Roadblocks to the adoption of economics in fisheries policy

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Executive Summary

This project has led to the development of three journal articles examining how the use of economic analyses and stock enhancement can lead to improved economic outcomes in Australian wild-capture commercial fisheries. The Seafood Cooperative Research Centre (Seafood CRC) Future Harvest (FH) projects identified some of the challenges and opportunities associated with implementing bio-economic approaches and stock enhancement within fisheries management frameworks. Much of this discourse was contained however in technical reports, newsletters and other project-linked documentation (e.g. milestone reports). Thus there was a need (and space) to document the adoption of bio-economics and stock enhancement within fisheries management frameworks, associated challenges and the process of change management in Australian fisheries within peer-reviewed journal articles.

The first journal article used a variety of case-studies to illustrate how economic analyses and instruments can improve economic performance in Australian wild-capture commercial fisheries. It discussed how economic objectives are prevalent in Australian legislation but economic data is often not routinely collected and economic analyses or instruments inconsistently applied across jurisdictions. Three large-scale commercial fisheries were used as case-studies of how bio-economic analyses can improve profitability, including: the Tasmanian Southern Rock Lobster Fishery, Western Australian Rock Lobster Fishery and Commonwealth Northern Prawn Fishery. Two small-scale commercial fisheries were also used as case-studies examining how empirical economic analyses can improve profitability for minimal cost, including: the Lakes and Coorong Pipi Fishery and the Western Australian Shark Bay Prawn Trawl Fishery. Challenges to the implementation of bio-economics and empirical economic analyses were also discussed, including: (i) short-term transition costs and associated trade-offs between ecological, economic, social and political objectives; (ii) scarce logistical and financial capacity to collect and analyse economic data; (iii) a lack of desire among industry to change and transition to economic targets such as MEY, particularly when it is associated with lower catches; and (iv) a lack of economic literacy among fisheries managers and industry. To an extent these can be overcome through clearly defining and articulating objectives initially and identifying stakeholder priorities and trade-offs for the fishery.

The second article discussed how the introduction of economic instruments, such as individual transferable quotas (ITQs), may not maximise the profitability of commercial fisheries due to residual externalities, such as assignment problems. This could lead to economic objectives not being met. The article stipulated that self-governing fishing cooperatives might be able to resolve these externalities through collective action but in Australia there are a number of barriers to this within ITQ fisheries including: (i) a breakdown of social capital among fishers caused by the privatisation of fishing rights, (ii) increasing heterogeneity in business structures, incentives and motivations among fishers due to the free transferability of quota units. Both of which lead to increased transaction costs of collective decision-making among fishers and reluctance among government to create enabling legislation to authorise and legitimise fisheries with the right to organise and self-govern. Case-studies to illustrate these barriers in the Tasmanian Southern Rock Lobster Fishery and other fisheries were considered. The article suggested that when there is an inability among industry to self-govern or reach collective agreement on issues that improve the profitability of the entire fishery, government needs to be proactive in implementing changes and have the willpower to see those changes through to fruition.

The third article examined the potential for stock enhancement among high value invertebrates to improve fishery profitability. Using recent examples from Australian abalone, rock lobster and beche de mer it was highlighted that commercially-viable enhancement programs have a number of key challenges that must be overcome in order to proceed. These include: adequate technical capacity in the areas of hatchery production and wild release methodologies as well as sufficient governance capacity that accounts for complexities and the interdisciplinary nature of stock enhancement. It was emphasised that successful stock enhancement could lead to substantial changes in the focus of fisheries management as managers have more control over recruitment levels and resulting exploitable biomass.

Once the papers are published in peer-reviewed journals they will be distributed to fisheries management agencies information to improve awareness of the implementation of Seafood CRC FH initiatives, barriers to their uptake in fisheries management frameworks and how these might be overcome. They will also potentially be used in the various fisheries economics training courses (e.g. FishEcon Masterclasses) and other workshops.

Keywords

Bio-economics, stock enhancement, wild-capture fisheries, MEY, MSY, economics, government, industry, fisheries management, cooperatives, self-governance, translocation, re-seeding

Introduction

The Seafood CRC FH theme projects have highlighted the ability of bio-economic and empirical modelling as well as stock enhancement (including re-seeding and translocation) to increase economic returns within commercial wild-capture fisheries above that which would be possible using standard fisheries management approaches. Most of the results and subsequent discussion of these outputs have been confined however to technical reports, newsletters and other project-linked documentation (e.g. milestone reports) and not referred literature. While industry-based initiatives place little value in journal articles, there is a need (and space) to critically analyse and document in peer-reviewed journals the use of bio-economics and stock enhancement within Australian fisheries management frameworks, associated challenges and how these have been resolved. These articles can then be disseminated to fisheries managers and utilised within manager training courses and workshops to improve knowledge, understanding and ultimately management decision-making around economic objectives. Scientific papers, while allowing for the exchange of ideas globally, also enable managers to defend decision-making on the basis of the peer-review system.

Objectives

- Examination of the process of changing management in Australia resulting in the increased use of economics

- Increase factual awareness of the potential for enhancement in Australia

Journal Article 1:

Incorporating economics into fisheries management frameworks

Abstract

There is a large gap between the current and optimal economic performance of wild-capture commercial fisheries in Australia. Economics has the potential to assist fisheries to bridge this gap through, for example, the use of bio-economic models that take into account not just the biology of the stock but the cost structure of fishing in determining the optimal level of harvest. Despite economic objectives being prevalent in overarching Australian fisheries legislation, economic data is often not collected and economic analyses or instruments are not broadly applied in fisheries management frameworks. This paper reviews selected Australian fisheries to demonstrate the accrued economic benefits through either the use of formal bio-economic models or empirical analyses of the relationship between fishing effort and market prices. Challenges to the implementation and continued use of economic analyses and instruments are discussed including: (i) short-term transition costs and associated trade-offs between ecological, economic, social and political objectives; (ii) scarce logistical and financial capacity to collect and analyse economic data; (iii) a lack of desire among industry to change and transition to economic targets such as MEY, particularly when it is associated with lower catches; and (iv) a lack of economic literacy among fisheries managers and industry. It is contended that many of these challenges initially arise from an absence of clearly identified and prioritised objectives within overarching legislation and management plans. Once objectives are prioritised, limited resources can be allocated more efficiently within that fishery to improve data collection, economic analysis and increase awareness, education and training of managers and industry.

Introduction

Many countries manage their fisheries resources to achieve an assortment of ecological, economic, social and political objectives (Hilborn, 2007, Dichmont et al., 2010) . For example, Australia has adopted a national ecological sustainable development (ESD) framework in legislation for managing its fisheries, establishing a requirement to consider ecological, economic and social implications when setting catches of target species. This requirement is reflected in the Commonwealth *Fisheries Management Act 1991* (FMA 1991), which stipulates that the management agency must ensure "that the exploitation of fisheries resources and the carrying on of any related activities are conducted in a manner consistent with the principles of ecologically sustainable development". While most countries have similar broadly phrased goals or higher order objectives within overarching legislation, these are often not defined in any operational manner within fisheries management plans and policies in such a way that allows them to be measured and attained (Begg et al., 2014, Cochrane, 2000, Pascoe et al., 2013b).

According to Lackey (1998) management objectives must be: (i) explicit and specific, with no broad generalisations; (ii) quantifiable, if not empirically then at least subjectively; (iii) contain measure(s) to evaluate performance and; (iv) have enough flexibility to respond to changes in societal preferences and ecological constraints. Thereupon, objectives can direct the decision-making process by delivering a mandate to fisheries managers, while providing a means of identifying potential conflicting activities, evaluating performance and measuring results (Barber and Taylor, 1990, Pascoe et al., 2014a).

Fisheries objectives, however are often countervailing and cannot be simultaneously optimised (Pope, 1997). For example, the New South Wales *Fisheries Management Act 1994* stipulates that the objects of the Act include, the promotion of “viable commercial fishing and aquaculture industries”, while also ensuring “quality recreational fishing opportunities” and providing “social and economic benefits for the wider community of New South Wales.” This example illustrates how disagreement and conflict among stakeholders may arise if trade-offs are not considered and priorities agreed upon prior to decision-making.

There is often considerable difference of opinion among stakeholder groups as to the relative importance of particular objectives and the values they place on the resource (Pascoe et al., 2009, Pascoe et al., 2013b, Hanna and Smith, 1993). For example, Pascoe et al. (2009) in a study of objective preferences among stakeholder groups within Australian Commonwealth fisheries showed how industry stakeholders prioritised economic objectives, environmental stakeholders prioritised ecological objectives relating to minimising environmental impacts, while social scientists and recreational fishing groups prioritised social objectives relating to minimising externalities. In the absence of well-specified and prioritised objectives therefore, decision-makers will be left uncertain as to how best to make fisheries management decisions that balance objectives, as well as know what is expected of them (Cochrane, 2000, Pope, 1997, Pascoe et al., 2009). This can lead to reactive, short-term decision-making and the prioritisation of a particular issue above others for political reasons (Cochrane, 2000). Principal-agent costs may then be incurred when the manager’s (agent) decision do not effectively represent the interests of the owners or society at large (principal) (Jensen and Meckling, 1976, Smith, 1776). Empirical analyses indicate that these costs can be considerable (Mardle and Pascoe, 1999), with various authors attributing widespread failures in fisheries management to inappropriate consideration and prioritisation of objectives (Hilborn, 2007, Cochrane, 2000).

Historically, the focus of management decision-making in fisheries has been based on achieving ecological objectives relating to maximising sustainable production (Ward and Kelly, 2009, Pascoe et al., 2009, Pascoe, 2006). This was a product of the inherent values of most fisheries managers, who had a background and education in biological sciences (Ward and Kelly, 2009, Barber and Taylor, 1990) as well as the United Nations Law of the Sea Convention (LOSC) and other international conventions that obligated fisheries managers to set catches for target species at the maximum sustainable yield (MSY). Economic objectives, such as maximising the economic efficiency of the fishery are gaining increasing importance however, in driving the outcomes of management decision-making (Norman-López and Pascoe, 2011, Pascoe et al., 2009, Pascoe, 2006). This is in part due to the general desire to increase fishery profitability by rebuilding wild capture fish species to more precautionary (and economically efficient) targets to offset the rising costs of fuel and declines in fish prices caused by fluctuating exchange rates and increased aquaculture production (Reid et al., 2013, AFMA, 2007). Additionally, as competition over marine resource space intensifies, there is increased pressure to make commercial fisheries more profitable so fishers are not economically outcompeted during negotiations by other industry groups advocating alternative uses (Pascoe, 2006).

With this shift in focus has been an increased interest in incorporating economic analyses and instruments into fisheries management frameworks (Dichmont et al., 2010, Norman-López and Pascoe, 2011). This is reflected in Australia, particularly at the Commonwealth level with the release of Ministerial Direction to the Australian Fisheries Management Authority (AFMA) in 2005. The Ministerial Direction required AFMA to transition all managed fisheries to individual transferable quotas (ITQ) (unless this could be proven to be not cost effective) and measure the impact of its decisions on the economic performance of its fisheries through the development of a harvest strategy policy applied to all targeted stocks (DAFF, 2005). The harvest strategy policy was implemented in 2008 and required fisheries managers to set the biomass corresponding to the maximum economic yield (B_{MEY}) as the default target reference point in order to meet the revised economic objective within the *Australian Fisheries Management Act 1991* of “maximising the net economic returns to the Australian community from the management of Australian fisheries” (Rayns and Read, 2007).

While the application of economic instruments, such as ITQs are expanding within Australia, the objective (or reasoning) behind their use remains foremost ecological (Gardner et al., 2015a, Gardner, 2012). For example, the impetus behind the Australian Commonwealth Government's Ministerial Direction to AFMA and adoption of ITQs as the preferred management system was foremost to recover overfished stocks (of which there were 24 in 2005) and to prevent future overfishing in Commonwealth-managed fisheries (Smith et al., 2008, Grafton et al., 2007). Likewise in the Tasmanian southern rock lobster fishery, ITQ management was introduced in 1998 on the basis of needing to "constrain catch to a sustainable level" following a prolonged period of stock decline (Gardner, 2012, Ginn et al., 1997). Similarly, in the Western Australian rock lobster fishery the main impetus to reduce effort in the fishery and transition towards a maximum economic yield (MEY) target was to address low recruitment levels and achieve stock conservation objectives (Reid et al., 2013, Caputi et al., 2015).

While the justification for the implementation of economic instruments or use of economic analyses within Australian fisheries may primarily be to achieve ecological targets, the resulting economic benefits from optimal levels of harvest create further justification for their continued adoption (Gardner et al., 2015a, Caputi et al., 2015, Dichmont et al., 2010). According to Pascoe (2006) economics can provide an important framework for management of the marine environment through: (i) valuation of non-market environmental services; (ii) assessing the performance and outcomes of different management measures and policies; and (iii) through the use of economic instruments (such as ITQs), provide a means of ensuring efficient allocation of resources. Bio-economic modelling, in particular has been advocated as an important tool in managing fisheries for economic and ecological sustainability (Gardner, 2012). Bio-economic models recognise that the optimal harvest is determined not only by the biology of the stock but also the cost structure of the fishery and the value of the harvest (Larkin et al., 2011). By combining factors that influence the biological side (e.g. stock recruitment relationship) and economic side (e.g. prices and fixed and variable costs of fishing), decision-makers can model interrelationships to provide an insight into the optimal management of the stock (Prellezo et al., 2012, Larkin et al., 2011). Optimal management is often considered a value-based objective such as MEY. MEY occurs at the effort level (E_{MEY}) and corresponding revenue value (R_{MEY}) that creates the largest difference between the total costs of harvesting and the revenues obtained from the catch (Kompas, 2005). On Figure 1, MEY is identified as the point where the slope of the revenue curve is equal to the slope of the cost curve (Norman-López and Pascoe, 2011). More often than not, this point occurs at effort levels less than at MSY (particularly as the costs of fishing increase) and results in a larger overall stock size. This has led to various economists advocating the ecological benefits of a MEY target relative to MSY, as it provides further protection against unexpected and undesirable reductions in stock size caused by environmental variation (Kompas, 2005). While a MEY target may result in an overall higher stock size, in most instances it requires an initial reduction in fishing effort to enable stock rebuilding. A rapid transition to a lower level of fishing effort and a MEY target may have a number of short-term consequences (e.g. reductions in employment) for the fishery and will also vary through time as market prices and harvesting costs change (Grafton et al., 2012). Consequently, trajectories, timeframes and trade-offs associated with transitioning to a MEY target need to be understood and regularly evaluated through bio-economic analyses. It is only then that stakeholders can make informed decisions as to their preferred transition to MEY that balances competing objectives.

While bio-economic models that explicitly consider economic factors in harvest controls rules (e.g. MEY) have been developed for various fisheries internationally, most are not directly used for management purposes (Punt et al., 2010). This paper will discuss some Australian fisheries where economic considerations have been incorporated into management decision-making, through either the use of formal bio-economic models or empirical analyses of the relationship between fishing effort and market prices. The accrued economic benefits to these fisheries will be highlighted as justification for the widespread adoption of economic analyses within other fisheries to identify and reduce the gap between current and optimal fisheries performance. Consideration will also be given as to why economic analyses such as bio-economics are not more broadly applied to assist management decision-making, despite implicit or explicit economic objectives being prevalent in overarching Australian jurisdictional fisheries legislation.

Case studies from Australian fisheries

Tasmanian rock lobster fishery – insightful ways of increasing value

Bio-economic analysis has been utilised in the Tasmanian southern rock lobster (TSRL) fishery in Australia as a way of increasing the economic value of the fishery through informing TACC setting and approaches to spatial management. The TSRL fishery is a single species pot fishery targeting the spiny lobster *Jasus edwardsii* with a total annual revenue (i.e. gross value of product [GVP]) of $59 million in 2011/12 (Hartmann et al., 2013). Despite regional heterogeneity in the biology of the stock (caused primarily by water temperature impacting lobster growth and reproduction), the fishery is managed by a single TACC under an ITQ system. Following the introduction of ITQs in 1998, the fishing fleet rationalised and catch rates increased (Hamon et al., 2009). In the late 2000s however, an unprecedented period of poor growth of legal sized stock and recruitment of new lobsters led to an increase in fishing effort, decline in exploitable biomass and a consequent reduction in catch rates. Due to an inability among stakeholders to agree to sufficient reductions the TACC was non-binding (i.e. TACC set too high) for a period of three fishing seasons (Emery et al., 2014a).

