred value to that estimated for the Natural Rebuild strategy.

Introduction

Analytical Framework

The basis of this analysis is that a constructed model of abalone growth and reproduction, viral impact, and recovery with or without reseeding and translocation is used to predict the comparative population trajectories across a range of rehabilitation scenarios, and the indicative cost and benefits of comparative scenario are estimated on the basis of model estimates. The detail of the model is provided below.

Implicit in this method is the fact that a model can only estimate scenario trajectories on the basis of the assumptions and data built into the model and outlined above.

This analysis was conducted in two parts.

The initial model development occurred prior to the 13 August 2009 WADA workshop at which the model and an initial analysis was presented. It should be noted that up until that time the analysis was based on the initial place-holder values for the main parameters. The WADA workshop provided estimates of the likely cost of the rehabilitation strategies and instruction on the range of parameter values they wished analysed. The major difference between the values WADA agreed should be used in this final analysis and the value used in the initial phase of the analysis is that the first model assumed seed would cost 20c per abalone, while this final analysis uses 40-50-60c per abalone as its range of analysed values. This change means that the analysis presented here makes reseeding options more expensive than assumed in the initial analysis. The initial analysis also did not include the associated costs of outplanting either juveniles or adults and including these costs in the final model to some extent makes both forms of rehabilitation more expensive in this final model. The workshop also provided good length-weight data which was incorporated into the model which then had to be refitted to describe historic trends. This process re-estimated the natural productivity of the modelled stocks suggesting the modeled stock is a smaller but more productive stock. However, as a sensitivity test the various rehabilitation scenarios were analysed using the initial biological model with the slightly larger less productive stock and the results produced were similar to those produced by the new model.

The preliminary analysis presented to the 13th August 2009 WADA workshop proceeded relative randomly analyzing potential rehabilitation strategies, 30-40 scenarios involving time period and size of releases were compared to the predicted natural rebuild scenario. That initial analysis clearly showed that scenarios tailored to rebuild the missing year classes rapidly back to the level which produces optimal recruitment to new year classes (i.e. B_{MSY}) were optimal in terms of both rates of return and gross levels of income above that expected from Natural Rebuild. As might have been expected the analysis showed restocking above B_{MSY} reduced the profitability of the intervention. At the other extreme smaller interventions sooner could be most profitable in terms of rate of return on expenditure by pushing the stock further up the steep left hand part of the stock recruitment curve (Figure 4). But these interventions could only be limited in size, because the steep part of the SRR curve occurs over a relatively small range of biomasses. Thus the highest rates of return are limited to the smallest interventions, many of which produce very small absolute levels of income above Natural Rebuild. Likewise immediate intervention is more profitable than delayed intervention for the same reason, as a stock recovers naturally each addition of biomass provides a proportionally smaller boost to future recruitment. All of these features remain in this final version of the model but are not as fully explored in this final analysis.

This report on the final analysis does not fully elaborate again on the features described in the preliminary report to the WADA workshop. Instead it has been focused more tightly around the reseeding program proposed at the WADA workshop. As proposed this would involve a commitment to producing 500-600,000 seed for three years. Consequently, the focus of this final analysis is variations of enhancement strategies that occur over a three year period. A range of reseeding and translocation strategies of parallel value and both greater and larger than that proposed have been analysed and compared.

Model Description

The analytical model has been developed as a series of spreadsheets, all using the same underlying age based population model.

A reader using this documentation to inspect the model mechanics is referred in the spreadsheet accompanying this report to the spreadsheet entitled 'Natural Rebuild' which is the most basic version of the population model as it only describes the natural behavior of the Kilarney Reef Code.

The population proceeds forward in time across the sheet in annual steps through time denoted with the subscript 't' from 1965 in Column E, to 2064 in Column CZ. The model also proceeds in annual ageing steps towards older age classes. These annual aging year classes will be denoted with the subscript 'a'. The age classes or cohorts in each year of the simulation run down the rows starting at Age 1 in Row 21 and running down to Age 36 in Row 56. In this context Age 1 refers to abalone during their first year of life, which might be referred to as the 0+ age class.

The first major blocks of lines E21 out and down to CZ36 describes the basic population dynamics of each scenario in numbers (thousands) of individuals. Within this major block of lines, and also the three similar replicated blocks below this primary table, each cell represents an age class in a year of the simulation. Reading along a row shows how that age class is estimated to have changed each year through the simulation.

Replicated blocks of lines below; E60 down and out to CZ95 give the same figures for each scenario but in weight (tonnes) of spawning age abalone left after fishing, E99 down and out to CZ134 give the weight (tonnes) of the catch, and E138 down and out to CZ173 is the weight (tonnes)of emergent mature biomass before fishing.

Basic Population Model

For any reader looking at the excel code underlying the model, apologies for the convoluted 'If' statements aimed at stripping out non-zero estimates that confound the fitting routines used to fit modeled trends to observed trends.

The first year of each scenario is in Column E and is denoted as 1965, however it only serves to initialize the population age structure.

The initializing number of Age 1 abalone for E21 is 'Virgin Recruitment' which is set in B11.

Natural Mortality

Subsequent age steps between rows in Column E simply remove the proportion of each cohort dying each year:

Eq. 1:

$$N_{a+1} = N_a \text{ x } (1\text{- Proportion Dying in year t})$$

The proportional rate of natural mortality for each age class is specified in cells B3:8 and the schedule is shown in Table 1. Mortality is assumed to decline from 90% in the first year to 15% by the 6th year of life.

Biomass Estimation

In the replicated blocks of lines below; E60:CZ95, E99:CZ134 and E138:CZ173 for each year of the scenario the population modeled by age in numbers of individuals in E21:CZ56 is converted into estimates of biomass (weight) based on the weight at age matrix contained in C21:C57.

The weight at age matrix is based on an assumed Gompertz growth curve which approximates the growth pattern know for the area

Eq. 2:

$$\text{Length} = W_0 \text{ x } \exp(G \text{ x } (1\text{-}\exp^{(-g \text{ x } age)}))$$

Where Length is in mm and Age in years, $W_0 = 1.62$, $G = 4.56$ and $g=0.498$

While the length-weight relationship for the Port Fairy area provided by Mr Glen Plummer based on a sample of 1,049 abalone from the area.

Eq. 3:

$$\text{Weight} = 0.0001 \text{ x Length}^{3.0474}$$

The proportion emerging and maturing at each age is defined B21 to B36 using a relationship that is typical for blacklip abalone in southeastern Australia. The schedule for age, weight and proportion emerging and maturing is shown in Table 1.

Catchable Biomass

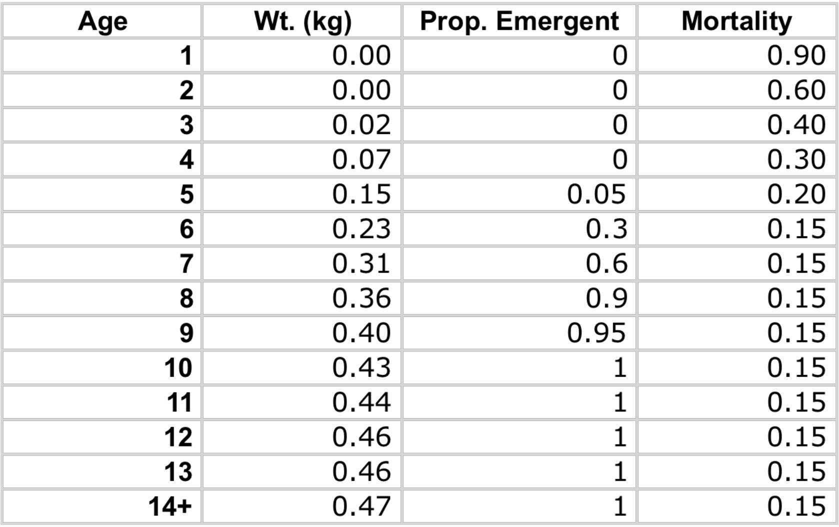
Logically the last of these blocks of code E138:CZ173 is the first to consider. In this block of code the numbers in each age class (a) in each year (t) is converted to a weight of emergent and catchable breeding abalone in each age class (a) and in each year (t).

If the age of the year class is equal to or greater to the Minimum Age Limit set in B10 then:

Eq. 4:

$$\text{Catchable Biomass}_{a,t} =$$

$$\text{Proportion Emergent in year (t) x Weight (kg) at Age (a) x Number}_{a,t}$$



Age	Wt. (kg)	Prop. Emergent	Mortality
1	0.00	0	0.90
2	0.00	0	0.60
3	0.02	0	0.40
4	0.07	0	0.30
5	0.15	0.05	0.20
6	0.23	0.3	0.15
7	0.31	0.6	0.15
8	0.36	0.9	0.15
9	0.40	0.95	0.15
10	0.43	1	0.15
11	0.44	1	0.15
12	0.46	1	0.15
13	0.46	1	0.15
14+	0.47	1	0.15

Table 1. Schedules used in the model to specify the weight, proportion maturing and emerging and proportional natural mortality for each age class.