The sudden declines in exploitable biomass precipitated the need to decrease the TACC. Bio-economic analysis of the fishery was undertaken using a population dynamics model, representing the size and sex structure of the stock across eleven sub-zones (See Figure 2) and an economic model, calculating the discounted profits of the fishery across alternative TACC scenarios (Gardner et al., 2015a). Inputs to the population dynamics and economic models are described in more detail in Gardner et al. (2015a). Model projections indicated that the current TACC was in excess of that which would produce MEY (assuming long-term average recruitment) and that economic yield was highly responsive to changes in the TACC. Even small annual changes (1 – 4% per annum) were shown to have the potential to improve fishery biological performance in terms of exploitable biomass, catch rates and egg production (Gardner et al., 2015a).

The outputs from the bio-economic model were presented to industry and were critical to securing agreement to support a reduction in the TACC and develop new target reference points to guide the TACC setting process. New target reference points included: (i) rebuilding state-wide biomass to the 2005/06 level (the most recent peak in stock abundance) by 2021 with >70% probability and; (ii) rebuilding the state-wide catch rate to 1.2 kilograms per potlift by 2021 and area catch rates equivalent to the 2005/06 level. While there remains no explicit MEY objective within overarching legislation or regulatory harvest strategy framework, the chosen target reference points for exploitable biomass and catch rate approximated a stock rebuilding pathway towards maximising the economic yield of the fishery (Gardner et al., 2013).

The use of bio-economics to improve the yield of the TSRL fishery has not been limited solely to TACC setting. The population dynamics and economics model(s) have also been used to assess whether: (i) regional size limits aligned with local growth rates could increase fishery yield (Gardner et al., 2014) and; (ii) commercial scale translocation of lobsters could improve the economic profitability of the fishery (Gardner et al., 2015b).

In Tasmania, there are large spatial differences in the biology of lobsters, with those inhabiting the southern areas of the State, growing slower and reaching sexual maturity at an average smaller size than those in the northern areas (Punt et al., 1997). Gardner et al. (2015a) using the aforementioned bio-economic model examined whether egg production could be increased in the fishery through the institution of minimum size limits more attuned to the local biology of the stock, than the then current state-wide male and female limits of 110 and 105 mm carapace length (CL) respectively. Four alternative size limit scenarios were examined in terms of changes in egg production, biomass and economic indicators: (i) status quo; (ii) 110mm CL size limit statewide for females; (iii) 110mm CL size limit for females in all areas, except southern areas (1, 7 and 8); and (iv) 105mm CL size limits in southern areas (1 – 3 and 8) , 110 mm CL in mid-west coast (6 and 7) and 130mm CL in northern areas (4 and 5). All alternative scenarios resulted in modest improvements to egg production (1 – 4% increases after 10 years) but decreases in economic profitability due to reductions in the exploitable biomass. Increased egg production was greatest for scenario (ii) and (iv) but their regional impact was divergent. While increasing egg production in the already productive southern areas, scenario (ii) reduced the probability that egg production would be above target reference points in northern areas of the state. In contrast, scenario (iv) improved the probability that egg production would be above the target reference point in all areas.

Gardner et al. (2015b) used the bio-economic model to examine the outcome of translocating 100,000 lobsters in the TSRL fishery in the context of the existing management system. Previous research had highlighted how slow-growing lobsters from deep-water areas when translocated, experienced increased growth, egg production, changed colour and body shape, stayed at the release site and suffered no increased mortality (Chandrapavan et al., 2010, Green and Gardner, 2009, Chandrapavan et al., 2009). The outcomes from this research were used as inputs to the bio-economic model to assess the economic benefit of three translocation options for the fishery: (i) southern translocation, moving lobsters from deep water areas directly inshore; (ii) broad translocation, moving lobsters from the same areas inshore but also northwards and; (iii) northern translocation moving lobsters from the same areas as far north as possible (area 5). Translocation increased legal sized biomass and catch rates of lobsters equivalent to a 4.5% and 7% reduction in the TACC under the broad translocation and northern translocation option(s). The increased catch rate combined with a constant TACC was equivalent to a $17, $26 and $38 million increase in cash flow after 15 years, or 4.9%, 7.4% and 10.9% increase in profitability above that achieved without translocation for each of the three respective options.

Each of these examples illustrates how bio-economics has been used in an innovative way to assess the economic outcomes of different management scenarios, with the overall aim of increasing the value of the TSRL fishery. While an agreement was reached among industry to support a reduction in the TACC and adopt MEY driven target reference points (Gardner et al., 2015a), there has been a lack of consensus around regional size limits. Regional size limits more attuned to the biology of the stock could improve future economic yield but there have been difficulties in reaching agreement to support changes in boundaries and concerns surrounding enforcement, with commercial scale translocation seen as a more attractive option (Cartwright, 2013). While regional size limits remain a viable option for future consideration industry has been actively pursuing translocation of lobsters from deep-water to shallow water through the adoption of a three year commercialisation trial of the broad translocation option (Gardner et al., 2015b). Central to achieving majority support for pursuing translocation was the explicit avoidance of a reduction in TACC that would have been necessary to fulfil the requirements of the harvest strategy.

Western Australian rock lobster fishery – a fortuitous shift to MEY

Bio-economic analysis also supports the TACC decision-making process in the Western Australian rock lobster (WARL) fishery with the aim of optimising the economic utilisation of the fishery, while ensuring the maintenance of ecological sustainability (Reid et al., 2013). Worth an estimated $200 – $400 million annually, the WARL fishery targeting the Western Australian rock lobster (*Panulirus cygnus*) is the most valuable single-species fishery in Australia (Caputi et al., 2015, Reid et al., 2013). Historically managed under input controls and later a total allowable effort (TAE), catches averaged around 11,000 tonnes up to the mid-2000s (Reid et al., 2013). Since the 2010/11 fishing season however, the fishery has been managed under output controls and individual transferable quotas (ITQ), with recent catches averaging 5,000 – 6,000 tonnes (Caputi et al., 2015).

In 2008, a MEY assessment of the WARL fishery was undertaken using a length-based, spatial population dynamic model to assess a range of fixed levels of fishing effort on the catch and catch rate, followed by an assessment of the economic effects of revenue, number of vessels, costs and profits for those respective effort levels (Reid et al., 2013). The aim of the MEY analysis was to estimate the level of fishing effort that would maximise profitability over a six year period (2008/09 – 2013/14). Results indicated that effort reductions to achieve MEY were in the order of 50 – 70% of the 2007/08 level (Reid et al., 2013). While there was little support for the transition to MEY at the time (Caputi et al., 2015), an unexpected environmentally-driven reduction in puerulus settlement forced managers to act and reduce fishing effort by 44% and 73% in 2008/09 and 2009/10 respectively. This fortuitously was in the range required to achieve MEY and led to a significant improvement in the profitability of the fishery (despite lower catches) through reduced operating costs driven by catch rate increases. A rise in average price of lobster due to declines in supply further improved profitability of the fishery (Reid et al., 2013, Penn et al., 2015).

The reductions in fishing effort made industry cognisant of the benefits of MEY as a proposed target harvest rate, which they formally endorsed using in TACC setting from 2010/11 when the fishery transitioned to ITQs (Caputi et al., 2015) Given the stock assessment used reliable predictions of recruitment to the fishery over the following three to four years based on puerulus settlement, it allowed industry and managers to have increased confidence in projected catch, effort, CPUE and egg production at different harvest levels relative to those stock assessments using the preceding year's CPUE or catch-effort relationship (Caputi et al., 2015, Reid et al., 2013). According to Caputi et al. (2015), the MEY assessment indicated that an additional profit of $30 - $40 million could be made annually through fishing at the higher end of the MEY range (around 5,780 – 7,370 tonnes) compared to lower levels, equating to around 40% of the legal biomass. In 2014/15 the TACC was set at 6,000 tonnes falling within this optimal catch range.

The WARL fishery example highlights again how bio-economics has been used (in this case driven initially by a significant reduction in recruitment to the fishery) to support management decisions that have significantly improved economic value, while concurrently enabling the biological sustainability of the fishery to be maintained. The modelling work has also affirmed for industry the benefits of incorporating economic analyses into decision-making.

The Northern Prawn fishery – a dynamic MEY target

The Commonwealth northern prawn fishery (NPF) has also incorporated economics into decision-making (consistent with the 2007 Commonwealth Harvest Strategy Policy (DAFF, 2007)), through the use of a dynamic MEY strategy since 2008 for two tiger prawn species (Smith et al., 2013). Australia's most valuable commonwealth fishery, the NPF targets several species of prawns including banana prawns (*Penaeus merguiensis*), grooved tiger prawns (*Penaeus semisulcatus*) and brown tiger prawns (*Penaeus esculentus*) (Punt et al., 2010, Deng et al., 2015). The NPF is managed through controls on fishing effort including, limitations on the number of vessels, season length and amount of tradeable gear units (effectively individual transferable effort [ITE] units). Historically the fishery had produced profits of around $20 million annually but by the middle of the 2000s, returns were negative due to increased fuel costs. By 2007/08 however, they had rebounded to around $8 million due to increases in total revenue and the removal of inefficient vessels through a government buy-out (Kompas et al., 2010, Wang and Wang, 2012).

A multispecies weekly size and sex structured population model integrated with an economics model has been developed for the NPF and is used to set the total allowable effort (TAE) (Punt et al., 2010). According to Deng et al. (2015) the population dynamics model is unique in that it attempts to make biological and economic forward projections of the three target species under different levels of fishing effort for two fleets (those targeting *Penaeus esculentus* and *Penaeus semisulcatus*) for the next seven years, Based on these projections the sets of effort levels are selected that maximise the differences between total revenue and costs. The TAE for both species of tiger prawn are then combined, despite being assessed separately. Furthermore, due to the sensitivity surrounding assumptions about the cost structure of industry and prices of prawns the MEY estimate is recalculated every few years (Pascoe et al., 2013a, Smith et al., 2013).

The use of a bio-economic model has provided guidance to managers on the trajectory to MEY. For example, both a 33% increase in total gear was approved by AFMA in 2008 and a buyout initiated between 2006 and 2009 to achieve MEY objectives (Wang and Wang, 2012). According to Pascoe et al. (2013a) the buyout increased profits of the fishery by $22 - $25 million between 2006 and 2009.

By targeting MEY the NPF has adopted a more conservative approach to management as the MEY for both *Penaeus esculentus* and *Penaeus semisulcatus* occurs at stock sizes that are larger than the previous fishery target of MSY, reducing the costs of fishing, lowering impacts on the broader ecosystem and increasing profits (Punt et al., 2010, Kompas et al., 2010).

Lakes and Coorong pipi fishery – managing supply

The previous three fishery case studies have all used bio-economic models to determine MEY and a trajectory for reaching this target reference point. But these fisheries are fortunate in that they have available a large dataset of biological and economic information to employ and are profitable enough to continually revise and re-evaluate the MEY target as the industry cost structure and market price changes through time. The next two fisheries highlight how economics can still be incorporated into fisheries management frameworks in the absence of quantitative data on economic and biological parameters.

The harvest strategy framework for the South Australian Lakes and Coorong pipi (*Donax deltoides*) fishery incorporates economic indicators through the use of a fishery gross margin (FGM) model rather than a complex bio-economic model. A single-species fishery operating along the ocean beach of Younghusband Peninsula in South Australia, this small-scale fishery of 25 quota holders and GVP of $3.6 million supplies a growing human consumption market. Originally managed through input controls, sharply declining CPUE in the mid-2000s precipitated the institution of an ITQ system in 2007/08 and the development of a harvest strategy framework (Ferguson, 2013). Biological indicators include: mean annual relative biomass and presence/absence of pre-recruits, which are assessed through multi-annual fishery-independent surveys. These surveys suggested that the stock was rebuilding from 2009/10, probably due to conservative TACCs set around 300 – 400 tonnes (Ferguson, 2013). The improvement in the state of the stock led to increased consideration of economics in decision-making and the development of the FGM model with which to evaluate the impact of TACC increases on market price.

The FGM model is simplified relative to a bio-economic model as it does not take account of time and the fixed costs of fishing (e.g. business overheads) or assume a link between variable costs of fishing and relative biomass (i.e. marginal costs of fishing constant over all TACC levels) (Morison, 2012, Morison et al., 2014). This means the biological and economic data are less costly and time-consuming to collect (Morison, 2012). Key economic inputs are estimates of expected prices and market shares (at different TACC levels) as well as elasticity of demand and variable costs. These are all sourced through targeted industry surveys. The baseline FGM model then uses this data to estimate how the GVP for the fishery will vary as the proportion of catch assigned to human consumption and to be used as bait changes along with market price (Morison, 2012). The results provide a simple to understand proxy for MEY that improves understanding of the economic drivers of the fishery among industry (Morison et al., 2014).

The FGM model provides a clear example of how economic data can be collected at low-cost to inform the TACC setting process and improve profitability through managing supply. In 2013/14 for example the TACC for pipi was set at 400 tonnes rather than 450 tonnes (which was still biologically sustainable), due to the FGM model revealing that higher profitability could be achieved through reduced supply (Morison et al., 2014).

Shark Bay prawn trawl fishery – cooperative management to MEY

The Shark Bay prawn trawl fishery is an example of how collective action among fishers and empirical economic analyses can lead to improved economic outcomes in the absence of complex bio-economic models. Western Australia's largest and most valuable prawn fishery, the Shark Bay prawn trawl has a GVP of $25 million, an annual catch of 1,800 tonnes and is managed through input controls. Seasonal closures and a limit on the number of fishing days are key management tools used to ensure biological sustainability in the fishery (Kangas et al., 2008). As the price of prawns has decreased through time and costs of fishing have increased, industry have taken an active role in working within the existing management framework to optimise economic efficiency. For example, they have been responsible for opening and closing the fishery to shift effort away from times of reduced catch rate and increase the size, quality and thus market value of prawns (Kangas et al., 2008). In 2015, industry chose to delay the start of the fishing season by a month, as part of a plan to let stocks increase and to fish the stock at its maximal value (Bell, 2015).

The increasing profitability of the Western Australia rock lobster fishery through targeting MEY also increased the receptiveness of the prawn industry to considering the use of economics in management decision-making (De Lestang et al., 2015). Consequently, an empirical MEY analysis is now undertaken, which although it doesn't specify a target, provides an indication of the state of the fishery in relation to the optimal allocation of effort. This has been used to increase profitability in the fishery through, for example, the extension of existing moon closures to shorten the intervening fishing periods and prevent industry from losing money either side of the closure, while concurrently increasing the size (and market value) of prawns harvested later (De Lestang et al., 2015). This had the impact of reducing fishing effort and moving the fishery slightly towards MEY. The analysis indicates that around 140 fishing days is optimal and the fishery is currently sitting around 170.

Similar to the South Australian Lakes and Coorong pipi example, the empirical MEY analysis does not take into account time and what impact the reductions in fishing effort will have on future recruitment, the size of prawns and amount of spawning stock into the future, as this information is not readily available. However in the absence of this complex biological information, economics has still been incorporated into decision-making to improve economic profitability and reaffirm the usefulness of these types of analyses to industry.

Challenges to the application of economics within fisheries management frameworks

Trade-offs among objectives

Some of the greatest limitations to the increased use and adoption of economic instruments, decision rules and target or limit reference points in fisheries management is concern about the short-term transition costs and associated trade-offs between ecological, economic and social objectives (Grafton et al., 2012, Kompas et al., 2010). The adoption of MEY as a target reference point is, for instance, usually associated with substantial cuts in harvest (that are unequally distributed across individuals), reductions in the number of fishing vessels, crew and a decline regional economic activity (Kompas et al., 2010, Reid et al., 2013, Dichmont et al., 2010). At the Commonwealth level for example, the transition to an MEY target under the HSP has already seen major changes within coastal fishing communities five years on (Smith et al., 2013).

The benefits of targeting MEY in the long-term have been the subject of intense debate within the literature (Bromley, 2009, Norman-López and Pascoe, 2011, Christensen, 2010, Grafton et al., 2012, Wang and Wang, 2012). Various authors ascribe that when the whole value of chain of fishing is taken into account, (i.e. not just the fishing sector but processing and retail etc.) there is a net economic loss to society incurred through lost catch, reduced employment in all sectors and expenditure in fishing communities (Wang and Wang, 2012), such that MSY would be a more appropriate target (Christensen, 2010), particularly if other employment opportunities are limited (Béné et al., 2010). The latter point concerning employment opportunities is particularly relevant to many coastal communities in Australia. A related issue is the equity concerns regarding the distribution of economic rent to those remaining fishers through the targeting of MEY (Bromley, 2009). A case in point is the South Australian Lakes and Coorong pipi fishery where the setting of a reduced TACC in 2013/14 than what was biologically sustainable, while optimal for fishers because economic rent is maximized, was sub-optimal for processors and consumers because supply was reduced and in the latter's case, price multiplied. Alternatively, Norman-López and Pascoe (2011) empirically showed that increased profits to the fishing sector from targeting MEY would offset the losses from fleet reductions and lower economic activity, provided the catches were no less than the pre-adjustment level. Furthermore, the effect on local communities was likely to be beneficial, provided the additional income gained by the fishing sector was spent in those communities. Similarly, in an examination of the Western and Central Pacific Ocean tuna fisheries Grafton et al. (2012) calculated that the long-term economic gain per job lost from pursuing MEY was higher than the per capita income of some Pacific Island countries. Both Norman-López and Pascoe (2011) and Grafton et al. (2012) also suggest that the additional inputs and revenue generated in the processing and retail fishing sectors at a non-MEY level are an artifact of inherent market failure in fisheries and could be better employed elsewhere in the economy.

The implementation of economic instruments such as ITQs, have also been the subject of intense debate surrounding trade-offs. While ITQs have shown to improve the economic efficiency of the fishery (Grafton and McIlgorm, 2009, Grafton et al., 2000) there are various social issues such as *inter alia*, reductions in employment, concentration of capital and market power and loss of cultural and community values (Cochrane, 2000, Olson, 2011, Copes and Charles, 2004) that have reduced their desirability. Concerns over some of these have led to many fisheries such as the TSRL fishery compromising the extent of economic benefits by introducing controls on concentration of ownership for example in an effort to sustain local fishing communities (Gardner et al., 2013).

Ultimately the trade-offs associated with targeting MEY or implementing ITQs will vary across fisheries and local communities differently in the short and long term. The decision to target MEY and/or implement ITQs in a particular fishery should ideally be driven by government policy and reflected though clearly articulated management objectives, contained in legislation. These objectives need to take into account all available information, show how trade-offs have been made and be sufficiently operational as to be able to be given effect in the selection of management settings.

Costs of economic data collection and analysis

Assessing trade-offs in order to determine the suitability of an MEY target or even incorporate economics into decision-making can be even more difficult when fisheries are multi-species, small-scale or low value. These features increase the logistical and financial difficulties associated with economic data collection and analysis.

For multi-species fisheries, assessing the trade-offs associated with an MEY approach can be convoluted and require detailed bio-economic analysis. Often consideration is only given to two or three species at best and there are questions over the capacity and resources required to appropriately apply bio-economic models to all targeted species (Smith et al., 2013). Furthermore, even if the logistical, technical and financial capacity is available to determine a MEY for multi-species fisheries, implementation may not proceed due to the reality of competing objectives. For example, in an examination of the trade-offs between ecological and economic objectives in the groundfish bottom-trawl fishery in the U.S., which harvests 34 stocks, Hilborn et al. (2012) determined that profit was maximized at a 2.8% harvest rate but the current harvest rate was set at 1% to ensure that no stocks fell below the biomass that produced MSY. This however, led to 55% reduction in potential yield and 48% reduction in potential profit. Therefore, the trade-offs between ecological and socio-economic objectives in multi-species fisheries can be quite severe when decision-makers target a high conservation status for all stocks (Cheung and Sumaila, 2008).

A lack of investment in the collection and analysis of economic data is a common issue for decision-makers (Hilborn et al., 2005), which Hanna (2011) perceives as a legacy problem of fisheries management being seen primarily as a biological exercise. Bio-economic models require extensive data sets of disaggregated catch-at-age data and associated biological information on the stock, along with data on the economic characteristics of fishery (e.g. variable and fixed costs) and how different levels of vessel participation and fishing effort change the costs structure, market price and therefore profit of the fishery (Pascoe et al., 2014c, Pope, 1997). Bio-economic models also need to be constantly updated and the MEY target revised as the market price and cost structure of fishing changes through time, increasing maintenance costs (Gardner et al., 2015a, Punt et al., 2010, Dichmont et al., 2010). This can make it difficult for many decision-makers to assess the economic viability and profitability of their managed fisheries. In Australia, many jurisdictions have overarching economic objectives implicit or explicit in management plans but a lack of financial and logistical capacity to collect and analyse economic data prevents assessment against these objectives (Begg et al., 2014). For example, despite having a legislative objective to "maximise net economic returns to the Australian community" and a HSP with B_{MEY} as a target reference point, AFMA does not routinely collect economic data for all managed fisheries (AFMA, 2012). Presently only GVP estimates and basic economic survey data is collected for some fisheries by the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES), often one to two years in arrears, with uncertainty surrounding its representativeness (AFMA, 2012). In a review of Commonwealth fisheries legislation in 2012, it was highlighted that AFMA required increased resources to collect and analyse economic data to meet its objectives (Borthwick, 2012).

Data rich fisheries have the option of setting multiyear TACs, which reduce the need for, and costs of collecting economic data and/or running bio-economic models on an annual basis. Costs associated with the use of bio-economic models in low value or small-scale fisheries has led to an increasing reliance on using empirical analyses, biological/economic trigger limits (that initiate further monitoring and assessment) or FGM models (Smith et al., 2013). Research is also continuing on novel ways of incorporating economics into decision-making in small-scale fisheries. For example, Pascoe et al. (2014b) have indicated that proxy values for economic target reference points, such as MEY can be developed for small-scale fisheries through estimating the cost: revenue ratio and where unavailable; can be inferred from limited fishery information such as average price, boat size and main fishing methods.

Lack of consensus and awareness among industry

Fishers are often highly traditional, independent and suspicious of any change, which is generally viewed as a threat to established processes and their future financial viability (Eayrs et al., 2014). They also often believe that concerns over efficiency and optimum levels of fishing effort are the business of industry not research or governmental departments (Cartwright, 2013). This often leads to a desire to maintain the status quo and reluctance to provide economic data to government and/or transition to economic targets such as MEY, particularly when industry associates decreased catches with lower (rather than higher) profits (Dichmont et al., 2010, Cartwright, 2013, Eayrs et al., 2014, AFMA, 2007). For example, in the WARL fishery, initial results of an MEY analysis conducted in 2008 were that fishing effort needed to be reduced by 50% but there was little support for this adjustment at the time due to the associated transition costs (Caputi et al., 2015). Being forced to go through the process of transition a few years later however, to address biological concerns, led to a realisation among industry of the economic benefits and general acceptance of MEY as a target for TACC setting into the future (Caputi et al., 2015). The economic benefits evident in the WARL fishery have also been acknowledged in other Western Australia fisheries, such as the Shark Bay prawn trawl, initiating consideration of economics in decision-making (De Lestang et al., 2015).

Given the paucity of fisheries in the past that could be looked on as examples, fishers can be forgiven for their lack of awareness and understanding of the benefits of incorporating economics into decision-making (Dichmont et al., 2010, Begg et al., 2014). The WARL fishery along with others described in this review however, provide examples that can assist leaders in other fisheries and government to drive evolutionary change through informed dialogue, learning and understanding of the benefits of economics to fisheries decision-making.

The difficulties associated with changing the perception of industry are magnified in large, heterogeneous fisheries where there are distinct groups of fishers (e.g. quota owners and lease fishers in ITQ fisheries) who have diverse business structures, incentives, motivations and to whom management changes can have unequal impacts. For example, Thébaud et al. (2014) illustrated how the effect of unfavourable catch rates impact heterogeneous fishers differently under ITQ management. Lease fishers, while affected by the decreased profitability of fishing operations, will benefit from reduced lease prices, while quota owners will be subjected to both reduced profitability of harvest and loss of income from reduced lease prices. These subtle differences can lead to divergences in opinion during decision-making and given most industry resolutions are made through consensus, or at least rely on majority decision, the high transactions costs of decision-making can lead to the slowing or preventing of management changes (Cartwright, 2013, Johnson and Libecap, 1982, Libecap, 2008). For example, there has been no agreement among industry through the Tasmanian Rock Lobster Fishery Association (TRLFA) on the implementation of more effective spatial management in the form of regional size limits, despite conclusive bio-economic evidence of the advantages for the fishery (Gardner et al., 2014, Cartwright, 2013). This is in part due to the heterogeneous nature of the TRLFA membership and equal voting structure that historically made it difficult to reach a consensus among individuals (Cartwright, 2013). In this instance strong leadership is needed for driving management changes and reducing transaction costs through effective communication of the vision and championing of the cause to others (Eayrs et al., 2014, Gilmour et al., 2013).

Lack of education and training among fisheries managers

There is also a need to increase the education and awareness of fisheries managers who often have limited working knowledge of fisheries economics methods and concepts, particularly in Australia (AFMA, 2007, Begg et al., 2014). Most managers come from biological science backgrounds (Ward and Kelly, 2009, Barber and Taylor, 1990) and while they are expected to be familiar with economic theory have no formal training except what is learned on the job (Branagh, 2015, Barber and Taylor, 1990). This can also lead to an absence of drive and support from government officials to pursue management changes, particularly in an absence of industry consensus, which is why in recent years there has been an impetus to increase economic education and capacity building of Australian fisheries managers through targeted workshops (Branagh, 2015).

Conclusion

There is often a large gap between the current and optimal economic performance of wild-capture commercial fisheries (Cartwright, 2013, Gardner et al., 2013). Economic analyses and instruments have the potential to assist fisheries to bridge this gap but there are a number of challenges associated with their implementation, particularly in Australia. These challenges initially arise from an absence of identified and prioritised objectives within overarching legislation and management plans (Pope, 1997, Cochrane, 2000). Objectives need to be identified and prioritised through stakeholder engagement as there is often a considerable difference of opinion as to the relative importance of ecological, economic and social objectives among stakeholders in a fishery (Pascoe et al., 2013b, Pascoe et al., 2009), particularly as the public has become more engaged in debate (Tracey et al., 2013). Through stakeholders engaging and reviewing each other's proposals from their perspective, trade-offs can be elucidated and evaluated to assist in a consensus being reached (Pascoe et al., 2013b). By reviewing the trade-offs initially and prioritising particular objectives, decision-making becomes proactive rather than reactive (Barber and Taylor, 1990) and measurable targets can be set, which guide the decision-making process and reduce future conflict (Gardner et al., 2015a)

Bio-economic models and other economic analyses (e.g. experimental economics) can assist stakeholder's understanding of the associated trade-offs in objectives by modelling the impact of management decisions or external forces (e.g. climate change) on the stock, ecosystem and various stakeholder groups and comparing the costs and benefits (Pascoe, 2006, Gardner et al., 2013). For example, Péreau et al. (2012) used a dynamic bio-economic model to show the feasibility conditions under which ITQs could achieve ecological, economic and social objectives through time. Concurrently, these types of analyses can also increase the economic literacy of managers and educate industry as to the fleet-wide (rather than individual) benefits of incorporating economics in decision-making. With industry supportive of innovation, government will also be more likely to act and implement management changes (Cartwright, 2013). Bio-economics can also assist in developing measurable targets and reference points, which can be used to assess the current status and progression towards intended outcomes as in the WARL and TSRL fisheries (Gardner et al., 2015a, Caputi et al., 2015). Where commercial fisheries are low value or small-scale and do not have the financial or technical capacity to implement bio-economic models, there is an increasing range of innovative ways of collecting economic data at low cost as well as generating proxy values for economic target reference points with limited fishery information (Pascoe et al., 2014b, Morison et al., 2014).

Over time it is likely that more wild-capture commercial fisheries will actively seek to operationalise objectives and use economic analyses within decision making as the benefits are observed in other fisheries and there are reductions in profitability caused by fluctuating exchange rates, increased aquaculture production and fuel prices. Furthermore, as conflict for resource space intensifies, commercial fisheries will be under increased scrutiny to justify their efficient use of the space and economic significance relative to other competing industries, such as tourism and recreational fisheries (Pascoe, 2006). This will result in the narrowing of the gap between the current and optimal economic performance of wild-capture commercial fisheries.

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Journal Article 2:

The role of government in pursuit of economic objectives in fisheries management

Abstract

Fisheries management is often characterised by multiple, conflicting objectives. Historically, managers have focused on attaining biological objectives but recently there has been an increasing appreciation of the importance of economic and social objectives. Individual transferable quotas (ITQs) are an economic instrument increasingly used in fisheries to rebuild stocks but also improve economic efficiency. ITQs, however, will never maximise economic efficiency while externalities such as assignment problems remain unresolved. Assignment problems are caused by spatial and temporal heterogeneity in the productivity of the stock, which results in fishers overexploiting the stock and congesting fishing grounds when or where the quota unit value is highest, causing economic rent dissipation. Self-governing fishing cooperatives could resolve assignment problems through collective action, however this paper will maintain that their capacity may be diminished under ITQs due to: (i) a breakdown of social capital caused by the privatisation of fishing rights; and (ii) increasing heterogeneity in business structures, incentives and motivations among fishers due to free transferability of quota units. These changes can create significant transactions costs associated with decision-making and remove incentives for fishers to negotiate among themselves and with government to implement management changes. Furthermore, divergent views among fishers and a lack of motivation to pursue collective action and self-governance can lead to reluctance among government to create enabling legislation to authorise and legitimise fishers with the right to organise and self-govern. In this case, governments must be proactive in implementing measures that achieve overarching economic objectives and have the willpower to see these changes through to fruition.

Key words: *fisheries management, objectives, assignment problems, co-management, self-governance, government, industry, ITQ, transaction costs*

Introduction

There is a need to set explicit, measurable and attainable objectives that support overarching goals to ensure effective fisheries management (Barber and Taylor, 1990, Hilborn, 2007, Cochrane, 2000). Objectives should identify what managers aim to achieve, as well as guide their decision-making process, ascertain conflicts, ensure accountability, enable the coordination and effective use of operational capacity and provide a means of reviewing and evaluating fishery performance (Barber and Taylor, 1990). Historically, fisheries management objectives have been broad and lacked specificity, frequently countervailing and with no accompanying strategy or time frame for their attainment or method of prioritisation (Cochrane, 2000, Pope, 1997). Consequently, interpretation of objectives is open to influence and decision-making becomes highly politicised and reactive, with a resulting failure to appropriately consider or understand outcomes for the fishery (Cochrane, 2000). Lackey (1998) considered that many of the historical and contemporary failures of fisheries management relate to the inability of managers to take a holistic approach to consultation and consider the needs and desires of all stakeholders when initially developing management objectives. This has led to disagreement and conflict over decisions due to the varying importance placed on biological, economic, social and political objectives by various stakeholder groups (Pascoe et al., 2013b, Pascoe et al., 2009) and the inability to simultaneously optimise them (Pope, 1997).

While fisheries management is characterised by multiple objectives (Crutchfield, 1973), managers have historically focused on attaining biological objectives in relation to maximising fisheries production (Pascoe et al., 2014a, Barber and Taylor, 1990). This is probably in part a derivative of their codification in Article 61 of the Law of the Sea Convention (LOSC) in 1982, which specified the aim to "maintain or restore populations of harvested species at levels which can produce the maximum sustainable yield" (MSY) and Article 7.2 of the Food and Agricultural Organisation (FAO) Code of Conduct for Responsible Fisheries, which states that measures should be adopted that are "capable of producing maximum sustainable yield." While there is growing recognition of the importance of economic and social objectives and usefulness of economic and social tools in *inter alia* assessing fishery performance and allocating resources (Pascoe, 2006, Van Iseghem et al., 2011), managers continue to frame issues and responses as biological, rather than economic or social (Hanna, 2011). Barber and Taylor (1990) stipulate that this could be due to manager's inherent values, which are a reflection of their educational background being predominately in biological or natural sciences (Ward and Kelly, 2009).