In this version of the model the minimum Age of Recapture was set at 8 year throughout. If the age of the year class is less than the Minimum Age Limit set in B10 then:

Eq. 5:

$$\text{Catchable Biomass}_{a,t} = 0$$

Catch

The proportion of the emergent population removed by fishing in each year (F_t) is found in rows E14:CZ14 having been calculated in the following manner.

The total biomass available to be caught in each year (Catchable Biomass_t) is estimated for each year in E12:CZ12 by summing rows 138:173 in each column. The annual TAC_t is defined for the scenario in E10:CZ10.

As long as TAC_t is less than Catchable Biomass_t (i.e. $TAC_t <$ Catchable Biomass_t)

Eq. 6:

$$F_t = TAC_t / \text{Catchable Biomass}_t$$

However if $TAC_t <$ Catchable Biomass_t then $F_t = 1.0$

In the block of lines from E99:CZ134 the weight (tonnes) of catch from each age class (a) in each year (t) is calculated.

Eq. 7:

$$\text{Catch}_{a,t} = F_t \text{ x Catchable Biomass}_{a,t}$$

The total catch in each year, Catch_t is computed in E13:CZ13 by summing rows 99:134 in each year.

Within the basic numerically based population model found in E21:CZ56 the number being removed by fishing in each year is accounted for in the annual time step so that:

Eq. 8:

$$N_{a+1,t+1} = (N_{a,t} - (\text{Catch}_{a,t}/\text{weight at age a})) \times (1 - \text{Proportion Dying at age a})$$

Breeding Biomass

Finally the block of lines E60:CZ95 estimates the weight (tonnes) of breeding age abalone left in each age class (a) after fishing in each year (t).

Eq. 9:

$$\text{Spawning Biomass}_{a,t} = \text{Catchable Biomass}_{a,t} - \text{Catch}_{a,t}$$

The total amount of spawning biomass left to breed after fishing in each year Spawning Biomass$_t$ is computed in E11:CZ11 by summing rows 64:95 in each year.

Stock Recruitment Relationship

Recruitment in numbers resulting from breeding in any year t, Recruits$_t$ is estimated from a standard recruitment equation which prescribes an curving convex parabola from zero breeding biomass at the origin up towards an upper asymptote at Virgin Spawning Biomass. This standard recruitment curve is used in the same way in the regional assessment framework and is parameterized as follows:

Eq. 10:

$$\text{Recruits}_t = \text{Virgin Recruitment} \times (\text{Spawning Biomass}_t / ((\text{Alpha} \times \text{Virgin Spawning Biomass}) + (\text{Beta} \times \text{Spawning Biomass}_t))$$

The fixed values for Virgin Recruitment and Virgin Spawning Biomass, along with the constants Alpha and Beta must all be supplied to the model. The constants Alpha and Beta are derived from the steepness factor 'h' which by definition is the proportion of virgin recruitment produced at 20% of Virgin Spawning Biomass.

Thus:

Eq. 11:

$$\text{Alpha} = (1-(h - 0.2) /(0.8 \times h))$$

And

Eq. 12:

$$\text{Beta} = (h-0.2)/(0.8 * h)$$

For this study steepness 'h' is assumed to be 0.6. This is the value assumed in the regional assessment.

The values for Virgin Recruitment and Virgin Spawning Biomass used in this study have been estimated through simulating the catch history and stock dynamics of the Kilarney reef code and fitting these values, so that simulated model trends are as close to possible to the observed trends in the fishery. In this way these crucial stock and recruitment parameters are estimated from what historic and biological data exist. When the same approach was followed with a parallel model developed for another Western Zone Reef Code – The Crags, the fitted model behaved similarly to the Zonal Assessment model Dr David Bardos had adapted to formerly analyses that reef codes trends. Dr Bardos did not develop a model for Kilarney but the essential similarity of the model, the estimated trends and the two reef codes provide some basis for confidence in the trends and parameters estimated by this approach as at least being the most plausible and defensible in this situation. Noting that in this analysis all the differing treatments will have their costs and benefits assessed using these shared parameters

A novel aspect of the approach being followed here is that the units of measurement being used and derived in the stock-recruitment equation changes. Thus recruitment is estimated in thousands of individuals which will be the first age class in the following year of the simulation, while spawning biomass is estimated in tonnes. To make this transition the Virgin Spawning Biomass is measured in weight (40t) while the Virgin Recruitment (9,750) is given in thousands of individuals.

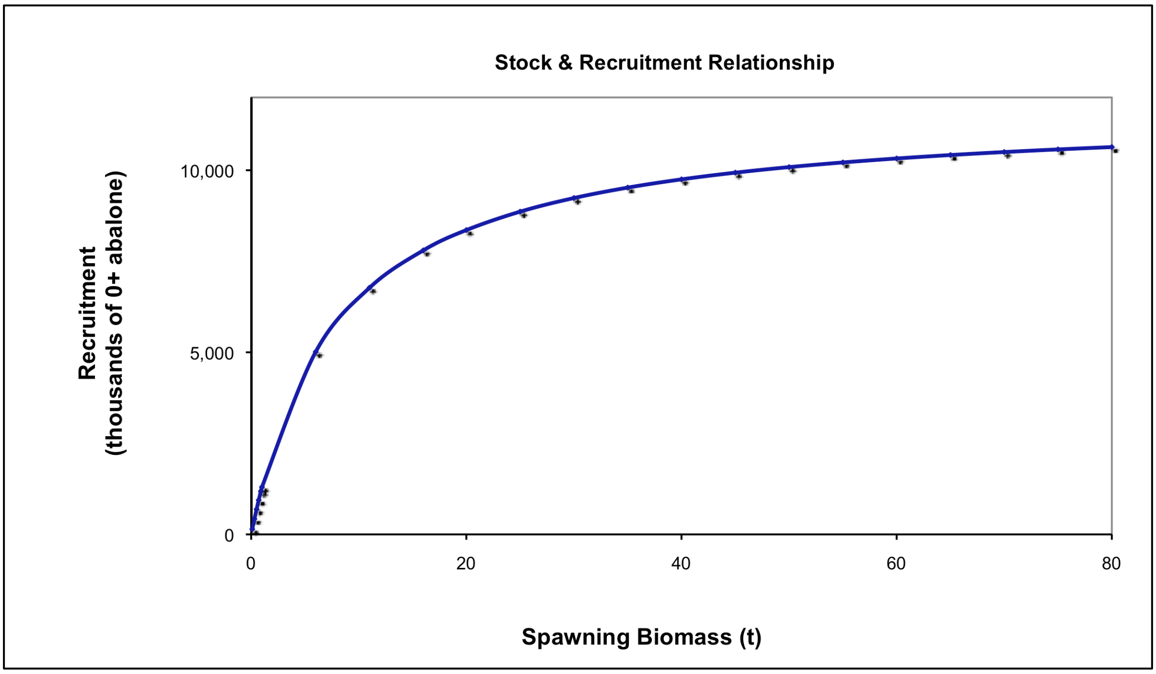


Figure 1. Plot of the assumed stock recruitment curve used in this model. Note discussion in the text about the fact that this widely accepted general shape assumed here precludes depensatory effects at low biomass levels.

The resulting relationship suggests that around 50t of breeding biomass a maximum level of around 10 million 0+ abalone, but that around 8 million 0+ abalone would be still produced by about 20t of breeding biomass. At B_{MSY} the population described by the model sustains a yield of 33t from a residual breeding biomass left after fishing of about 32t.

Conditioning the Population Model

The basic population model described above was conditioned to simulate the population dynamics thought to be occurring at Kilarney prior to the viral outbreak of 2007. This simulation through to 2007 includes the known history of catches (E13:AU13) and the modeled impact of the virus itself in 2007. The stock recruitment and biomass parameters estimated through that process suggests that the virgin biomass of Kilarney pre-fishing was 150t, and that its maximum sustained yield

(MSY) is around 33t and that this annual yield can be produced from an adult standing stock of around 65t prior to harvest.

The viral impact in 2007 is modeled by the basic population model. Any level of viral mortality can be set in Cell B16 where a proportion (0-1) is entered for the additional mortality caused by the virus in 2007. The viral mortality rate is applied in Column AU (2007) to cells 21:56, as a uniform additional level of mortality on all age classes including the recruiting year class. The default setting for the level of viral mortality for this analysis has been 0.95.