In Australia, jurisdictional governments responsible for commercial fisheries have adopted the principles of ecological sustainable development (ESD) which is clearly reflected in Commonwealth's *Environment Protection and Biodiversity Conservation Act* (EPBC Act 1999) (Pascoe et al., 2014a). In doing so there has been recognition of the importance of economic and social objectives in a holistic approach to fisheries management and these are reflected in high level objectives across jurisdictions. For the most part however, economic and social objectives are implicit and unclear as to their level of importance in accompanying fisheries management plans and policies, prohibiting their appropriate implementation and measurement (Begg et al., 2014, Pascoe et al., 2014a). This has led to decision-making by managers that is primarily focused on attaining biological objectives, even to the extent of justifying the institution of individual transferable quotas (ITQs), which is an economic instrument, on the basis of stock rebuilding (Gardner et al., 2015a). For example, Gardner et al. (2015a) highlight that ITQs were introduced in the Tasmanian southern rock lobster fishery in 1998 due to the need to "constrain the catch to a sustainable level" (Ginn et al., 1997). While the objectives were later expanded to include the development of market mechanism (Ford, 2001) that promoted industry restructure, managers were only guided by the biological objective to set a total allowable commercial catch (TACC) that provided for a sustainable harvest and minimised criticism from an industry driven by the objective of maximising catch. There was no explicit consideration given to economic objectives in TACC setting that are usually pronounced in theoretical discussions of ITQ design (Grafton et al., 2006, Grafton, 1996). Similarly, in the Commonwealth southern shark fishery, ITQs were introduced in 2001 due to stock assessment findings that indicated the biomass was between 15 – 46% of unfished levels in 1995 and a high probability that current effort would lead to further reductions in population size (Sachse and Richardson, 2005). A report commission by the Australian Fisheries Management Authority (AFMA) concluded ITQs were the most appropriate management tool to reduce school shark catches relative to a competitive TACC (FERM, 1997). Again there was little consideration of the economic benefits to the fishery and in the words of Hoydal (2007), biological advice was "twisted and kneaded by management to underpin essentially economic arguments."

While the institution of ITQ management in Australian fisheries could be seen as a reflection of the need to respond to pressing biological issues, their adoption allows managers to concurrently meet economic objectives through the initial design, TACC setting process, development of reference points such as maximum economic yield (MEY) and harvest control rules. ITQs have been introduced in over 121 different fisheries in 22 countries (Deacon, 2012, Chu, 2009), which is a reflection of the increasing interest in the use of economic instruments and analyses in the management of global fisheries (Pascoe, 2006, Sanchirico, 2003, Péreau et al., 2012). ITQs allocate to an individual or firm the right to harvest a proportion of the TACC for a given fish stock in a particular area over a specified time period. While strictly speaking more a use right than a property right in the sense that owners do not own the resource and cannot choose how much to harvest on aggregate (Criddle and Macinko, 2000, National Research Council, 1999), they have been treated as such and shown to increase the aggregate economic value of fisheries by reducing overcapitalisation, increasing fisher efficiency and expanding market opportunity (Costello et al., 2010, Grafton et al., 2000, Grimm et al., 2012).

Maximising the profitability of wild fisheries consistent with overarching economic objectives will only become more important through time as the cost of fuel continues to increase, aquaculture continues to ramp up price-competitive production and market exchange rates fluctuate. ITQs have been largely successful in resolving appropriation externalities uncovered by both Gordon (1954) and Scott (1955) that dissipate economic rent. However, within-season stock and congestion externalities may remain in temporally and spatially heterogeneous fisheries (Boyce, 1992, Grafton, 1996, Deacon and Costello, 2007). Stock externalities occur because the productivity of one unit of effort correlates to the density of the stock being exploited and congestion externalities occur because the catch per unit effort (CPUE) in a given location is correlated with the total amount of effort applied (Boyce, 1992, Grafton, 1996). As the market price for ITQs will not reflect these externalities, fishers will not internalise these effects during decision-making (Deacon and Costello, 2007). Deacon (2012) asserts that residual externalities under ITQ management can be resolved and the economic profitability of the fleet increased furthermore through self-governing fishing cooperatives. According to Basurto et al. (2013) a fishing cooperative is usually defined as "an autonomous association of persons united voluntarily to meet their common, economic, social and cultural needs and aspirations though a jointly owned and democratically controlled enterprise." Theoretically, fishing cooperatives could reduce economic rent dissipation by coordinating the fishing effort of their members such that the harvest in each area is timed to maximise profit, thereby removing competitive behaviour among individual harvesters that increases the total costs of fishing (Fell, 2009). This would allow governments to meet overarching economic objectives for the fishery without having to re-allocate ITQs spatially or temporally or by using concomitant input controls. There are various historical and contemporary examples of fishing cooperatives that have been successful at devising and enforcing (often through social pressure) self-imposed regulations on members; these include, Japanese and Turkish fisheries, New Zealand Challenger scallop enhancement company as well as the United States (U.S.) Gulf of Maine lobster fishery, Pacific whiting, Pacific halibut and Chignik Lake salmon fisheries (Branch et al., 2006, Cancino et al., 2007, Deacon, 2012, Deacon and Costello, 2007).

This paper takes a novel perspective to the role of government in attaining economic objectives under ITQ management and the potential for fishing cooperatives to prevent economic rent dissipation and maximise economic value through collective action and self-governance. In recent times, governments have prioritised biological over economic objectives, the latter has usually being considered the responsibility of industry (Rayns and Read, 2007). However the capacity for industry to maximise economic profitability under ITQ management through self-governance may be difficult due to: (i) a breakdown of social capital caused by the privatisation of fishing rights; and (ii) increasing heterogeneity in business structures, incentives and motivations among fishers due to transferability of ITQs. These issues can create reluctance among government to devolve management responsibility to industry or fashion enabling legislation to support self-governance. Furthermore, these issues increase the transaction costs associated with negotiation and create powerlessness within industry to change according to their own agenda and maximise the economic value of their ITQ right. This can lead to reluctance within industry to agree to revolutionary change, despite it being in their long-term economic interest. It is maintained that governments must prioritise and clearly articulate their fishery economic objectives and in the absence of effective self-governance or collective agreement among industry take a strong role in pursuing measures that meet them.

The nature of ITQs and residual externalities

Individual transferable quotas define a set of users and allocate them rights to harvest from a shared resource (Townsend, 2010b). ITQs have been termed usufructuary rights, as fishers can decide when and how to use their quota share but do not own the full rights to the resource or have the ability to decide how much of the resource can be harvested in its entirety (Criddle and Macinko, 2000, National Research Council, 1999). While there is a perception among fishers that ITQs are de facto property rights (van Putten et al., 2014, FAO, 2013) this varies among fisheries based on the strength of the property right characteristics.[1] ITQs that provide exclusive access to the resource, are perceived as secure, with no restrictions on transferability and are durable (i.e. permanent) are more likely to be considered private property by their owners and deliver efficient economic outcomes. In reality, however, the strength of these property right characteristics is often modified (i.e. weakened) by decision-makers in order to meet multiple objectives (Sumaila, 2010).

As fishers remain joint users of a common pool resource under ITQ management however, externalities remain. Externalities occur when the costs of a fisher's behaviour are not fully borne by that individual but the entire group (National Research Council, 1999). Despite being allocated an ITQ right, owners have no assurance that others will refrain from fishing practices or behaviour that reduce the economic profitability of the entire fishery (Criddle and Macinko, 2000). Equally, government has no guarantee that economic objectives for the fishery can be attained with simply the institution of ITQ management.

While ITQs promote individual economic rationality they do not support collective economic rationality, with residual within-season stock and congestion externalities in many ITQ fisheries creating assignment problems. Assignment problems dissipate economic rent and impede governments maximising fishery profitability in line with economic objectives. Assignment problems are caused by heterogeneity in the productivity of the stock such that the economic value of a single quota unit varies based on location, time and market preference (Costello and Deacon, 2007, Deacon and Costello, 2007). For example, inshore areas close to ports and market facilities or spawning grounds with high stock densities would have reduced costs of fishing relative to other areas and consequently a higher quota unit value. Fishers will race each other to overexploit productive areas, times or market prized fish sizes first, increasing associated costs of fishing through depleting the stock (or sub-stocks) and congesting fishing grounds (Copes, 1986, Boyce, 1992, Costello and Deacon, 2007, Copes and Charles, 2004). Furthermore, as fishers have no incentive to share information about their fishing effort or the location of productive fishing grounds with others, economic rent can be further dissipated through redundant effort (e.g. searching for productive fishing grounds) (Deacon and Costello, 2007).

There are various examples of fisheries where the aggregate economic rent is below the maximum because ITQs fail to provide effective incentives to distribute fishing effort both spatially and temporally (Wilen et al., 2012). For example, Bisack and Sutinen (2006) showed that in the New Zealand southern scallop fishery, fishers harvested a higher number of scallops early in the season than what was optimal, resulting in profits that were 10 – 20% less than the maximum. Similarly, Holland (2011) modelled ITQ management in the Maine lobster fishery and showed that profits could be reduced by as much as 30% due to unresolved within season stock and congestion externalities.

In order to resolve assignment problems and maximise the economic value of the fishery, either the ITQ allocation needs to be spatially and temporally delineated with the aim of regulating harvests or there needs to be formal rules or informal conventions among fishers to coordinate their fishing effort (Holland, 2011, Costello and Deacon, 2007).

The advantages of fishing cooperatives

Realistically it may not be possible for governments to define an optimal ITQ structure that matches the scale of management to local dynamics of the resource, as the cost of implementation are likely to

[1] For more information on property right characteristics see Ridgeway, L., Schmidt, C.-C. (2010) Economic instruments in OECD fisheries: issues and implementation. In: *Handbook of marine fisheries conservation and management.* (Eds. R.Q. Grafton, R. Hilborn, D. Squires, M. Tait, M. Williams), Oxford University Press, New York ; Oxford, pp. 310-323.

be prohibitive (Townsend, 2010a, Prince, 2005). Furthermore, the perceived value and costs of doing so are likely to change through time as a result of new technology, scientific information and market fluctuations (Holland, 2004, Demsetz, 1967). Perhaps this is why such spatially and temporally delineated ITQ programs are yet to be designed and tested (Yang, 2011) and in a number of Australian fisheries, such as the Tasmanian southern rock lobster fishery (DPIPWE, 2009) and the Commonwealth southern and eastern scalefish and shark fishery (Slope Resource Assessment Group, 2009) managers have refrained from spatially re-allocating ITQs, despite the presence of stock externalities and overarching objectives to maximise economic efficiency.

This inability or unwillingness from centralised government to impose further regulations on fishers or accept the logistical and financial burden of additional monitoring, control and surveillance and scientific research has led authors to highlight the theoretical advantages of self-governing fishing cooperatives in resolving assignment problems and maximising the economic value of quota units (Gilmour et al., 2011, Gilmour et al., 2013, Costello and Deacon, 2007, Deacon, 2012). In fact, Scott (1955) and Scott (1988) initially conjectured that ITQ ownership could form the foundation for collective action through a sole ownership institution, such as a fishing cooperative. Self-governance is about fishers being empowered to make governance decisions (Townsend and Shotton, 2008). In doing so, self-governing fishing cooperatives could overcome the divergence between individual and collective rationality that leads to externalities, such as assignment problems, through collective action and strengthening three of the property right characteristics of the ITQ right (Yang, 2011, Townsend, 1995). First, through operating as a collective, the bargaining power of quota owners would increase and their ITQ right would be more excludable to external challenges. Second, through negotiation and collective decision-making, information sharing and communication would increase, facilitating the transferability of ITQs. Third, through coordinating effort and potentially self-monitoring fishing practices, the security of the ITQ right would increase as fishers would feel more confident of receiving future benefits due to effective management. Increasing the strength of the property right characteristics of the ITQ would also feedback into the effective functioning of the fishing cooperative.

There are a number of historical and contemporary examples of societies that have independently constructed, enforced and maintained rules and norms to manage common pool resources (Feeny et al., 1990, Schlager et al., 1994). More specifically, there are various examples of fishing cooperatives regulating the effort of their members by rotating access to fishing grounds or assigning individual time slots for certain fishing areas in order to avoid assignment problems (Schlager, 1994, Townsend and Shotton, 2008). For example, in Japanese coastal waters, fishery management organisations (FMO), which are a derivative of fishery cooperative associations (FCA), coordinate fishing effort and share income among members in order to reduce within-season stock and congestion externalities (Cancino et al., 2007, Gaspart and Seki, 2003). Effort is rotated between productive and less productive areas to ensure equitable access for all members and reduce congestion externalities. Income is pooled and distributed equally among members to reduce stock externalities. By sharing income, members have a greater incentive to cooperate within the FMO as the benefit from over-appropriating the resource at a productive area or given time are not individually attained (Gaspart and Seki, 2003). Similar assignment rules for managing effort on fishing grounds have been used in some Turkish coastal fisheries with fishing sites rotated to ensure equitability and reduce congestion externalities (Feeny et al., 1990). Likewise, income sharing was agreed among members of the Lake Chignik sockeye salmon fishing cooperative, enabling the consolidation of fishing effort, change in harvesting practices and overall reduction in congestion externalities (Deacon, 2012, Branch et al., 2006). More famously in the Maine lobster fishery in the U.S., territorial harbour gangs functioning as fishery cooperatives have been effective in the past at introducing and enforcing rules to reduce assignment problems through the definition of fishing zones for each gang (Acheson and Gardner, 2014, Acheson and Gardner, 2010).

The coordination of fishing effort is not the only improvement to economic performance that could be attained through self-governing fishing cooperatives. Cancino et al. (2007) and Townsend (2010b) highlight several tasks that a cooperative could perform more efficiently than government including: self-monitoring, enforcement and sanctions, multi-species management and habitat enhancement, routine administration and research, market coordination and incorporation of economic information into decision-making and the development of ecosystem services. The Challenger Scallop Enhancement Company (CSEC) in New Zealand is a model example of a cooperative, which not only coordinates the effort of members but invests in stock enhancement through reseeding habitat and organises independent research on stock abundance (Deacon, 2012). In an empirical evaluation of the success of self-governance in the CSEC and the Bluff Oyster fishery in New Zealand, Yang et al. (2014) identified that self-governance can contribute to economic efficiency, institutional effectiveness and to a lesser extent resource and environmental preservation.

Requirements for successful collective action

While the advantages of self-governing fishing cooperatives may be clear, there is a large number of variables which have been identified as affecting the likelihood of resource users' engaging in collective action and their level of success (Ostrom, 1990, Ostrom, 2009). While there is no consensus on which of these conditions or combination thereof will result in effective collective action (Taylor, 1990), there are some key factors (see, Gutiérrez et al., 2011) that feature prominently in a review of the literature, including: users' perceived need and benefit for change, level of government legitimisation, strong leadership and existing social capital.

First, the emergence of collective action can often occur due to a perceived need for change associated with a major event or shock to the existing system (Acheson and Gardner, 2014). This is aligned with assertions made by Libecap (2008), who speculated collective action problems in resource management develop when resource degradation increases to the point that people perceive a need for change and concomitantly there is enough political support for politicians to pass laws. In a study of Australian abalone fisheries, Gilmour et al. (2011) highlighted that subjective perceptions of resource conditions were key determinants in industry-driven resource management. Inherently linked with the perceived need for collective-action, is the requirement that individuals who invest time and effort to self-organise clearly benefit from the development of rules, otherwise there will be a lack of support in generating them (Knight, 1992, Ostrom and Schlager, 1996). In a study of self-management among five Australian abalone fisheries Gilmour et al. (2013) identified industry perceptions of the benefits of collective action as critical to participation.