Reseeding Population Model

Both the Reseeding population model found in the spreadsheet and entitled 'Reseeding' and the Translocation population model entitled 'Translocation' replicate the basic population model described above in every detail. In addition to the basic model they have added into them the various aspects needed to describe the form of rehabilitation being analyzed.

The essential difference made to the Reseeding Model allows for a program of reseeding to be described. Reseeding rehabilitation scenarios can be entered in Cells D2:Q3.

The Reseeding Scenario in this version runs from 2010 through to 2019 is entered in 1000s of seed into Cells H2:12. Their unit cost in cents is added in Cell E3, along with the number of seed released per diver day (E6), the cost per diver day (E5), the number of seed per release module (E17), the cost of release modules (E4). Estimates for all these parameters were provided by the 18 August 2009 WADA workshop (Table ZZ).

The model computes the total cost of the reseeding strategy in each year in Cells H3:Q3 and the total cost of the Reseeding program is computed in Cell E7.

Eq. 13

Total Cost of Reseeding Strategy = (Number of seed x cost per seed) + (Cost per Diver Day x Number of seed/Seed released per diver day) + (Cost of Release module x Number of Seed/# Seed per release module)

In the model it is assumed that reseeded animals will have completed their first year of life in the hatchery. However, because the seed suffer a different mortality rate in their first year in the wild, and for ease of programming, the model does not add them to the 2nd year class in 2010, but rather to the 3rd year class in 2011 (Cell AY23) after reducing their number by the reseeded rate of mortality in that year. So that:

Eq. 14

$$N_{3,2011} = (N_{2,2011} \text{ x } (1\text{-Mortality Rate Age 2}) \\ + (\text{N Seeded in 2010 x } (1\text{-Seed Mortality Rate})$$

In the base case model the mortality rate of seed during their initial year of mortality is assumed to by 70%.

The numbers seeded from 2010 – 2019 are treated similarly AY23:BH23. Having been added to the population of abalone in their 3rd year of life, the model no longer distinguishes seeded abalone from wild abalone in the population. From that point in time they are assumed to contribute to normal population dynamics.

Translocation Population Model

The Translocation Model is found in the spreadsheet entitled 'Translocation'. The essential difference made to the Translocation Model allows for a program of translocation to be described. The Translocation Scenario which in this version runs from 2010 through to 2019 is entered in number of abalone translocated into Cells H2:12. Their average weight and price/kg are entered into Cells E3 and E4 respectively. The number of abalone released per diver day is entered into E6 and the cost per diver day into E5. The model computes the total cost of the translocation strategy in each year in Cells H3:Q3 and the total cost of the Reseeding program is computed in Cell E7.

Eq. 15:

Total Cost of Translocation Strategy =

(Number translocated x average wt. x cost per kg) +

(Cost per Diver Day x Number of translocated/Number of abalone released per diver day)

In the model it is assumed that translocated animals will be collected from mature emergent populations elsewhere and subject to an initial translocation mortality, specified in B2. Translocated abalone are added into the 9th year class of the model in the year of their translocation Cells AX29:BG29. So that:

Eq. 16:

$$N_{9,2010} = (N_{8,2009} \text{ x } (1\text{-Mortality Rate -Adult}) + ((N_{\text{Translocated in 2010}} \text{ x } (1\text{-Translocation Mortality})/1000)$$

In the base case model the mortality rate of translocated abalone is assumed to by 5%.

The number of animals translocated is entered in numbers and the model computes in 1000s of abalone, so in the above equation the number being translocated is divided by 1000.

The numbers translocated from 2010 – 2019 are treated similarly AY29:BH20. Having been added to the population of abalone in their 9th year of life, the model no longer distinguished translocated abalone from the local wild population. From that point in time they are assumed to contribute to normal population dynamics.

Economic Parameters

On the 13th August 2009 a workshop was held for WADA members and associated scientists and aquaculturists and the models preliminary analysis was presented for discussion. That meeting used its collective experience to provide estimates of the costs of key parameters and indicate the range of values they wish analysed.

Reseeding Strategy and Associated Costs

At the workshop Mark Gervis of Southern Ocean Mariculture outlined construction constraints and proposed a feasible strategy involving a three year commitment from

WADA to purchase 500-600,000 seed per year. He noted that larger numbers were infeasible while smaller numbers would be un-economic in the light of the infrastructure needed. Mr Gervis estimated that in this strategy the cost of seed would be approximately 2c/mm and the seed would be 20-30mm with a mean of 25mm at about 12 months old.

The workshop agreed that the model should analyse the range of seed prices; 40, 50 and 60c per seed and the range of survival rates after seeding to be considered should be 0.1, 0.25 and 0.35.

The workshop estimated the cost associated with outplanting as:

6 divers @ 7,000 seed/day = 100,000 seed / day

Cost per diver day supported by vessels = $1500/day

Cost of Release Modules: $10 -Onion Bags & Ab-Shells or PVC and cement.

Abalone / release module = 200

Translocation Strategy and Associated Costs

The meeting agreed that the price of abalone assume should be $20, $25 and $35/kg. That the range of translocation mortalities that should be assumed should be 1, 5, and 10%

Abalone translocated per diver day = 1000 abalone

Cost per diver day supported by vessels = $1500/day

Methodology - Comparing Strategies

In this analysis three measures of success are used to compare the different strategies. In all cases the comparison is against the do nothing Natural Rebuild scenario, i.e. the cost and income above that expected with the Natural Rebuild scenario.

Breeding Biomass Trajectories

This first basis for comparison was used more heavily in the preliminary analysis and relies on the visual comparison of the expected trajectories with which the differing strategies rebuild the spawning stock. In this report only a few plots are presented showing the biomass rebuilding trajectories of a representative few scenarios involving varying lengths and magnitude of intervention. The primary focus of this final analysis is on variants of three year strategies as suggested by the WADA workshop. These comparisons have been pursued using both a more descriptive and a quantitative approach.

Estimated cost benefits relative to Expected Income 2011-2025

In the more descriptive analysis of the various rehabilitation strategies a base case biological model was established and used to estimate:

- The cost of the strategy
- Time taken to recover stock to B_{MSY} and
- The Income that could be expected from 2011-2025.

In this method there is no accounting for the discount of value with time.

Discounted Deferred Value of the Resource upon Recovery

For the final quantitative approach applied to the estimates provided by the model I am indebted to Drs Sarah Jenning and Satoshi Yamazaki (UTAS) for suggesting the following methodology for succinctly comparing the differing economic outcomes expected from the different scenarios. In this analysis the bench-mark used to compare strategies is the value of the stock rebuilt to B_{MSY}, discounted according to cost of the treatment and the years taken to return the stock to its B_{MSY} equilibrium, which in this final version of the model equates to a biomass 65t producing which is estimated to sustain a catch of 33t per annum.

As observed in the fishery the capitalized value of the B_{MSY} catch is calculated as a multiplier of the income produced each year. The 13 August 2009 WADA workshop agreed the multiplier that should be used is 7, being approximately the long term value observed in the fishery.

Eq. 17:

$$\text{Capital Value} = 7 \text{ x MSY Catch x Price of Abalone}$$

So given the number of years (years) taken for the stock to return to B_{MSY} the deferred Capital Value is estimated as

Eq. 18:

$$\text{Deferred Capital Value} = \text{Capital Value x (Discount Rate)}^{\text{Years}}$$

Beside the Deferred Capital Value of each scenario the difference between the Natural Rebuild scenario and other scenarios is calculated as

Eq. 19:

$$\text{Difference to Natural Rebuild} =$$

$$\text{Deferred Capital Value}_{\text{Natural Rebuild}} - \text{Deferred Capital Value}_{\text{Rehabilitation Strategy}}$$

Results

Rebuilding Trajectories

In these initial projections it is assumed that no catches are taken after the viral event in 2007 and so only the predicted rate at which the Breeding Biomass is rebuilt is being compared. A wide range of Reseeding and Translocation Scenarios were initially created with the aim of exploring a range of scenarios for each type of rehabilitation technique, and for investigating the magnitude of the differing interventions needed to markedly change the predicted Natural Rebuild trajectory. These scenarios varied in both duration and magnitude of the intervention.

The base case economic model used unless specifically noted assumed abalone to be worth \$25/kg, seed abalone 40c/piece, the initial mortality of translocated abalone to be 5% and of seeded abalone 70%.

Reseeding Trajectories

Illustrative Comparison of Biomass Rebuilding Trajectories

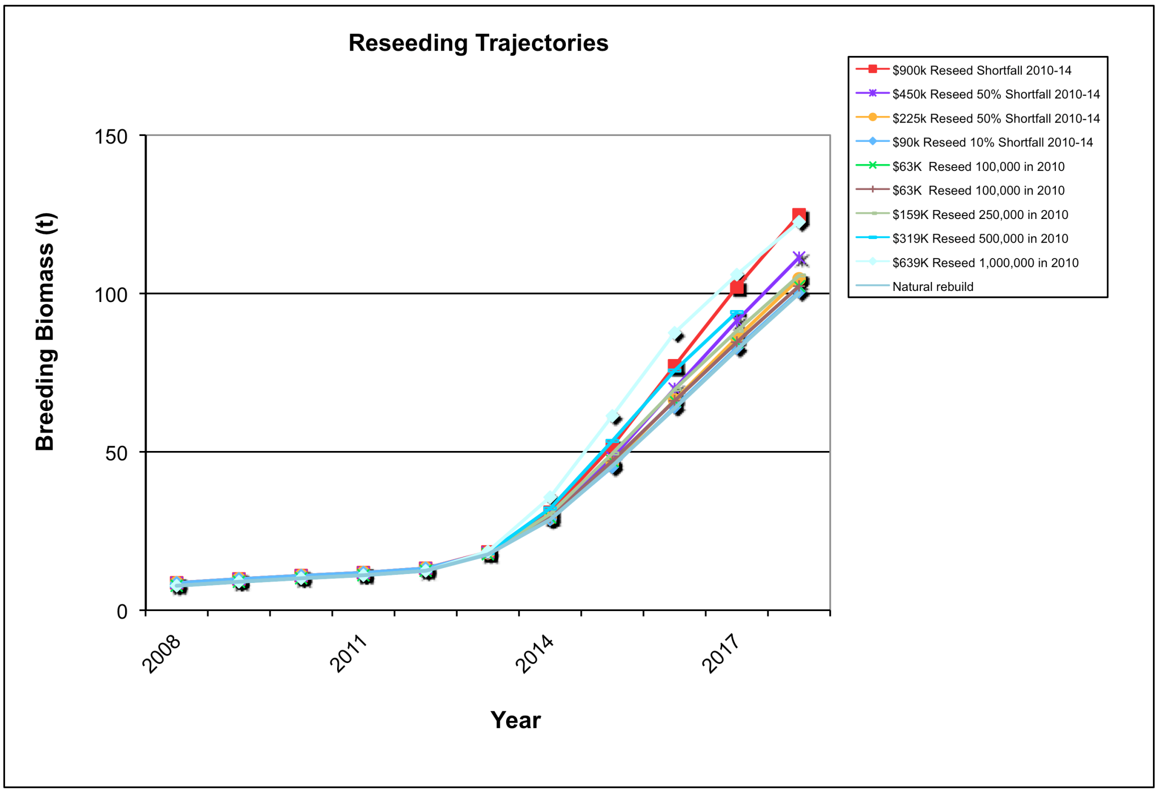


Figure 2. Plot of adult biomass over time under an initial range of reseeding scenarios. The biomass trajectory predicted without reseeding, Natural Rebuild, is provided for comparison.

Figure 2 plots the predicted trend in time of breeding biomass with the 9 reseeding strategies. The predicted breeding biomass trend with no reseeding (Natural Rebuild) is also plotted for comparison. The assumed form of the stock recruitment curve which remains highly productive at low biomass levels causes the model to suggest that even without reseeding the breeding biomass will be starting to recover strongly by 2013. The various reseeding scenarios do little to decrease that time lag because the young seed must themselves age and grow before they contribute to the breeding biomass. All the trajectories plotted suggest the biomass rebuilds to around >60t by the year 2016. The model computes 66t is the adult B_{MSY} and predicts that even the Natural Rebuild scenario reaches 63t in 2016.

Descriptive Analysis

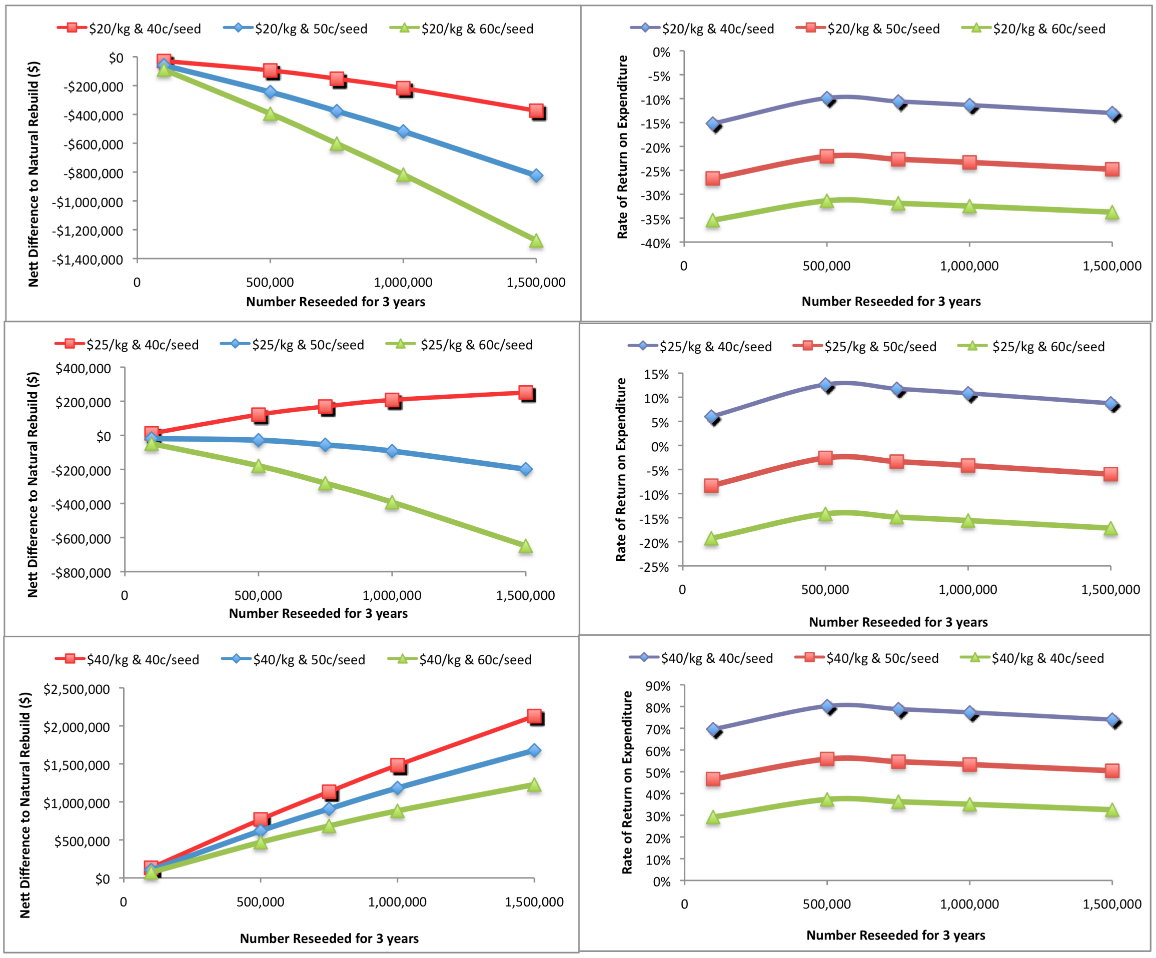


Figure 3. Estimated income ($) above Natural Rebuild for comparative three year reseeding scenarios (100,000 – 1,500,000 seed per annum) in the left hand panels, and the estimated rate of return on the cost of reseeding, right hand panels. Estimates of income and return are provided across a range of abalone prices (20, 25, 40 $/kg) and seed prices (40, 50, 60c/seed).

Figure 3 shows the results of the models analysis of comparative reseeding strategies based on three equal releases beginning in 2010. The analysis suggests that reseeding strategies of around 500,000 per year for three years should be marginally more profitable than smaller or larger alternatives. However the analysis also shows that whether the reseeding strategy is expected to be profitable or loss-making depends entirely on the value of abalone on the market and the cost of the seed.

At $20/kg for abalone none of the reseeding strategies were estimated to be profitable. At $25/kg only the scenario with seed at 40c/abalone is espected to be profitable with a return of 5-15% expected on expenditure. At $40/kg all reseeding scenarios were estimated to be profitable with returns on expenditure ranging from around 30% if seed were 60c per piece, to around 80% if seed is assumed to be 40c/piece. Assuming $40/kg for the market value of abalone estimated returns on the cost of the strategy were as high as $2.5 million was estimated for 40c seed and 1.5 million releases annually. Assuming $20/kg for the market value of abalone estimated losses of 10-35% were predicted, as high as $1.2 million in absolute terms depending on the cost of seed and size of releases.