Second, the ability of resource users' to engage in collective action through creating, establishing, enforcing and maintaining local rules and regulations is influenced by the level of government legitimacy and authority given to the local institution and it's arrangements within legislation (Pomeroy, 2001, Deacon and Ovando, 2013, Pomeroy and Berkes, 1997). For example, government legitimisation of local rules and regulations is critical in enabling fishers to deter and sanction non-compliant individuals, an area where historically they have had poor capacity and limited success (Gilmour et al., 2013) but which is critical to the long-term maintenance of self-governance (Ostrom, 1990, Nowak, 2006). If governments don't at least minimally recognise the legitimacy of resource users' rules and regulations then it will become difficult for them to maintain them into the future (Ostrom, 1994, Pomeroy, 2001, Pomeroy and Berkes, 1997). For example, in an empirical comparison of self-governance within two New Zealand fisheries, Yang et al. (2014) revealed how the full benefits of self-governance cannot be realised without government support in the form of legally authorising (and thereby facilitating) local institutions to self-govern.

Third, local leadership has also been identified as critical for ensuring the success of collective action (Pomeroy et al., 2001). Leaders lower the transaction costs associated with collective action through providing direction, facilitating discussion, resolving conflict, influencing opinions and bringing enthusiasm and energy to the process (Pomeroy et al., 2001). Furthermore, leaders can play a visionary role by framing the benefits of collective action in such a way that appeals to the underlying social norms influencing an individual's decision to cooperate (Gilmour et al., 2013). In a study of

130 co-managed fisheries Gutiérrez et al. (2011) was able to empirically identify leadership as the most important contributor to success. Pomeroy (2001) also highlighted that a lack of leadership was a major limiting factor in the ability of fishers in the Philippines to progress from recognising a problem and discussing potential solutions to actually taking action to formally organise and develop institutional arrangements to mitigate the problem.

Fourth, communities of resource users that have a long history of connectedness through kinship or friendship, relations of trust and reciprocity as well as common rules or norms (broadly termed social capital) have been shown to be more conducive to collective action (Ostrom, 1990, Gutiérrez et al., 2011, Pretty, 2003). This is because strong social capital within communities lowers the transaction costs associated with negotiation, decision-making, implementation and monitoring of rules and regulations (Jentoft et al., 1998, Singleton and Taylor, 1992, Pretty, 2003, Ostrom, 2009). For example, in a study of Australian abalone fisheries Gilmour et al. (2011) showed that higher levels of trust and cooperative capacity led to greater levels of industry driven resource management, while lower levels led to greater reliance on external agents (i.e. government) to generate cooperative management outcomes. The level of social capital may also be linked to the size and heterogeneity of the resource users (Gilmour et al., 2013, Ostrom, 2009). Experimental and some empirical evidence suggests that small and homogenous groups are more likely to cooperate and agree upon self-governing rules (Acheson and Gardner, 2010, Hackett et al., 1994, Johnson and Libecap, 1982, Kanbur, 1992). For example, Acheson and Gardner (2014) highlighted that harbour gangs in the Maine lobster fishery were able to develop and enforce self-governing rules as: (i) fishing practices were homogenous, ensuring rules impacted everyone equally and; (ii) gangs (i.e. coastal communities) were relatively small in size, ensuring rules could be appropriately monitored.

The presence/absence of these key requirements of collective action and self-governance will be influenced by the overarching management system in each fishery. For example, a fisher’s perceived benefit from engaging in collective action may be higher under ITQs than other forms of management, particularly if the exclusivity or security of the ITQ right is perceived to be strong. Concurrently, ITQs may make it more difficult for industry to engage in collective action to resolve residual externalities and maximise the economic value of their ITQ right. This is due to the breakdown of social capital caused by privatisation and increasing heterogeneity among resource users caused by transferability of quota units. Both of these may have concomitant impacts on government and industry decision-making, with the former reluctant to devolve management responsibility and create enabling legislation to support self-governance and the latter beset by difficulties in reaching unanimous agreement on management changes that increase collective economic profitability.

Potential difficulties with self-governance under ITQs

Breakdown of social capital and increasing fisher heterogeneity

Various empirical and experimental studies have shown that collective action and the potential for self-governance among resource users is more likely to be successful when there is a history of interconnectedness, familiarity, trust and reciprocity as well as local rules or norms (i.e. social capital) (Gutiérrez et al., 2011, Ostrom, 1990, Ostrom et al., 1994, Jentoft et al., 1998). Concurrently however, it has been shown that the institution of market-based institutions may “crowd out” these existing self-regarding preferences or trigger a “market instinct” resulting in resource users behaving in a self-interested profit seeking manner, which could inhibit their ability to cooperate (Reeson and Tisdell, 2010, Cárdenas et al., 2000, Bowles, 1998). More specifically, the institution of ITQs may undermine informal cooperation or implicit contracts based on trust and reciprocity through: (i) the highly mobile nature of quota markets, which may lead to shifting ownership and weaken the credibility of long-term contracts and; (ii) the allocation of quota units, which may change the relative bargaining power of resource users (Seabright, 1993). In other words, social relations and significations are subverted and interactions between fishers become profit-based monetary exchanges (Helgason and Palsson, 1997, Gibbs, 2009). A clear example of this is the facilitation of quota trading following the institution of ITQs, with less fishers trading quota or entitlements directly with each other but preferring to use brokers (van Putten and Gardner, 2010, Pinkerton and Edwards, 2009). In the Tasmanian rock lobster fishery for example, the number of broker facilitated trades rose from 20% in 2010 to an estimated 60% in 2015 (C. Gardner, pers. comm.) with fishers reporting they preferred to use brokers to distance themselves from the trade and avoid feelings of guilt from pursuing higher prices at the expense of others (Bradshaw, 2004b, van Putten and Gardner, 2010, Helgason and Palsson, 1997). This behaviour also aligns with experimental evidence, which suggests that in a competitive market (such as ITQs) with multiple anonymous buyers and sellers, self-interested rather than self-regarding preferences will be observed (Reeson and Tisdell, 2010).

Alternatively, some authors venture that the institution of ITQs could foster collective action and self-governance by aligning the self-interest, collective goals and normative values of the prescribed owners (Yang, 2011, van Putten et al., 2014, Van Slyke, 2007). However, individuals sharing a fishing ground, technology or even a access right (what Jentoft et al. (1998) term "functional groups") are not necessarily equivalent to sharing a history and bonds of kinship and friendship ("territorial groups"), where commitment and continuity are likely to be superior (Jentoft et al., 1998).

Collective action and self-governance among "functional groups" of ITQ fishers has also been made more difficult through time as the free transferability of quota units and increasing globalisation of markets has increased fisher heterogeneity. Various empirical and experimental studies have shown that collective action is increased when groups are homogeneous in terms of social preferences, skill, wealth and underlying business structures (Cárdenas, 2003). In many ITQ fisheries however, there is an increasing separation of ownership and control with a clear division between capital and labour (Bradshaw, 2004b, Pinkerton and Edwards, 2009, Bradshaw, 2004a) as fishers prefer to lease out their quota units to others, rather than sell them and gain passive income from their asset (Connor and Alden, 2001, van Putten and Gardner, 2010). For example, ten years after the implementation of ITQ management in the British Columbia halibut fishery and Tasmanian southern rock lobster fishery, 79% and 37% of the quota respectively was leased out by quota owners rather than directly fished, with the number of lease fishers rising (Pinkerton and Edwards, 2009, van Putten and Gardner, 2010). This separation of ownership and control of wealth was highlighted by Adam Smith when discussing the advent of the modern corporation (Smith, 1776) and later classified as a principal-agent problem (Jensen and Meckling, 1976). Using the example of an ITQ fishery, lease fishers (agents) may have different objectives and utility functions to quota owners (principals). The latter has an incentive to ensure the future stream of benefits from the resource are maximised, while the former is exposed to strong incentives to maximise short-term revenue in order to pay the rising costs of leasing (Munro et al., 2009, Emery et al., 2014b, Davidson, 2010, Eythorsson, 1996). Additionally, lease fishers may be less likely to support measures trying to ensure long-term sustainability of the fishery due to a lack of ownership (van Putten et al., 2014, Parslow, 2010). Given the ITQ right is usufructuary and quota owners cannot perfectly monitor the actions of individuals who lease their quota, this divergence in behaviour will lead to costs being incurred (Davis et al., 1997, Marks, 1999).

Increased transaction costs of collective action

One of the issues with the separation of the ownership and control of wealth and the breakdown of social capital is that the transaction costs associated with negotiation both among industry and with government for collective action and self-governance increase. This is because lease fishers and quota owners (or similarly small and large vessel operators), who vary in terms of wealth and underlying business structures, will be impacted differently by management changes (Emery et al., 2014b). For example, while the presence of a non-binding TACC and declining catch rates will adversely impact quota owners, in terms of reduced fishing profitability and loss of income from lower lease prices, lease fishers will suffer reduced fishing profitability but benefit from lower lease prices (Emery et al., 2014a, Thébaud et al., 2014, Gardner et al., 2013). Efforts to reduce the TACC and improve the economic profitability of the fishery may therefore generate disparate responses among heterogeneous fishers due to these differential impacts (Thébaud et al., 2014). Similarly, studies have shown how the implementation of rights based management or ITQs have been delayed due to the variation in harvesting costs among fishers (due to skill), which reduces their willingness to adopt alternative management measures due to the unequal redistribution of wealth (Karpoff, 1987, Johnson and Libecap, 1982, Grainger and Costello, 2012). This can lead to a continued powerlessness among fishers to reach a coherent decision and maximise their economic profitability through collective action.

Difficulties in reaching a consensus or agreed position on management changes due to heterogeneity could be overcome if collective decision-making rules were based on corporate governance structures where voting rights were proportional to the ownership of ITQ shares (Townsend et al., 2006). In many ITQ fisheries however, fishing cooperatives or industry associations require unanimous consent among fishers, many of whom are not quota holders, for any rule to be self-imposed and government often maintains the right to overturn any decision at is sole discretion (Townsend et al., 2006). Both increase the transaction costs and reduce the willingness of individuals to negotiate due to the uncertainty surrounding outcomes. This may be why many successful examples of industry self-governance are within fisheries with either a small number of participants (usually less than 15) or whose participants have homogenous characteristics (e.g. vessels, fishing gear, wealth) (Townsend and Shotton, 2008, Hilborn et al., 2005). The former being inherently rare in fisheries (but may be increased through ITQ consolidation) and the latter increasingly absent due to the privatisation of fisheries and free transferability of quota units.

Government reluctance and industry inertia

The transaction costs associated with heterogeneity and decision-making rules can reduce both the likelihood that fishers can change according to their own agenda and that management changes proposed by government will be accepted by fishers more broadly. This inertia can make the pursuit of objectives aimed at maximising the economic value of the fishery more problematic, particularly when government is often seeking widespread approval for management changes from fishers prior to implementation (Stone, 2005). An example of this problem was evident in the Tasmanian southern rock lobster fishery, where modelling indicated that spatially-delineated size limits for lobsters could improve the economic profitability of the ITQ-managed fishery. While initially supported by fishers on steering groups, the proposed management changes were consistently opposed by the broader industry at fishery cooperative meetings due to *inter alia* equity concerns about zonal restrictions and economic issues surrounding market prices (C. Gardner, pers. comm. 2015; Gardner et al., 2014). Similarly, in the ITQ-managed Commonwealth gillnet hook and trap fishery (GHAT) in Australia, small-sized vessel owners have actively opposed the authorisation of automatic longline fishing methods, which are designed to improve the economic efficiency of the fleet. This is because they are unable to utilise automatic longline gear on their gillnet vessels and consequently see their individual businesses financially disadvantaged relative to large-sized vessel owners who have the ability to utilise both methods (D. Power, pers. comm. 2015). The opposing perspectives among industry in the GHAT fishery have culminated in the formation of two industry associations to represent their views to managers, further increasing the costs of management (D. Power, pers. comm. 2015).

An overall reluctance within industry to change and the perception of the need to do so could also be an important factor explaining the inability of industry to overcome high transaction costs associated with collective action under ITQ management. Fishers are often highly independent and view change with trepidation suspicion and scepticism. Revolutionary change, which requires fishers significantly alter their business operations, is seen as a particular threat to established processes and financial viability, resulting in resistance (Eayrs et al., 2014). For example, Eayrs et al. (2014) revealed how New England groundfish fishers refrained from adopting an alternative fishing gear, despite knowledge of the benefits and low associated risks to profitability, seemingly based on a desire to maintain the status quo. This aligns with the finding of Gilmour et al. (2011) who in examining

Australian abalone fisheries showed that even if the capacity to cooperate existed among fishers (e.g. due to strong social capital), there was still a preference to maintain the status quo if they *perceived* the marginal benefits of change were less than the associated costs.

The difficulties of collective action caused by transaction costs and general resistance to change may not only reduce the desire and ability of fishers to manage themselves but lead to a reluctance among government to create enabling legislation to devolve responsibility for certain management functions to industry, which is also critical for its success (Townsend, 2010b, Yang et al., 2014, Pomeroy and Berkes, 1997, Pomeroy et al., 2001). For example, in the Tasmanian rock lobster fishery, there was continued resistance among industry to proposed TACC reductions despite falling catch rates and decreases in quota unit value (Gardner, 2012, Gardner et al., 2015a). While fisheries managers advocated the advantages of setting the TACC at MEY in order to maximise collective economic profitability, there was a reluctance among wider industry to accept that economic yield and asset values could increase with lower catches (Gardner et al., 2015a). This led to period of three years where the TACC was non-binding (i.e. not caught) due to inadequate reductions (Emery et al., 2014a). A similar pattern of resistance among industry to TACC reductions was also evident in the South Australian rock lobster fishery at a similar time (Linnane et al., 2010). It is these types of examples that demonstrate the inability of fishers to act as a cohesive group, undermining the confidence and trust of government has in the ability of fishers to unilaterally agree on management decisions.

Co-management has often been introduced by government as a way of increasing collaborative engagement with industry, building trust and facilitating information exchange, while simultaneously allowing an evaluation to be made on the potential for future devolution of management functions to industry institutions. For example, AFMA commenced a co-management trial in three fisheries between 2008 and 2010 through amending its legislation to support delegation of functions to industry institutional structures. While the trials highlighted the capacity for industry institutions within a specific fishery to successfully manage a range of fisheries management functions (e.g. data collection), this was limited by the level of support (representation) among respective fishers (Bolton et al., 2015). In many Australian fisheries there is a distinct lack of industry institutions or peak bodies with close to 100% representation. A lack of representation increases uncertainty about the benefits and stability of co-management and reduces incentives for fishers to volunteer time and money to industry institutions when non-members potentially derive benefits (Bolton et al., 2015). Furthermore, heterogeneity among fishers increases the likelihood of individual interests not being appropriately represented (e.g. as evident in the GHAT fishery). This means government is still required to consult on management changes with individuals rather than the collective, reducing the likelihood of reductions in management costs. The inability of industry institutions to represent the majority of fishers and present a collective viewpoint, again, provides government with further justification for refraining from making legislative or policy changes that enable industry to take on a greater role in management of the fishery through self-governance.

Conclusion

If fishers are unable to collectively agree to management changes or proactive in pursuing self-governance then government will be even less inclined to provide enabling legislation to authorise and legitimise the right to organise and self-govern (Pomeroy and Berkes, 1997). This precipitates a self-perpetuating cycle of increased transaction costs of decision-making and negotiation with government on the part of industry and a general inertia and resistance within to change, which may be difficult to reverse. There is no doubt self-governing cooperatives could resolve assignment problems caused by temporal and spatial heterogeneity in the stock and thereby maximise the value of their quota units more effectively than government. The introduction of ITQ management has in many instances however, made this more difficult to achieve through the breakdown of social capital caused by privatisation and the free transferability of quota units, which has increased heterogeneity among fishers and lead to a separation of ownership and control. For the most part, successful examples of self-governance are often in fisheries where: (i) the number of fishers are small which increases the likelihood of fisher homogeneity; (ii) the target species is sedentary, which offers closed and predictable biological systems for management thereby reducing associated transaction costs (Townsend and Shotton, 2008) or; (iii) the fishery is geographically isolated with high transaction costs of commercialisation, which precipitates collective action (Basurto et al., 2013). Even when self-governance has evolved among large ITQ fisheries it has been under unique circumstances, such as homogeneity among fishers, which is often absent in a globalised fishery (Townsend, 2010b, Townsend and Shotton, 2008),

Governments are now left with two issues. Firstly, how to construct, interpret and implement meaningful overarching economic objectives under ITQ management relative to other (e.g. social) objectives. Secondly, government is faced with the conundrum, of maximising fishery profitability when consensus is absent but often (informally) required (desired) to implement management changes. Where this is the case, government must first clearly outline and articulate to stakeholders the importance of attaining fishery economic objectives. Furthermore, they must be proactive in implementing measures that achieve overarching economic objectives and have the willpower to realise these changes even in the face of considerable opposition. Stone (2005) provides the example of the Commonwealth northern prawn fishery, where lack of will from the management agency to make a decision without unanimous industry support led to inaction on reducing fishing effort and it wasn't until a conscious political decision was made to prohibit lobbying and threaten to proceed with changes without approval that industry chose to negotiate collaboratively with government. Similarly, Yandle (2006) highlights how strong political will in the face of considerable opposition to the implementation of catch restrictions in the New Zealand rock lobster fishery was responsible for precipitating a collective desire within industry to negotiate collaboratively with government on the effective implementation of ITQ management.