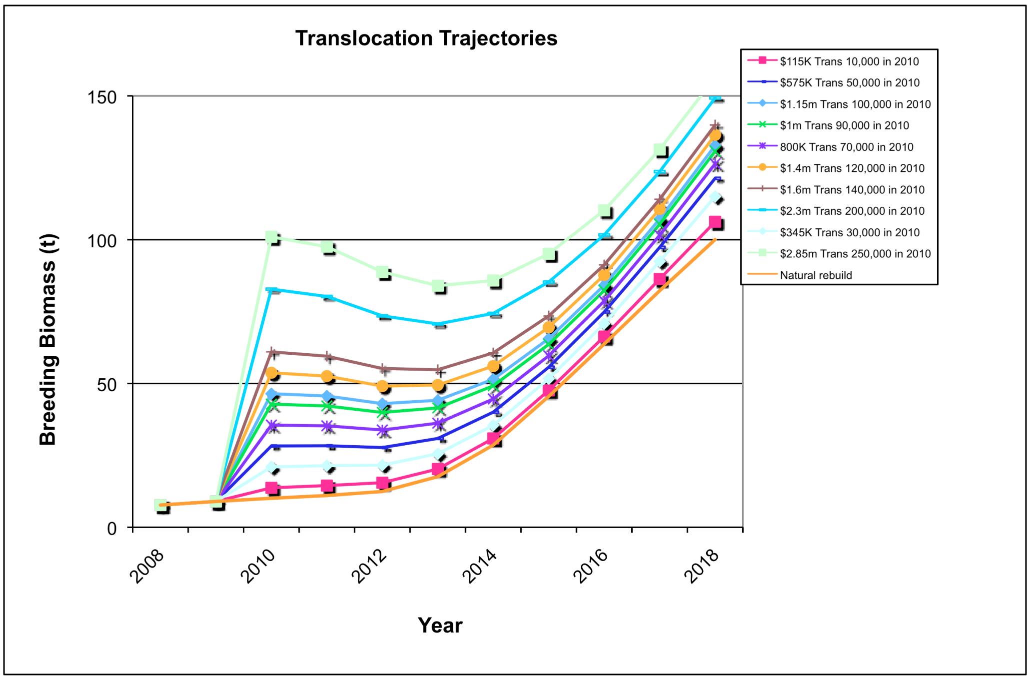


Figure 4. Plot of adult biomass over time under an initial range of translocation scenarios. The biomass trajectory predicted without translocation (Natural Rebuild) is provided for comparison.

Also of note is the way that the translocation scenarios that involve large translocations in the first years, show a gradual decline when the translocation ends, until around 2014, in contrast to scenarios where similar levels of translocation were spread over 5-6 years which display smoother continuous increases. These declines occur because of the 5-6 year time lag between the initial enhancement of breeding biomass through translocation, and the additional recruitment to the breeding stock that produces, during which the breeding biomass is declining by the rate of Adult Mortality (0.15) if the translocation program stops.

In contrast to these ‘frontloaded scenarios’, in scenarios where similar levels of translocation were spread over 5-6 years trajectories were more linear. Despite the differing early trajectories of the larger translocation strategies, most of the translocation scenarios initially selected achieved relatively similar ranges of Breeding Biomass above the B_{MSY} of 63t by 2016. On the other hand, as expected, scenarios that translocated comparatively low numbers had trajectories similar to the Natural Rehabilitation scenario.

Translocation Scenarios

Illustrative Comparison of Biomass Rebuilding Trajectories

Figure 4 plots the predicted trend in time of breeding biomass for the 10 translocation strategies. The predicted breeding biomass trend with no translocation (Natural Rebuild) is also plotted for comparison. In these scenarios there is no time lag between the commencement of the strategy and the rebuilding of the breeding biomass. If the amount of translocation is large enough the breeding biomass could theoretically be rebuilt to B_{MS} within a year. Consequently most of these initially arbitrarily selected trajectories rise rapidly at the beginning in contrast to the Natural Rebuild or low level translocation scenario.

Figure 5 plots the predicted trajectories for the larger, more effective rehabilitation strategies for comparison. The largest translocation strategies produced breeding biomasses >75t in the initial year 2011, but most reached a breeding biomass around the B_{MSY} of 63t by 2015, after several years with biomasses around or building up to B_{MSY}. In contrast by augmenting juveniles the reseeding translocations do not begin building significant levels of breeding biomass until around 2004 and by 2015 most have recovered to around B_{MSY}.

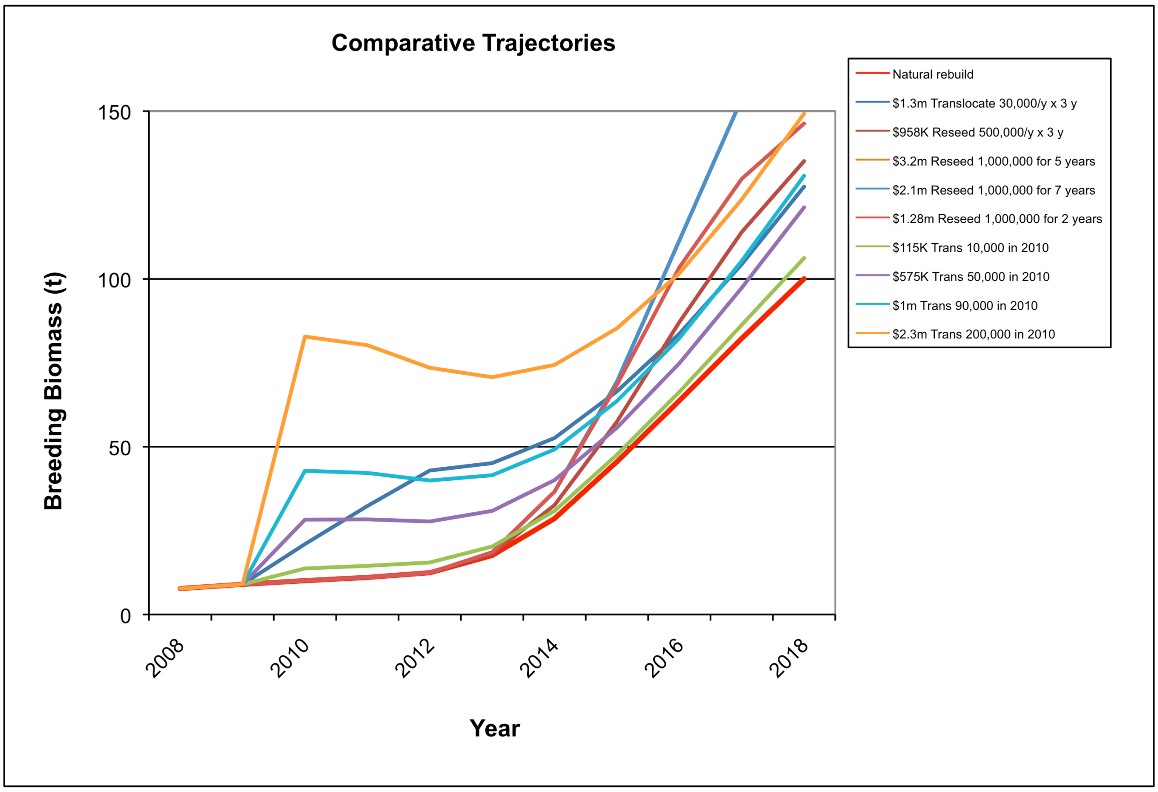


Figure 5 Plotted trajectories of nine reseeding and translocation scenarios, selected as relatively cost-effective in building the biomass rapidly towards B_{MSY} and above. These are plotted against the Natural Rebuild scenario.

Descriptive Analysis

The descriptive analysis (Figure 6) suggests that three year translocation strategies of around 20-40,000 adults per year for three years should be more profitable than smaller or larger alternatives. Potentially returning an income $200-600,000 after expenditure, a return of 20-50%.

The analysis suggests that the price of abalone makes very little to the expected rate of return. It should however be noted that price stability was assumed so that abalone were bought for translocation and sold for income at the same price. It would be expected that assuming a lower purchase price than sale price would make scenarios more profitable while a falling price would reduce profitability. Only those translocation scenarios (100,000 adults/year for 3 years) that stocked well in excess of B_{MSY} were estimated to produce losses.

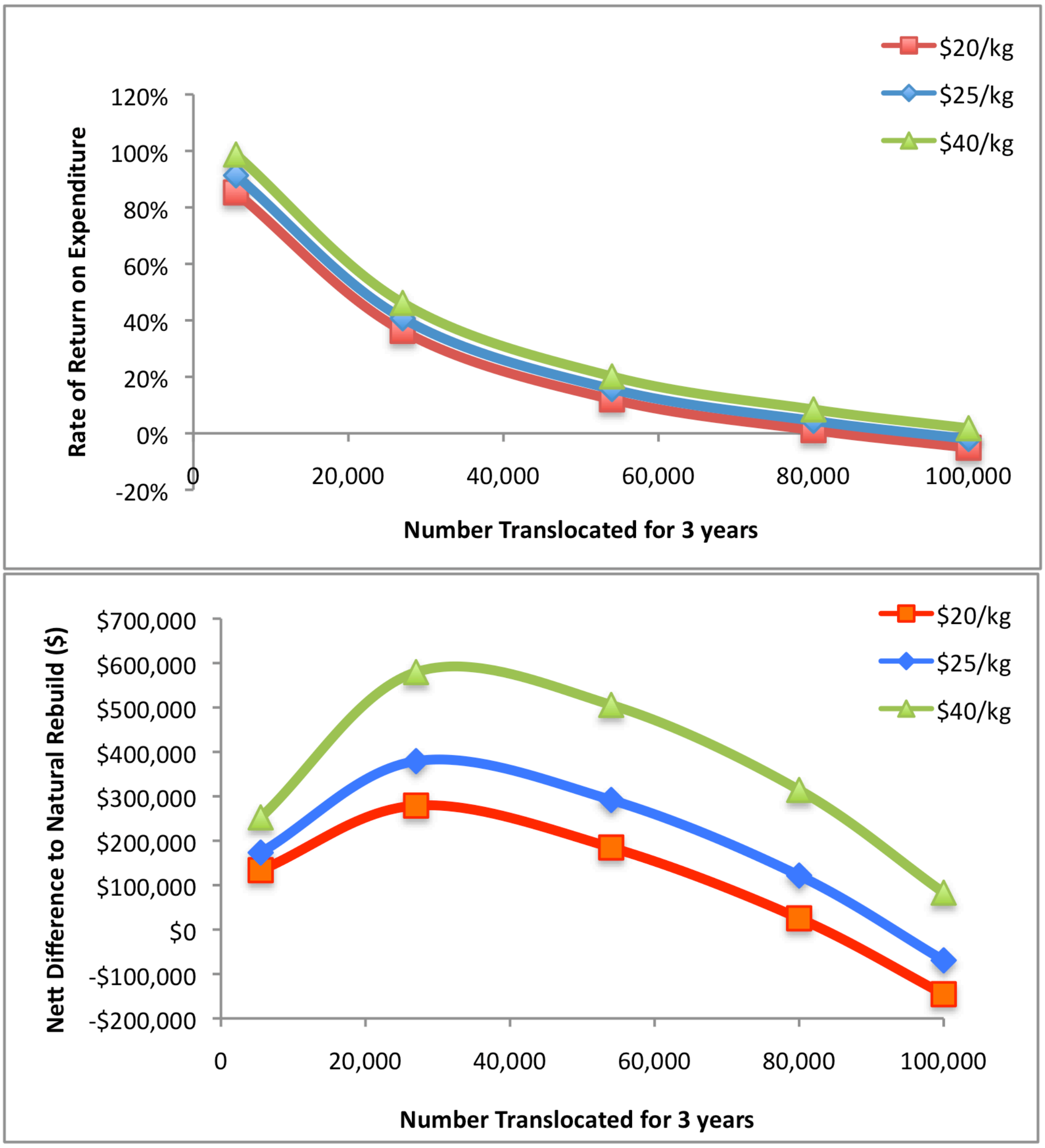


Figure 6 Estimated income ($) above Natural Rebuild for comparative three year translocation scenarios (5,500 – 100,000 adults per annum) in the left hand panel, and the estimated rate of return on costs in the right hand panel.

Comparative Cost-Benefit Analysis

For this analysis the bench-mark used to compare the differing approaches is the value of the stock at B_{MSY} its MSY equilibrium, discounted according to cost of the treatment and the years taken to return the stock to its B_{MSY} equilibrium, which in this version of the model equates to a breeding biomass of about 65t producing a sustainable yield of about 33t per annum. As observed in he fishery the capitalized value of the catch is calculated as a multiplier of the income expected each year. The 18 August 2009 WADA workshop agreed the multiplier that should be used is 7, being approximately the long term value observed in the fishery.

The model was used to estimate for each intervention scenario the time frame over which the stock is expected to return to B_{MSY}. The base case model suggests that the Natural Rebuild scenario recovers the stock to B_{MSY} by 2017, and that the primary scenarios analysed; 27,000 adults or 500,000 seed recover the stock to B_{MSY} in the third quarter and the last quarter of 2015 respectively.

Table 2 presents a basic analysis using the base case model to compare reseeding 500,000 juveniles for three years, with translocating 27,000 adults. In the left hand column the basic model parameters being analysed. In the second column are the various assumed discount rates 0.3, 0.5, 0.7 that are applied to the expense of the rehabilitation strategies and the elapse of time until the stock recovers to B_{MSY} under the differing scenarios. In the top half of the third and fourth column the deferred value of the quota and cost of rehabilitation is estimated for each scenario. In the bottom half of the last three columns the difference in Deferred Value to that estimated for the Natural Rebuild strategy is shown. In the bottom right six cells of the table the estimated capitalized difference between each treatment and the alternative of natural rebuild are shown. Using the base case model and assumptions (abalone \$25/kg and seed 40c/piece) this analysis suggests taking into account discount rates the translocation strategy would lose \$877,000-952,000 while the reseeding scenario would loose \$955,000-1,085,000.

		Natural Rebuild	**Trans 27,000 x 3y**	**Reseed 500,000 x 3y**
Multiplier on Catch	**Disc. Rate**	**Deferred Value of Stock**	**Deferred Value of Stock**	**Deferred Value of Stock**
7	0.03	$4,666,100.93	$3,788,292.66	$3,710,602.73
Abalone Price ($/kg)	0.05	$4,032,897.88	$3,137,222.64	$3,030,466.52
25	0.07	$3,474,822.53	$2,523,003.20	$2,389,818.53
Quota Value ($)		**Difference to Natural Rebuild**	**Difference to Natural Rebuild**	**Difference to Natural Rebuild**
$5,775,000	0.03	$0	-$877,808	-$955,498
Price of Seed (c)	0.05	$0	-$895,675	-$1,002,431
40	0.07	$0	-$951,819	-$1,085,004

Table 2 Tabulation of the basic analysis using the base case model to compare reseeding 500,000 juveniles for three years, or translocating 27,000 adults for three years. Model parameters are shown in the left hand column. The second column shows assumed discount rates (0.03, 0.05, 0.07) applied over the period taken to rebuild to B_{MSY}. The top half of the third, fourth and fourth column show the discounted deferred value of the stock at B_{MSY} and compounded cost of rehabilitation for each scenario and assumed discount rate. The bottom half of the last three columns show the difference in discounted deferred value to that estimated for the Natural Rebuild strategy.

Table 3 shows the sensitivity of the model's estimates of the translocation scenario to varying the assumed price of abalone. The estimated capitalized loss rises from \$730,000-797,000 at \$20/kg up to \$1.32-1.42 million at \$40/kg.

	Trans 27,000 x 3y	**Trans 27,000 x 3y**	**Trans 27,000 x 3y**
	Ab. Price $20/kg	**Ab. Price $25/kg**	**Ab. Price $40/kg**
Disc. Rate	**Deferred Value of Stock**	**Deferred Value of Stock**	**Deferred Value of Stock**
0.03	$3,002,044.34	$3,788,292.66	$6,147,037.64
0.05	$2,477,998.59	$3,137,222.64	$5,114,894.81
0.07	$1,983,147.86	$2,523,003.20	$4,142,569.22
	Difference to Natural Rebuild	**Difference to Natural Rebuild**	**Difference to Natural Rebuild**
0.03	-$730,836	-$877,808	-$1,318,724
0.05	-$748,320	-$895,675	-$1,337,742
0.07	-$796,710	-$951,819	-$1,417,147

Table 3 Tabulation of the basic analysis comparing the translocation of 27,000 adults each year for three years with the assumed abalone price varying between $20, 25 and 40/kg. The top half of the table shows the estimated discounted deferred value of the stock at B_{MSY} and compounded cost of rehabilitation for each scenario, assuming a range of discount rates (0.03, 0.05, 0.07). The bottom half of the table shows the difference in discounted discounted deferred value to that estimated for the Natural Rebuild strategy.

Table 4 shows the sensitivity of the model's estimates of the reseeding scenario to varying the assumed price of abalone and seed. The estimated capitalized loss rises from $847,000-887,000 at $40/kg and 40c/seed, up to $1.35-1.19 million at $20/kg and 60c/seed.