Consequently, where the key requirements of collective action are absent, there is an urgent need for government to act swiftly with the courage and will to establish appropriate management frameworks and make decisions even in the face of considerable opposition. At best this may have the advantage of removing political lobbying and delaying tactics and provide an impetus for collaborative management among industry, while at worst it will make the government unpopular in the short-term with sections of the industry. In the long-term however, the government can ensure it attains overarching legislative objectives aimed at maximising fishery profitability.

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Journal Article 3:

Commercial scale invertebrate fisheries enhancement in Australia: experiences, challenges and opportunities

Abstract

Stock enhancement or "assisted recruitment" for fisheries management in Australia is at an experimental R&D phase. Development of the science has focused largely on recreational finfish; however it is considered that high value invertebrates will be the best candidates for commercial scale fisheries enhancement. Three main ingredients are required; technical capacity, governance capability, and the 'correct' species. The technical capacity needed is in the area of hatchery production and wild release methodologies, whilst the governance capability needed is informed policy that accounts for the complexities and interdisciplinary nature of stock enhancement. In particular, the appropriate articulation of policy to support economic development and integration into wild fisheries is currently lacking. If successful stock enhancement is implemented, the nature of fisheries management changes because the recruitment side of the fisheries equation is under substantial control, rather than just the production side. Management responses will require significant innovation, with a renewed emphasis on understanding the stock, rather than policing the fishers. By way of illustration, recent initiatives and key challenges encountered in Australian invertebrate fisheries are investigated through case studies. An example of a commercially-viable enhancement fishery that reflects solutions to the key challenges is also presented. The review ends with an argument to re-establish the context of stock enhancement in the discipline of ecological enhancement. This is a crucial and positive step forward for it recognises that, in principle, any renewable aquatic ecosystem has the potential to be enhanced instead of just depleted.

Introduction

The validity of releasing or moving animals to enhance marine fish stocks as part of a management toolbox is gaining wider acceptance as the science develops. Enhancement science in fisheries or more specifically, the study of "assisted recruitment" is an integrated and interdisciplinary practice that supports an ethically-based set of management principles known as the "Responsible Approach" [1–3]. The principles provide guidance and consilience to a diverse discipline including aquaculture, economic analysis, fisheries stock assessment, population and evolutionary genetics, fish behaviour studies, underwater technology development, disease management, and policy governance frameworks. The diversity arises from the recognition that management intervention in complex natural systems is unlikely to be successful and responsible unless there is a greater understanding of not just the ecological, but also the economic, social and institutional/governance issues [4,5]. Marine enhancement programs do not have a large record of success stories however; the allure of a 'quick fix' is a legacy that still influences a popular perception of the practice [6], namely that stock enhancement is a 'slow and dirty' response to an environmental degradation or perturbation, poor fisheries management, or a failing aquaculture industry.

Successful implementation of stock enhancement is a complex endeavour requiring the knowledge and resources of socio-economic and evolutionary-ecological disciplines and there are few examples where all principles have been covered [7]. Thus a pragmatic approach has been advocated in Australia to date, with an emphasis on developing enhancement science through small-scale experiments, and a focus on recreationally important finfish rather than commercially fished species [7,8]. Consequently there has been limited incentive to extend the reach of enhancement programmes beyond the initial breeding and release phases, which do support commercially viable aquaculture hatcheries. Enhancement of commercial fisheries however, can only occur when successful culture and release methodologies are combined with well understood survival rates and Government policy which supports integration of enhancement into wild fisheries management.

The objectives of this paper are to review the development and challenges of commercial scale invertebrate fisheries enhancement in Australia, the roadblocks to be overcome, and likely successful candidates in the future. In the interests of maintaining the pragmatic position, the concept of a "stock" in a fisheries enhancement framework is first reviewed, to identify key limitations regarding the efficacy of this practice, followed by a summary of relevant studies. The complexities of enhancement are then distilled into two key challenges related to economics and stock identity. To complete the review, an example of a commercially-viable enhancement fishery that reflects solutions to the key challenges is presented, along with an argument in support of re-establishing the context of stock enhancement in the discipline of ecological enhancement. This is a crucial and positive step forward for it recognises that, in principle, any renewable aquatic ecosystem has the potential to be enhanced instead of just depleted

Self-correcting stocks and the limitations of enhancement

At the heart of the stock-enhancement conundrum is a conflict between the expected theoretical behaviour of fish stocks and their observed behaviour. Separate fish stocks are assumed, and best practice fisheries management is able to make assessments, conclusions, and statements about "stock status". A stock is considered, implicitly or explicitly, a self-replicating unit and the objective of controlling fishing mortality (i.e. management) is to support this pre-existing regenerative capacity. Of what relevance therefore, is the concept of "stock enhancement" or "assisted recruitment" when a stock has already evolved its self-sustaining strategy? This paradigm, which confers "Lazurus-like" powers on the compensatory dynamics that connect a stock to its progeny, whilst increasingly challenged in recent times by stock collapses [9], and a general lack of evidence [10], is still a major theoretical roadblock in discussions about the efficacy of enhancement.

To work in practice, the idea of a self-regenerating stock requires first, that there is an evolutionarily and ecologically stable group of organisms, a "stock", which can be uniquely identified; second, that a measurement of stock stability can be made, for example, through genetic and population analysis. Early synthesis of these ideas was presented in the self-regenerating models of stock and recruitment [11,12], which demonstrated mathematically, the plausible causal factors (e.g. density dependent growth and survival in the larval phase) that could underlie the perturbation and recovery trajectories in observed fish stocks in the North Atlantic following cessation of fishing during the first two world wars. The large positive effect from the reduction in fishing effort on stock size confirmed the fundamental importance of controlling fishing. Environmental effects however, were considered equally important, and both were viewed as key components to the self-regenerating model. Quantification of environmental factors has been problematic though, as their effect is often masked or exposed by fishing mortality depending on the extent of erosion of compensatory reserve in exploited stocks [9]. Only these species where pre-settlement processes dominate recruitment exhibit relatively consistent environmental influences [13].

Development of stock-recruitment-environment relationships for both invertebrate and finfish species has highlighted the variability and range of productivity within a single stock ([14–16]. An illustrative example is provided in the self-regeneration model for North Sea cod (*Gadus morhua*) [16]. This study found from an analysis of a 50 year time-series, that the shape and the position of the compensatory dynamic of the stock is not fixed, but varies in response to environmental conditions. Specifically, a high spawning stock (2,000,000 tonnes) could produce an order of magnitude variation in recruitment (range: 50 to 650 million), depending on the availability of zooplankton and sea-temperature. At a low spawning stock (40,000 tonnes) however, recruitment was far less variable (180 – 200 million) and largely independent of environment. Although their model only explained 45% of the variability in the time-series, it was large improvement in the 10% explanation by the more traditional stock-recruitment models [16]. Faced with the stochastic nature of natural recruitment, an explanatory R^2 of 0.45 may be approaching a maximum in many, if not most, species.

From an enhancement perspective, the challenge is to demonstrate how the ‘assisted recruitment’ process will co-exist with a given stocks’ self-correcting mechanisms. Two relevant conclusions from the North Sea cod model [16] are: firstly, enhancement will always result in increased biomass at low spawning stock; secondly, enhancement could result in an equally large increase in biomass at high spawning stock, or it may not have any effect at all. Success depends on the ability to understand and predict the environmental conditions, as well as acceptance of the intrinsic stochastic nature of natural recruitment. In simple terms, it is a calculated gamble that in any given year or location, the stock is experiencing recruitment limitation and enhancement will reap a positive effect on biomass.

Even with its uncertainty, the self-regeneration model for North Sea cod represents an exceptional case of knowledge of a “stock”. Management of most stocks has adopted the pragmatic definition, namely that a ‘unit stock’ be an operational, rather than a biological matter [17]. This pragmatism stems from a dearth of knowledge of the true population parameters of a stock (growth, natural and fishing mortality, reproduction, and recruitment), and a need to manage it with limited resources. To be successful with an assisted recruitment program however, requires a greater knowledge of critical stock parameters, in particular, the multiple pathways responsible for self-maintenance. For example, the theoretical concept of recruitment limitation [18] is a key idea in support for the efficacy of enhancement, but knowledge of the spatial and temporal scale of the ‘bottlenecks’ that cause the limitation is required in order to benefit from it. On the production side of the equation, growth-mediated regulation in the recruited stock may act to reduce the expected biomass resulting from successful enhancement [19], unless it has been specifically accounted for.

The key to successful enhancement is knowing when to “intervene” in any given stocks’ “regenerative cycle” to capitalise on the limitation present at a particular point in the cycle. Given the general lack of evidence for regularity in this cycle within exploited fish-stocks however [10], successful enhancement of commercial scale fisheries will require a substantial amount of empirical testing rather than relying primarily on theoretical evaluations. A pertinent example for Australian invertebrate fisheries is the ten years of research which preceded the commercial scale translocation of southern rock lobster (*Jasus edwardsii*) to enhance fishery yield [20]. This contrasts with the limited progress made from the “once-only” study on stock enhancement of the saucer scallop (*Amusium balloti*) [21]. Recent stock declines in *Amusium balloti* due to an anomalous environmental event, the marine heatwave [22], have re-ignited interest in enhancement for this species, however the sporadic nature of efforts into commercial-scale stock enhancement remain one of its major impediments to progress.

Enhancement of commercial invertebrate fisheries in Australia

Studies on invertebrate fisheries in Australia have been either small scale experiments focusing on one particular aspect of the enhancement principles, or modelling studies synthesising existing data into economic or ecological evaluations (Table 1). Analysis of growth and survival of hatchery reared animals have been the most common objectives [21,23,24] with particular emphasis on development of techniques for breeding and marking of hatchery reared animals [25]. Genetic evaluations have been undertaken [26,27] and investigations into habitat and ecological effects are well represented [28–31]. Six studies have evaluated the economic benefits of enhancement in Australian invertebrate fisheries. Chick [32] found limited support for a positive economic benefit in restocking of blacklip abalone populations in NSW, and identified the biological and economic limiting factors. Gardner and Van Putten [33] demonstrated that translocation of lobsters from slow to fast growing areas in Tasmania and subsequent biomass increase was economically viable if carried out at the appropriate scale. This eventually led to a commercial scale translocation programme to enhance fishery yields [20]. Hart et al. [34] predicted that enhancement was economically viable for greenlip abalone fisheries. Prince [35] investigated recovery in a blacklip abalone population after a virus induced mortality and found no scenario's in which enhancement was economically viable, but also that the assumed form of the underlying stock recruitment relationship (SRR) determined whether it was necessary or not. Ye et al. [36] conditioned an enhancement scenario for brown tiger prawns on an existing fishery model and found that there was only a 67% chance of enhancement being profitable. Under this outcome, the fishing company made a decision not to proceed to a full scale enhancement [8].

Table 1. Stock enhancement research into commercially harvested marine invertebrates in Australia.

Species	Class	Larval collection	Growth, survival, movement	Breeding, Genetics, and marking	Habitat and ecological effects	Bioeconomic analysis
Jasus edwardsii	Crustacea	[37]	[20,23,37–39]			[20,37,40]
Penaeus esculentus	Crustacea		[41,42]	[26,41–43]		[36,41,43]
Penaeus plebejus	Crustacea			[30]	[30,31,44]	
Holothuria scabra	Holothuroidea		[45]	[45–47]		
Amusium balloti	Mollusca		[21,48]	[25,27,48,49]		
Haliotis laevigata	Mollusca	[50,51]	[22,50–53]		[24,29,51,54]	[34]
Haliotis rubra	Mollusca	[56]	[32,35,56–61]	[59,60]	[61]	[32,35]
Trochus niloticus	Mollusca		[28,62–64]		[28,62]	

No major advance towards commercialising stock enhancement has resulted to date, however developments have occurred. First, the work of Tasmanian researchers on southern rock lobster [23,33,38,40] led to an annual commercial scale translocation of lobsters from slow to fast-growing areas, funded by ITQ holders, which resulted in an additional 50 tonne catch per annum [20]. Second, the work of abalone researchers [24,34,53] led to a significant attempt at commercialisation in Western Australia on greenlip abalone (*Haliotis laevigata*) in 2011/12. This attempt eventually failed due to the negative interaction between the aquaculture and wild fisheries sectors, particularly in the area of disease risk. Concern was further exacerbated by the lack of established policy principles that could integrate aquaculture and enhancement into wild fisheries management. A restocking and stock enhancement policy has since been developed for Western Australia [65], and two major biosecurity analyses and reports have concluded that disease risk could be sufficiently moderated [66,67]. This has resulted in the recent granting of Australia's first commercial scale sea ranching lease for *H. laevigata* in areas adjacent to commercially fished wild habitats, following successful experimental trials [54].

The newly established sea-ranching lease for *Haliotis laevigata*, which occurs under an aquaculture license, provides an illustrative example of the inability of current policy to support an enhancement fishery. The empirical and scientific evidence that established the viability of sea-ranching for this species arose from a decade of research into stock enhancement in wild stocks [29,34,53,68]. However, the wild fishing industry stakeholders, who initially supported enhancement, decided not to proceed with the commercialisation programme. Under the stakeholder-driven model of common property fisheries management, this was a decision by a sector body which needed to be upheld. However the proposed enhancement programme reappeared under a different administrative regime, with fewer of the safeguarding principles of the "Responsible Approach" [1,2] to guide management actions. A lack of clearly articulated policy in the "middle ground" between aquaculture (private ownership, total control of production) and wild fisheries (public ownership, no control of production) has been a significant roadblock to enhancement in this instance.

Challenge 1: The economic inequality of natural recruitment and stock enhancement

There is an economic inequality between natural recruitment and assisted recruitment (stock enhancement) which is relevant to the future of commercial scale fisheries enhancement. Currently, sustainable management in fisheries is achieved by restricting access to the fish until 'wild stock spawning biomass targets' are achieved, which in turn, create the future recruitment required to sustain annual harvests, as dictated by the stocks self-regeneration cycle. Management of major invertebrate fisheries in Australia is predicated on the assumptions of a reliable stock-recruitment-environment relationship for many species [14]. This reliability has arisen from both accurate measures of recruitment and predictions of future stock arising from that recruitment [69]. The economic value of natural recruitment is manifested in catch tonnages and subsequent income from the harvest and it operates in a highly defined and functioning free-market. Catch can be sold, access entitlements can be traded, and fishing licenses and vessels can be sold and bought. Operators in the industry can utilise the available economic information to make business investments on current and estimated future profitability.

In the case of stock enhancement however, there is no market-based mechanism in which its value can be objectively quantified because its products cannot be separated from wild catch, unlike aquaculture derived fish. This has led to calls for methods for genetic or physical tags [70], however the logistical difficulties of tagging millions of animals destined for an enhancement programme renders this a costly exercise. The lack of objective economic value has meant that estimates of resource rent (profitability), net present value (NPV), and other economic indicators, have only been derived from bioeconomic model predictions based on experimental studies with many assumptions [34–36]. The situation has led economists to conclude that stock enhancement is an activity that, '..*to* *an unusually great extent…,' s*uffers from the guidance of proper market signals and there is an urgent need of governance mechanisms that can induce the development of these [71,72].

Current bioeconomic analyses usually constrain the enhancement evaluation to limitations inherent in a particular stocks' regenerative model. This can be implicit, as in the ecological/logistic equation which underlies the modern economic theory of fishing [73], or explicit, as in the stock-recruitment-environment model of Olsen et al. [16]. The prevailing view is that unless it can be demonstrated that the stock is below the prescribed limits, as a result of recruitment overfishing, or other mortality, the benefits of stock enhancement are uncertain [18,70], or more succinctly, a waste of time and money [51]. This view presupposes that natural limits on wild stock productivity have already been attained through the well managed existing fisheries, even though it is increasingly accepted that these limits are not often well understood. An analysis of 230 fish stocks found that the majority of productivity regimes were constantly shifting, and only in 25% of stocks was there a 'repeatable relationship' between stock-size and long-term yield [10]. Given this lack of a 'repeatable relationship' it can be reasonably inferred that, in most situations, no prediction can reliably ascertain the response of a population to enhancement without empirical data to support it. This presents a challenge for effectively demonstrating commercial viability, and reinforces the present economic inequality between natural and assisted recruitment.

Bridging the inequality (1): the Patinopecten yessoensis *scallop fishery in Japan*

An historical example of an unpredictable change in population productivity from an enhancement program is available from the scallop fishery for *Patinopecten yessoensis* in Japan [74]. The fishery evolved three distinct phases in its history: First as a highly variable wild stock fishery producing between 8000 and 80,000 tonnes in the early 1900s; secondly as an overfished fishery with annual production limited to below 10,000 tonnes between 1945 and 1970; and finally to a >500,000 tonne enhancement fishery by 1995 [74]. This evolution was primarily the result of a successful enhancement solution to the intertwined issues of variable natural recruitment and overfishing, coupled with expansion of the production areas, or in ecological terms, a release from the habitat limitation which constrained the original productive capacity. The fishery has been formally recognised and accredited as a sustainably managed enhancement fishery by the Marine Stewardship Council [75].

When evaluating potential enhancement programmes, the current paradigm advises that an assessment model first be applied to the available data from a given fishery before undertaking a significant programme [1,70]. In the case of the *Patinopecten yessoensis* fishery, this data constitutes the 60 year time-series between 1910 and 1970. A brief analysis of the catch history data of this fishery reveals a salient lesson. Had it been undertaken according to the dictates of current thinking, i.e. by analysing the existing fishery data, the most likely outcome would have been predictions of a sustainable harvest level of considerably below 50,000 t. This is primarily because the historical catch distribution is typically log-normal, as it is for many scallop fisheries [76], with a maximum of 80,000 tonnes and a median around 20,000 to 30,000 tones. The prediction however, is an order of magnitude less than the capacity of the current enhancement fishery which is greater than 500,000 t [74].

Bridging the inequality (2): the Haliotis laevigata abalone fishery in Australia.

Greenlip abalone (*Haliotis laevigata*) is a commercially fished endemic mollusc on the South Coast of Australia, the subject of a moderately valued fishery with a GVP of around $20 million [77]. Evidence for a likely change in population productivity of *Haliotis laevigata* under commercial scale enhancement is available from comparisons of natural recruitment versus assisted recruitment from long-term enhancement experiments [24,29,34,53,68]. These studies showed that it was possible to increase recruitment in a controlled manner across multiple replicated experiments (n > 50). For example, in experimentally seeded habitats, median age 3 density was 2.2 per m^2 (n = 52), compared to 0.6 for wild stocks (*n* = 145) from the same area (Fig. 1). Overall, the rate of assisted recruitment was three times higher than natural recruitment.

For this situation, increasing the TAC (Total Allowable Catch) and lowering legal minimum lengths to compensate for the increased overall recruitment would be a valid management strategy that enhances the yield and GVP of the fishery. In the medium to long-term, an assisted recruitment programme of this nature will begin to replace natural recruitment as the biological and economic driver of the fishery. This is likely to occur even if natural recruitment rates are driven higher by the increased overall breeding biomass. This in turn will release pressure on the current management tools of wild stock sustainability (MSY, MEY, size-limits, effort, and catch restrictions), which are primarily about maximising yield, profitability, and/or egg production from a stocks uncertain natural regenerative cycle. The focus of wild stock management under this scenario will not be on enforcing catch quotas or minimal legal sizes, as these will be market driven. Instead wild stock sustainability in this instance will be about ensuring that the breeding and release programme of enhancement has both genetic and disease accountability to the highest level possible. However, unless policies can anticipate and plan for this fundamental shift in economic value of natural versus assisted recruitment, they are unlikely to be successful in promoting enhancement as a viable management tool

In summary, both the Japanese scallop and Australian greenlip abalone examples expose a conceptual hurdle regarding the management of stock enhancement in a fisheries context. That conceptual hurdle, which may well be termed the 'sustainability myth' [78], is that there is some pre-existing limit on the productivity of any given fishery, which good management practices have already attained, and that the population sustaining the fishery will naturally return to after perturbations. Therefore only a limited response can be expected by the addition of new recruits. This conceptual hurdle needs to be directly challenged with experimentation and observation if stock enhancement is to become a viable commercial practice. Limits to productivity are not understood well enough to assume compensatory behaviour is predictable.

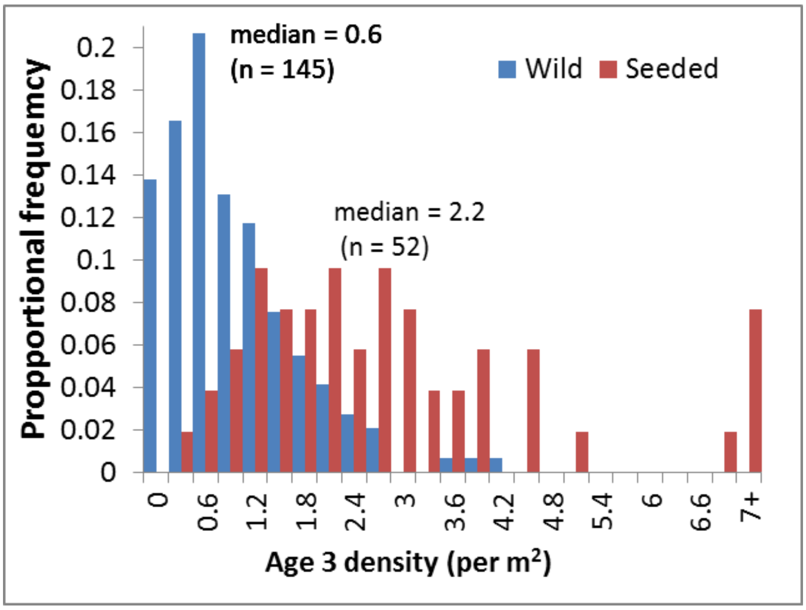


Figure 1: Proportional frequency distribution of recruitment of *Haliotis laevigata* (Age 3) in natural sites (wild), and sites subject to enhancement experiments (seeded).

Challenge 2. Self-replenishing natural populations: Genetically unique or evolutionarily transient?

In summarising a major review of restocking and stock enhancement in marine invertebrate fisheries Bell et al. [70] distilled the message of the responsible approach into two sequential objectives: (1) first understand the spatial extent of self-recruiting populations; (2) then demonstrate that assisted recruitment, whether it be releases of juveniles, or translocations of adults, will deliver greater benefits than other interventions, prior to the development of a major enhancement programme. In this section the relevance and definitions of self-recruiting populations from an evolutionary perspective will be examined. This issue is also considered an important roadblock on the road to commercial scale enhancement of invertebrate fisheries in Australia.

An important genetic objective of a responsible enhancement programme is to preserve the potential for adaptive evolution in the species by maintaining sufficient genetic variability in the wild stock. This objective can be achieved with a hatchery protocol that ensures a large number of broodstock and maximises individual male-female crosses. A good example of this is the chum salmon enhanced fishery in Japan, which releases 1 billion hatchery reared salmon annually, bred from 1.3 million spawners [79]. In management plans for threatened and endangered species, a minimum effective breeding population size of 500 has been recommended [80,81]. Hatchery programs that have a policy of "once-only" use of wild broodstock are the best suited to preserve existing variability. Even if relatively small numbers are used on an annual basis, say 100 animals, the cumulative result of a using a different set of 100 animals for each spawning event is a large effective population size well above limiting values of 500. This area of expertise has been well studied and Bell et al. [70] provide a summary of and direction to, the main literature.

There has been however, when advising on management protocols to achieve genetic targets, a clear preference towards identifying the self-recruiting stock unit as the primary method of genetic management [70], rather than controlling the breeding regime. This search for the 'genetic boundary' carries the implicit assumption identified earlier in this review, namely fishery management simply supports the innate capacity of "stock" to self-replicate and thus retain its identity and productivity. The idea reaches an advanced expression in discussions about genetic variability and management of Australian blacklip abalone (*Haliotis rubra*) stocks. It has been proposed that management has been unsuccessful in this species because it has been unable to focus at the correct genetic scale of the population [82] which, as a function of highly restricted dispersal, is composed of many independent substocks [83]. This view has been disputed; a recent review concluded that management over the past 40 years of this species has largely been successful, despite the 'mismatch' between genetic population scale and management units [77].

An alternative perspective on the importance of currently self-recruiting populations can be attained by considering the interaction between a particular species' ecology and its evolutionary history. Abalone (Haliotidae), for example, have always inhabited relatively shallow water (< 30 m), despite other invertebrates displaying an on-shore/off shore distribution pattern throughout their evolutionary history [84]. However analyses of the recent Pleistocene pre-history in the Australian region show that sea-level has been between 40 and 120 m below its current level for most of the last 180,000 years [85,86]. Consequently, it is likely that all populations of shallow water species with highly restricted dispersal (e.g. blacklip abalone) have become extinct and reformed numerous times over the past 180,000 years. From this perspective, current spatial boundaries of populations are a transient measure of connectivity, not a permanent index of a species potential for adaptive evolution.

To borrow another perspective, any suggestion that preservation of genetic uniqueness of human populations represented in current phenotypic expressions of skin and hair colour is more important than allowing the species to inter-breed and evolve is unlikely to be received as the latest wisdom. The documented rapid evolution of life history traits in response to high fishing pressure [87–89] serves to highlight the potential for well controlled breeding and/or release programmes in enhancement fisheries to counteract this rapid evolution. Evidence that genetic diversity of exploited populations has been altered by the fishing process already exists in commercially exploited abalone fisheries. Miller et al. [83] discovered different levels of genetic diversity between 'healthy' and 'collapsed' blacklip abalone populations and the main causal factor was the relatively high "migrant rate" in comparison to self-recruiting larvae, within collapsed populations. This is an example of 'natural genetic enhancement', but the exact same outcome could be achieved by a responsible breeding program. Lorenzen at al. [90] have summarised the latest developments in the practice of "breeding wild". Domestication is no longer an inevitable outcome of hatchery-programs; it is now a choice. Choosing to preserve the potential for adaptive evolution, to preserve "wild", can and must be made.

In an enhancement fishery, management will still be squarely focused on wild stock sustainability. Sustainability emphasis however will shift direction. Efforts directed at protecting productivity will decline, whereas efforts focused on managing genetic diversity and potential for disease introduction associated with movements of animals will increase. In particular, detailed policies which focus on the breeding and release regime to ensure that the potential for adaptive evolution in the species is preserved, will be necessary.

An integrated enhancement fishery for *Haliotis laevigata*

Introduction

Three main objectives of an enhancement program have been identified [4,72]. (1) Restocking – to restore depleted spawning biomass; (2) Stock enhancement – to optimise fishery biomass harvests; (3) Sea ranching – to optimise yield. These objectives are not mutually exclusive, however by virtue of governance arrangements, activities related to harvest in common property fishery resources generally fall under (1) and (2), whereas (3) tends to occur where exclusive access rights have been granted, such as an aquaculture lease. In a properly managed enhancement fishery however, there is nothing to preclude all three objectives from coexisting. A viable model for spatially explicit management in an enhanced invertebrate fishery is explored in this section. The model presents an integration of the three objectives of a fisheries enhancement programme implemented on a common property marine resource.

Governance frame works

For the purposes of simulating an appropriate governance framework for an enhancement fishery, it was assumed that the Minister for Fisheries in Western Australia, under powers vested in the Fisheries Act [91], excised a small proportion of habitat from the existing managed fishery and created a 'developing enhancement fishery' under a special exemptions. Exemptions from the 's.43 Order' (an order prohibiting fishing under the Fisheries Resources Management Act 1994) are the most common method for initiating developing fishery projects in Western Australia [92]. The intention is to develop an integrated enhancement fishery at a pilot experimental scale before scaling up to the area of the entire fishery. In the scenario examined, adherence to existing principles of resource access and sharing, as espoused in key policy statements [93], is an important objective.

Resource survey and allocation (Asset inventory)

The ecological asset/resource in question (32 ha of benthic habitat) was mapped out with a systematic survey design utilising relevant fishery independent survey methods [29,94]. Two 16 hectare blocks were apportioned into four general habitat categories (Sand, Seagrass, Reef, Abalone habitat) using methods developed by [29]. Each block was divided up into 4 ha areas and the production metrics and survey details for the 8 production areas are provided in Table 1.

In total, there was 6,200 m^2 of abalone habitat (0.62 hectares), out of the surveyed area of 32 ha, or 1.92% of the benthic ecological habitat. Areas of the other main habitat categories were 31% sand

(9.9 ha), 33% (10.6 ha) seagrass, and 36% was reef (11.5 ha). The existing population was 21,700 abalone (Table 1). The area of productive abalone habitat is then allocated to competing sectors (commercial, recreational, conservation) under the principles of integrated fisheries management [93], which is a key Government policy in Western Australia. This is an example where an enhancement programme can expand the tactical management tool box and provide opportunities to trade off different management interventions and competing interests [95].

The final allocation of abalone habitat was 91% commercial (5580 m^2), 6% recreational (320 m^2), and 3% (260 m^2) conservation (Table 1). Allocation between the commercial and recreational sector is based on historical comparisons of catch shares [93,96]. Allocation to the conservation sector is predicated on protecting a minimum viable population of 500 breeding adults [80]. To achieve this minimum viable population using the mean breeding stock density of 2.2 per m^2 [34] required >230 m^2 of abalone habitat. Many small scale marine protected areas are often viewed as the ideal conservation measure for sedentary invertebrate species with limited dispersal [97,98]. An enhancement fishery of this nature provides the opportunity to integrate conservation into fisheries management in more biologically relevant ways.

Resource enhancement - methods

Productivity and economic viability of the six production areas assigned for commercial scale enhancement (‘abalone paddocks’ - areas B, C, D, E, G and H; Table 1) was simulated under three different scenarios: (1) natural recruitment only; (2) natural recruitment with stock enhancement; (3) natural recruitment with ecological enhancement. The ecological enhancement scenario is an amalgamation of an enhancement program with the addition of abalone habitat. Hereafter termed Ecological Enhancement Units (EEUs; Fig. 2), the abalone habitat also received assisted recruitment. This approach is an integration of sea ranching as currently practiced in greenlip abalone [54], along with the type of enhancement program envisaged in [34]. For the purposes of minimising ecological impact, it was assumed the EEUs were deployed into a small proportion (10%) of the sand areas surrounding the subtidal reefs within the vicinity of existing abalone habitat. Total habitat area of EEUs for the ecological enhancement scenarios are shown in Table 2.

Two experiments were carried out to determine the potential enhancement capacity of the habitat. First, as an initial test on whether the chosen area was a viable candidate for commercial scale enhancement, the habitats were subject to a single enhancement with a release of 12,000 juveniles (Age 1) in 2011. Secondly, the key finding of the initial resource survey was that only 2% of the benthic area comprised abalone habitat (Table 1), and it resided within a mosaic of sand, sea grass, and low relief limestone and granite reefs that support productive algal, invertebrate, and finfish communities. This led to the hypothesis that the low profile EEUs described above (< 1 m height) could significantly enhance localised carrying capacity and result in substantial increases in algal and abalone production. Thus the second experiment involved the construction and release of two prototype EEUs into the natural reef mosaic (Fig 2a and 2b).