Sensitivity to Reseeding and Translocation Mortality Rates

Table 5 illustrates the base case model's sensitivity to varying the mortality rate assumed for translocated abalone between 0.01, 0.05 and 0.10. These three scenarios were estimated to recover to B_{MSY} by the second, third and final quarters of 2015 respectively, which is reflected in this table by the estimated capitalized loss compared to Natural Rebuild ranging between $832,000-858,000 and $922,000-1,044,000.

In Table 6 the effect of varying the assumed mortality rate of seed in the first year at liberty between 0.9, 0.75 and 0.65 is seen. This analysis uses the base case assuming abalone is worth $25/kg and seed 40c per piece. When the stock recovers to B_{MSY} after 6.5, 5.75 and 5.25 years respectively the capitalized loss from reseeding is expected to fall from $1.1 – 1.4 million with 90% release mortality, down to $864-897,000 if the mortality of seed is 65%.

	Reseed 500,000 x 3y	**Reseed 500,000 x 3y**	**Reseed 500,000 x 3y**
	Ab. Price $20/kg	**Ab. Price $20/kg**	**Ab. Price $20/kg**
	Seed 40c	**Seed 50c**	**Seed 60c**
Disc. Rate	**Deferred Value of Stock**	**Deferred Value of Stock**	**Deferred Value of Stock**
0.03	$2,741,166.03	$2,563,376.86	$2,385,587.69
0.05	$2,170,477.88	$1,971,900.52	$1,773,323.16
0.07	$1,628,863.60	$1,407,529.67	$1,186,195.75
	Difference to Natural Rebuild	**Difference to Natural Rebuild**	**Difference to Natural Rebuild**
0.03	-$991,715	-$1,169,504	-$1,347,293
0.05	-$1,055,840	-$1,254,418	-$1,452,995
0.07	-$1,150,994	-$1,372,328	-$1,593,662
	Reseed 500,000 x 3y	**Reseed 500,000 x 3y**	**Reseed 500,000 x 3y**
	Ab. Price $25/kg	**Ab. Price $25/kg**	**Ab. Price $25/kg**
	Seed 40c	**Seed 50c**	**Seed 60c**
Disc. Rate	**Deferred Value of Stock**	**Deferred Value of Stock**	**Deferred Value of Stock**
0.03	$3,710,602.73	$3,532,813.56	$3,355,024.39
0.05	$3,030,466.52	$2,831,889.16	$2,633,311.80
0.07	$2,389,818.53	$2,168,484.61	$1,947,150.69
	Difference to Natural Rebuild	**Difference to Natural Rebuild**	**Difference to Natural Rebuild**
0.03	-$955,498	-$1,133,287	-$1,311,077
0.05	-$1,002,431	-$1,201,009	-$1,399,586
0.07	-$1,085,004	-$1,306,338	-$1,527,672
	Reseed 500,000 x 3y	**Reseed 500,000 x 3y**	**Reseed 500,000 x 3y**
	Ab. Price $40/kg	**Ab. Price $40/kg**	**Ab. Price $40/kg**
	Seed 40c	**Seed 50c**	**Seed 60c**
Disc. Rate	**Deferred Value of Stock**	**Deferred Value of Stock**	**Deferred Value of Stock**
0.03	$6,618,912.83	$6,441,123.66	$6,263,334.49
0.05	$5,610,432.44	$5,411,855.08	$5,213,277.72
0.07	$4,672,683.34	$4,451,349.42	$4,230,015.49
	Difference to Natural Rebuild	**Difference to Natural Rebuild**	**Difference to Natural Rebuild**
0.03	-$846,849	-$1,024,638	-$1,202,427
0.05	-$842,204	-$1,040,782	-$1,239,359
0.07	-$887,033	-$1,108,367	-$1,329,701

Table 4 Comparison of reseeding 500,000 adults per year for three years with an assumed abalone price of $20, 25 and 40/kg and an assumed price of seed varying between 40, 50 and 60c and various discount rates. For each scenario, using a range of discount rates (0.03, 0.05, 0.07), estimates of discounted deferred value of the stock when it recovers to B_{MSY} reduced by the compounded cost of rehabilitation are shown, above the estimated difference in discounted deferred value to that estimated for the Natural Rebuild strategy.

	Trans M = 1%	Trans M = 5%	Trans M = 10%
	Trans 27,000 x 3y	Trans 27,000 x 3y	Trans 27,000 x 3y
	Ab. Price $25/kg	Ab. Price $25/kg	Ab. Price $25/kg
Discount Rate	Deferred Value of Stock	Deferred Value of Stock	Deferred Value of Stock
0.03	$3,833,695.94	$3,788,292.66	$3,743,112.75
0.05	$3,208,202.44	$3,137,222.64	$3,066,777.81
0.07	$2,616,604.05	$2,523,003.20	$2,430,291.02
	Difference to Natural Rebuild	Difference to Natural Rebuild	Difference to Natural Rebuild
0.03	-$832,405	-$877,808	-$922,988
0.05	-$824,695	-$895,675	-$966,120
0.07	-$858,218	-$951,819	-$1,044,532

Table 5 Comparison of translocating 27,000 adults each year for three years assuming the mortality of translocated abalone is 1%, 5% or 10%. The top half of the table shows the estimated discounted deferred value of the stock when it recovers to B_{MSY} reduced by the compounded cost of rehabilitation for each scenario and assuming a range of discount rates (0.03, 0.05, 0.07). The bottom half shows the difference in discounted deferred value to that estimated for the Natural Rebuild strategy.

	Reseed M = 90%	Reseed M = 75%	Reseed M = 65%
	Reseed 500,000 x 3y	Reseed 500,000 x 3y	Reseed 500,000 x 3y
	Ab. Price $25/kg	Ab. Price $25/kg	Ab. Price $25/kg
	Seed 40c	Seed 40c	Seed 40c
Discount Rate	Deferred Value of Stock	Deferred Value of Stock	Deferred Value of Stock
0.03	$3,575,648.58	$3,710,602.73	$3,801,662.87
0.05	$2,820,875.37	$3,030,466.52	$3,172,766.25
0.07	$2,114,613.52	$2,389,818.53	$2,577,477.81
	Difference to Natural Rebuild	Difference to Natural Rebuild	Difference to Natural Rebuild
0.03	-$1,090,452	-$955,498	-$864,438
0.05	-$1,212,023	-$1,002,431	-$860,132
0.07	-$1,360,209	-$1,085,004	-$897,345

Table 6 Comparison of reseeding 500,000 juveniles each year for three years assuming the mortality of reseeded abalone is 1%, 5% or 10%. The top half of the table shows the estimated discounted deferred value of the stock when it recovers to B_{MSY} reduced by the compounded cost of rehabilitation for each scenario and assuming a range of discount rates (0.03, 0.05, 0.07). The bottom half shows the difference in discounted deferred value to that estimated for the Natural Rebuild strategy.

Sensitivity to the Timing of Rehabilitation

In the context of earlier analysis suggesting that measured interventions that moved the stock up the SRR towards B_{MSY} were more profitable, the model was also used to examine its sensitivity to the timing of the rehabilitation. Given the context of WADA's discussion in all previous scenarios rehabilitation has been assumed to commence in 2010 three years during the third year of Natural Rebuilding, and 2010 has been assumed as the zero point for calculating the effect of economic discount rates. In this final sensitivity analysis (Table 7), the rehabilitation strategy is assumed to commence in 2008 the year following the modeled viral impact. In these final scenarios 2008 is also assumed to be the zero point for applying discount rates, necessitating the re-estimation of the value of the previous base case scenario.

The results of this analysis shown in Table 7 show that rehabilitation will be more cost-effective if commenced early so that the stock is more quickly moved towards more productive levels of breeding biomass. This is particularly true for the translocation strategy which adds directly to the breeding biomass. However, the model still estimated that these forms of rehabilitation are not cost-effective enough to cover the cost of the capital involved.

	Trans 27,000 x 3y	Trans 27,000 x 3y	Reseed 500,000 x 3y	Reseed 500,000 x 3y
	Begin Rehab. in 2010	Begin Rehab. in 2008	Begin Rehab. in 2010	Begin Rehab. in 2008
Discount Rate	Deferred Value of Stock	Deferred Value of Stock	Deferred Value of Stock	Deferred Value of Stock
0.03	$3,432,891.52	$3,925,183.04	$3,354,916.42	$3,756,022.00
0.05	$2,587,700.40	$3,351,812.77	$2,481,100.70	$3,101,351.57
0.07	$1,803,745.00	$2,806,591.40	$1,670,766.33	$2,483,209.53
	Difference to Natural Rebuild	Difference to Natural Rebuild	Difference to Natural Rebuild	Difference to Natural Rebuild
0.03	-$957,443	-$740,918	-$1,035,418	-$910,079
0.05	-$1,051,990	-$681,085	-$1,158,590	-$931,546
0.07	-$1,201,629	-$668,231	-$1,334,608	-$991,613

Table 7 Comparison of commencing each rehabilitation strategy (reseeding 500,000 juveniles x 3 yrs or translocating 27,000 adults x 3 yrs) either 1 year after the viral impact (2008) or three years after the viral impact (2010). The top half of the table shows the estimated discounted deferred value of the stock when it recovers to B_{MSY} reduced by the compounded cost of rehabilitation for each scenario and assuming a range of discount rates (0.03, 0.05, 0.07). The bottom half shows the difference in discounted deferred value to that estimated for the Natural Rebuild strategy.

Discussion

A clear result that comes from both the preliminary and final analysis is the importance of matching the scale of rehabilitation to the scale and productivity of the stock. These analyses clearly show that early interventions scaled to move the stock up the steep part of the SRR curve toward B_{MSY} are likely to be more profitable than larger, later interventions.

Interestingly both analyses also suggest that there is very little difference in cost effectiveness between the two rehabilitation techniques. The descriptive analysis which ignored the cost of capital and simply estimated costs of rehabilitation and comparative expected income through the period 2011-2025 suggested that across the range of pricing scenarios used the translocation strategies were more likely to produce a level of income in excess to that expected under a Natural Rebuild, and unlikely to incur the large losses or incomes, predicted under some reseeding and pricing scenarios. The main difference between reseeding and translocation strategies in this respect resulted from differentials in pricing between the price of seed and abalone. Thus translocation scenarios incorporated the price of abalone as a basis for costing the rehabilitation and estimating expected incomes, buffering the eventual result. In contrast the range of reseeding scenarios estimated costs on the basis of the cost of seed, and income on the value of abalone, thus scenarios incorporating a low price of seed and high price of abalone were likely to produce high estimated incomes, and high seed cost / low abalone price scenarios could produce high estimated losses.

The major result from this analysis is that taking into account the cost of capital by incorporating discount rates none of the rehabilitation strategies analysed here appear to be cost effective (Table 2-6). Ignoring the cost of capital and simply accounting for stock productivity and costs, figures 3 and 6 suggest that the returns from reseeding could exceed costs if the price of abalone approaches $40/kg. While most translocation scenarios should return greater income than costs at any of the abalone prices considered. However, taking into consideration the actual cost of capital, all the scenarios analysed in detail produce lower returns than the Natural Rebuild scenario.

This is a similar, although more extreme, result to that produced by the preliminary analysis which found that only if the highest discount rate was assumed (0.07) were some reseeding and translocation scenarios likely to be profitable. However, while the estimated time taken to recover to B_{MSY} was similar for both models, in the preliminary analysis only the cost of the translocated abalone and the seed was taken into account.

Depensatory Effects

The model used in this analysis suggests that natural rebuild scenario recovers the stock to B_{MSY} by 2017 while only the translocation scenario with 1% mortality was capable of recovering the stock to this level before 2015. The estimated difference in timing between Natural Rebuild and rehabilitation is just too small to pay for the real cost of the rehabilitation strategies. This outcome is driven entirely by the resilience of the assumed Stock Recruitment Relationship (SRR) at low biomass levels. The model is built to predict that the resource is rebuilding rapidly after the virus attack and consequently investing in either seed or translocation seems to produce relatively little extra benefit.

The standard form of the Stock Recruitment Relationship (SRR) used in Australian stock assessment is used in this analysis. The two most important features of this standard parameterization of the SRR for this analysis are:

C. The density dependence is between adult stock and recruitment, rather than as competition for crevice species by juvenile year classes.
D. The SRR implies increasing productivity per unit of spawning biomass as spawning biomass approaches zero levels.

This analysis proceeded on the basis of applying the same accepted SRR to all scenarios so that within the terms of the study no treatment is unfairly advantaged because the assumptions are uniform across all scenarios. However, the explanatory form of analysis performed here, while consistent with the stock dynamics modeled more formally with the Australian Abalone model structure, does not exclude other underlying and as yet unobserved dynamics. Almost nothing is known about the actual relationship between stock and recruitment (SRR) for abalone anywhere in the world (Campbell 2000). Only a few experimental SRR studies have been performed by Shepherd and Partington (1995) in South Australia with greenlip abalone and the non commercial species *H. scalaris*, and no studies have occurred anywhere in the world on any species, at the scale of a reef code like Kilarney.

In this analysis the assumed resilience at low biomass becomes problematic in driving the results of the analysis so strongly, because it is in stark contrast with the belief of many abalone ecologists who believe that at low biomass levels abalone populations become subject to depensatory effects which reduce the effectiveness of spawning causing lower than expected recruitment from the remaining biomass (Liermann & Hilborn 1997). Tegner et al. (1989) was the first to touch on "reduced fertilization efficiency" as one of a number of possible explanations for the observed decline in Californian landings. Although several Australian studies have become the mainstay of this concern (Shepherd & Brown 1993, McShane 1995, Shepherd & Partington 1995, Babcock and Keesing 1999). Two primary mechanisms are proposed for depensation at low biomass levels.

3. Some abalone ecologists (Shepherd & Brown 1993, McShane 1995, Babcock and Keesing 1999) emphasize the potential for low fertilization success in sparse fished populations where aggregations are fished before they can spawn reducing the density at which eggs and sperm are mixed in the water column during spawning reducing fertilization success.

4. The other school of concern (Miner et al. 2006) involves the conditioning of clean coralline algal substrates for the successful settlement of abalone larvae, through the grazing pressure of adult abalone and other demersal herbivores. When abalone populations are rapidly depleted much of the coralline algal surfaces previously maintained through abalone grazing pressure is overgrown over month and years by fleshier algal and sponge biota, reducing the availability of settlement sites for juvenile abalone.

The extent to which these factors result in depensation and alter the assumed shape of the abalone SRR curve is uncertain. Some studies (Shepherd 1986; Prince 1992; Shepherd & Partington 1995) emphasize the mobility of individual abalone in and out of aggregation and the ability to actively reform breeding aggregations if left undisturbed or fished lightly enough suggesting that unless heavy fishing is allowed through the spawning period abalone should be able to aggregate sufficiently to ensure optimal fertilization. While the CCA settlement surfaces are known to be created and maintained by the grazing of many benthic herbivores including abalone (Morse et al. 1979) so that growing aggregations of abalone might be expected to reestablish settlement surfaces around them as the aggregation grows and needs more settlement surfaces. Shepherd & Partington (1995) argue that it is the interplay of these effects that creates the standard form of the Ricker SRR that they described and which at low biomass is similar in shape to that used in this study.

To the extent depensation changes its shape, the abalone SRR might be expected to have a flatter approach to the origin than suggested by that used in this study. If this were the case the SRR curve would dip down low at some critical level of breeding biomass above zero. Below that threshold level of Breeding Biomass the productivity of each unit of biomass would be lower and the recovery of the stock would be very much slower than suggested by the SRR curve used. Incorporating a depensatory effect into the SRR curve would disproportionately reward any strategy that rapidly builds breeding biomass to above the threshold level, so enhancing the stock's own natural productivity. So a natural rebuild strategy would be penalized if it left a stock below that threshold. The difficulty for this analysis is that while the depensatory effects are surmised, there has been so little research on this aspect of abalone fisheries ecology (Campbell 2000) that there is no basis for deciding what level of biomass would constitute the threshold, nor of the reduced level of productivity that might occur below that threshold. Consequently attempting to capture this aspect of abalone ecology in this modeling exercise would necessitate incorporating a wide range of assumptions that have no defendable basis. In that case it is likely that built in assumptions would completely overwhelm the 'scientific data' incorporated into the model.

As unsatisfactory as this maybe; the only possible choice at this junction is to acknowledge that this factor is in no way incorporated into the analysis. This is a necessary weakness of the analysis and as a consequence caveats need to be placed upon its results. To the extent that depensatory effects exist for abalone at Kilarney this analysis will tend to favor the cost effectiveness of allowing natural rebuild, over the alternative interventions. But without understanding the phenomena better we cannot even guess whether this effect is significant or not.

In this context comments made by divers who have been observing the virus impacted areas recover at the 13 August, 2009 WADA workshop were of great interest. Several of the divers had originally in 2006 been supportive of rehabilitation in the most heavily impacted areas because they like most involved with the issue could not imagine these areas of stock recovering naturally. Implicitly most people were expecting some form of depensatory effect. So it was of great interest that at the WADA workshop in 2009 those same divers reported that in their opinion even the most depleted areas were visibly recovering now, and that they were now no longer convinced intervention was necessary. This might be anecdotal evidence that the assumed form of the SRR may be more applicable than feared.