Economic analysis involved converting the costs of initial habitat surveys, culture and enhancement, to a per-harvest area (per m^2) approach. Data utilised in this conversion were the harvest costs (per kg) previously calculated [34], and estimates of legal-sized density and fishing mortality [99]. These translated into a fixed cost of harvest of \$1.58 per m^2. Other relevant economic information (value of harvest, cost of enhancement, discount rate) were derived from an earlier study [34]. Costs of building and deployment of the EEUs (\$ per unit; 1 unit = 7m^2 of habitat) were provided by Craig Kestal (pers.comm; Ocean Grown Abalone Pty Ltd).

Resource enhancement - results

A summary analysis of the change in greenlip abalone size frequency distribution over 2011 - 2014 shows the enhancement cohort entering the population over a two year period (Fig. 3). This single experiment showed that the existing habitat was suitable for an enhancement fishery and confirmed previous work involving the same methods on multiple releases [29,34,53].

Initial visual surveys of the enhanced habitat (EEUs) detected colonisation by wild adult and hatchery-bred juvenile abalone (Fig. 2c; Fig. 2d), invertebrates such as shrimps (Fig. 2e) and polychaete worms, and many fish species. One of the designs resulted in high settlement of red algal communities, the preferred food of Australian abalone [100] (Fig. 2f), and overall, this brief experiment showed that the localised carrying capacity could be increased.

Total spawning biomass across the six production areas prior to enhancement was 7.2 t, which resulted in an MSY of 1.1 t (Table 2). In comparison, an enhancement fishery resulted in a 300% increase in yield, from 1.1 to 4.2 t (Table 2). The ecological enhancement scenario, which involved deploying EEUs (Ecological Enhancement Units) over 10% of the sand area within each production unit, resulted in an overall doubling of abalone habitat, from 0.5 to 1.1 ha (Table 2).

Looking at economic indicators, GVP and resource rent (profitability) varied significantly between production areas in terms of natural production (Table 3). A full-scale ecologically enhanced fishery increased the unit GVP (per m^2) by 65% ($7.36 to $12.23 per m^2), and the total GVP by 200% ($43,000 to $138,000; Table 3). When scaled up to the area of the total Western Australian fishery, the increase in GVP would be from the current value of $8 million (Hart et al. 2014) to $23 million, or from $20 million [77] to $60 million for the entire Australian fishery.

The unit value of resource rent (profitability per m^2) declined in both the SE and the EE scenario, even though total profitability increased (Table 3). This represents the large capital input required to develop the enhancement capacity and highlights the trade-off between maximising current profits under the status quo, or future profits under a strategic development plan. More importantly however, the high variability in unit profitability ($ per m^2) in the natural production areas (e.g. $57.13 in Area E vs $2.21 in Area D; Table 3) shows the inability of a standard ITQ, fixed for the entire fishery and devoid of spatial information, to represent the true stock value. A spatially allocated enhancement fishery would facilitate evolution of ITQs towards values that more accurately represent the true productivity of a stock, and consequently, a deeper understanding of the value of habitats that support sustainable fisheries.

Importantly, the variability in production inherent in the natural population was reduced by the enhancement programme. Prior to enhancement, 60% of the GVP came from one area (Area E; Table 3), whereas in a full-scale ecologically enhanced fishery, it was more evenly spread. Area E was still the highest producing and most profitable area, but was only supplying 25% of the GVP. The area with the least amount of natural habitat, but largest amount of enhanced habitat was not economically viable as a production area in isolation of the others, but contributed substantially to the overall increased GVP and increased profit (Area B; Table 3).

In summary, an enhancement fishery created under these scenarios increased yield by up to 300% (1.1 to 4.1 t), GVP by 200% ($43,000 to $138,000), and profitability by 50% ($34,000 to $51,000). This economic improvement is relevant to the 32 ha surveyed. When considered at the scale of the entire Australian fishery (>10,000 ha), the opportunities of an integrated enhancement program are considerable.

Table 1. Biological characteristics and resource allocation for an integrated enhancement fishery for *Haliotis laevigata*. All eight production areas (32 ha) encompass the same area of benthic habitat (4 ha; 40,000 m^2) over a depth range of 10 – 14 m. The area surveyed is considered representative of the benthic habitat supporting the fishery (> 10,000 ha).

	Habitat distribution (%) within each production area				Ecological parameters and resource allocation		
Area	Sand	Sea grass	Reef	Abalone habitat	Abalone Habitat (m^2)	Existing Abalone Population	Resource Allocation
A	81	7	12	0.6	260	480	Conservation
B	53	29	18	1.0	380	160	Commercial
C	16	45	39	1.0	390	1,000	Commercial
D	19	40	41	2.7	1,060	1,890	Commercial
Total	42	30	28	1.3	2,090	3,530	
E	6	36	58	5.0	2,010	12,000	Commercial
F	34	35	31	0.8	320	870	Recreational
G	11	55	34	2.2	860	3,000	Commercial
H	31	14	55	2.2	880	2,300	Commercial
Total	20	35	45	2.8	4,070	18,170	
Overall	31	33	36	1.9	6,160	21,700	

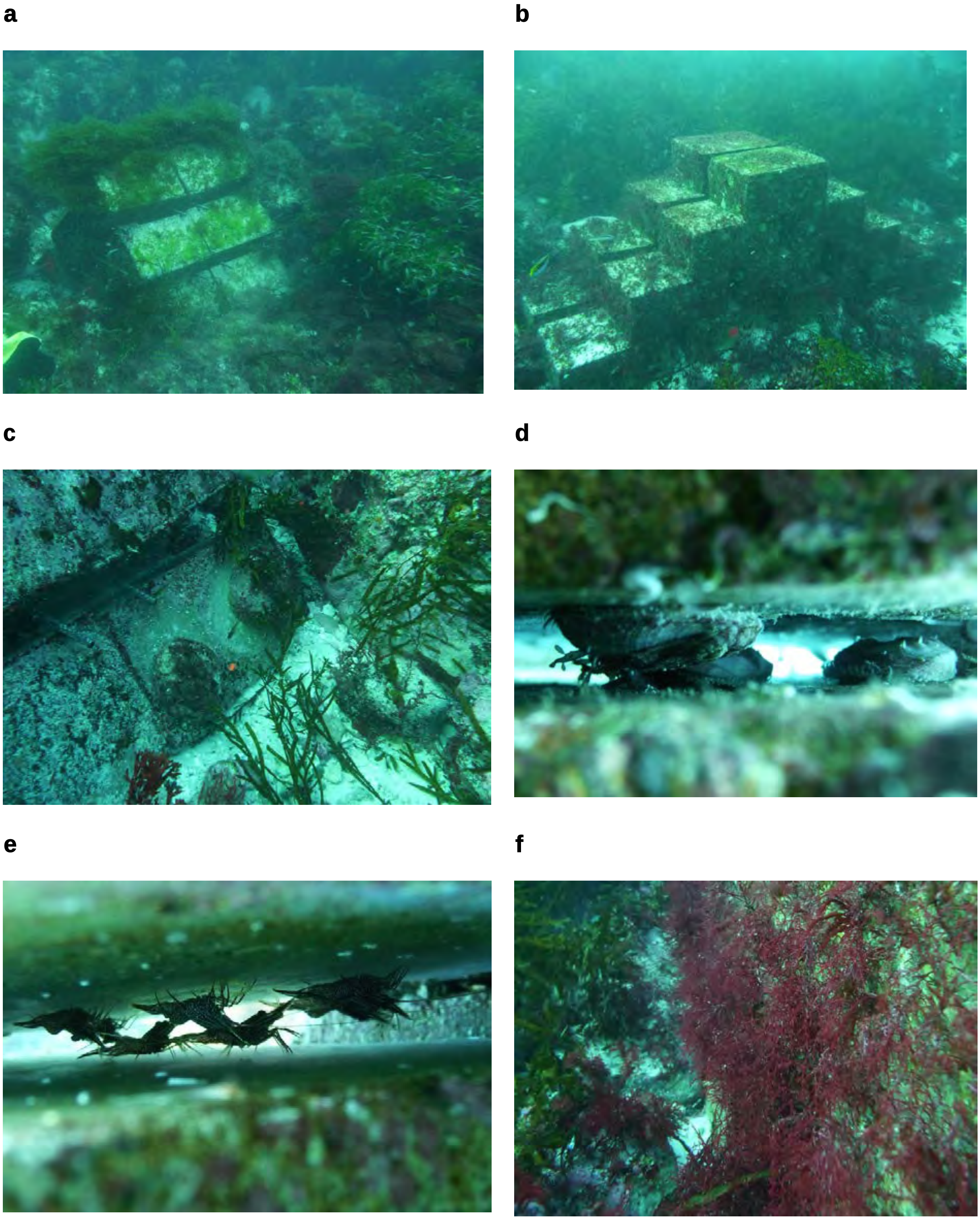
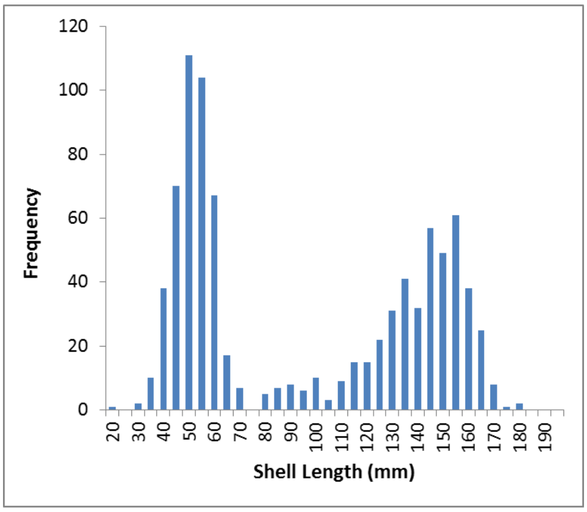


Figure 2: Prototype ecological enhancement units (EEUs) and associated ecological communities, as integrated into existing abalone habitats (a) Type 1 EEU, (b) Type 2 EEU, (c) adult and (d) juvenile greenlip abalone colonising a Type 1 EEU, (e) juvenile shrimp on a Type 1 EEU, (f) Red algae on a Type 2 EEU

(A)



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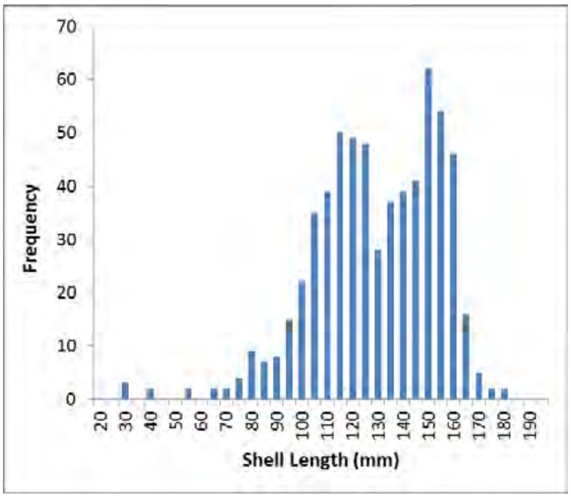

Figure 3: Size frequency distribution of *Haliotis laevigata* following a release of animals into natural habitat within experimental production areas (Table 2) in December 2011. (A) 6 months posts-release, (B) 24 months post release.

Table 2. Density, spawning biomass (kg), yields (kg), numbers released, and habitat created for an integrated enhancement fishery for *Haliotis laevigata*. Estimates based on a 32 hectare area of benthos which contained 0.5 ha of natural abalone habitat, to which was added 0.6 ha (860 × 7m^2) of ecological enhancement units (EEUs). Each EEU provided a habitat capacity of approximately 7 m^2.

	Pre-enhancement		#Post – enhancement scenarios							
Area	Density (per m^2)	Spawning biomass (kg)	Natural	Natural + SE	Natural + EE	Natural + SE	Natural + EE	Habitat created		
			Yield (kg)			Numbers released		# EEUs	Area (m^2)	% increase
B	0.42	60	10	14	800	100	18,100	336	2,400	620
C	2.57	350	50	90	450	600	4,900	104	730	190
D	1.79	670	100	160	590	1,100	6,200	122	850	80
E	6.00	4,200	640	900	1000	7,300	11,700	35	250	12
G	2.79	1,050	160	250	940	1,800	10,000	196	1,400	160
H	3.46	810	120	190	440	1,400	4,300	70	490	55
TOTAL	&3.67	7,200	1,080	1,604	4,220	12,300	55,200	860	6,120	110

Post enhancement scenarios [Natural, SE (Stock Enhancement), EE (Ecological Enhancement)] defined as follows:
<u>Natural:</u> MSY = 0.75*M*B, where M is natural mortality, B is spawning biomass. This formula assumes constant natural recruitment (Nr) only
<u>Natural + SE:</u> Yield = MSY + Stock Enhancement (SE) at a rate of 0.6 Nr; Methods for stock enhancement provided in Hart et al., (2013a; 2013b; 2013c)
<u>Natural + EE:</u> Yield = MSY + SE + Ecological Enhancement (EE). EE estimated from available sand area in each production area (10%; Table 2), and the total habitat area created by EEUs (Ecological Enhancement Units), when placed on the available sand areas. Trials for ecological enhancement using abalone designed habitats are found in [54].
&Total mean density is weighted by natural habitat area, so production units with a higher proportion of abalone habitat have a higher weighting

Table 3. Economic evaluations of an integrated enhancement fishery on 32 hectares of benthic habitat which supports *Haliotis laevigata*. Details of areas in Table 1.

Area	GVP ($ per m²)			Resource Rent ($ per m²)		
	Enhancement scenarios#			Enhancement scenarios		
	Natural	Natural + SE	Natural + EE	Natural	Natural + SE	Natural + EE
B	0.90	1.29	9.90	-(0.66)	-(0.50)	-(3.27)
C	5.43	7.80	12.19	3.85	4.87	3.67
D	3.79	5.45	9.50	2.21	2.92	3.00
E	12.66	18.18	17.80	57.13	69.30	11.18
G	5.94	10.51	13.00	5.74	7.10	4.65
H	4.61	7.92	10.28	3.94	4.97	4.32
&Mean	7.36	11.08	12.23	22.72	27.70	3.65
	GVP ($)			Resource Rent ($)		
B	340	490	23,300	-(260)	-(190)	-(7,700)
C	2,100	3,000	13,600	1,500	1,894	4,100
D	4,000	5,800	18,100	2,300	3,100	5,700
E	25,400	36,500	40,000	22,200	26,900	25,194
G	6,300	9,000	29,000	4,900	6,100	10,400
H	4,900	7,000	14,100	3,500	4,200	5,900
TOTAL	43,000	62,000	138,000	34,000	42,000	51,000

Enhancement scenarios: see Table 2 for description

&Means are weighted by habitat area, so production units with a higher proportion of habitat (Table 2), have a higher weighting

Resource enhancement - Discussion

An enhancement fishery of this nature addresses the key challenges identified in this review. Firstly, there is an explicit spatial dimension to management, i.e. the stock is uniquely defined by spatial delineation of the 8 production areas and measures of their production capacity such as area of natural abalone habitat or sand for deploying EEUs (Ecological enhancement Units) to enhance the habitat. This approach can be expanded to whatever scale is required. Secondly, the recruitment question is answered by scaling productivity according to the existing biomass and the pre-defined rules for the quantity of stock enhancement (60% of natural recruitment), and ecological enhancement (10% of the existing sand area) that can be undertaken. The parameter values of 60% and 10% are arbitrary initial values and, in commercial reality, would be optimised over time. The important point is that the initial limiting habitat area (0.5 ha) was changed to an increased area (1.1 ha), and combined with a significant stock enhancement component to create a greater production capacity in the fishery.

The release of juveniles into both natural habitats and the EEUs to increase recruitment highlights the need for explicit genetic management. Two complementary strategies were proposed. First, conservation of existing diversity based on concepts of minimum effective population size [80,81], and second, by ensuring best-practice “breeding wild” principles [95] adhered to for all juveniles destined for release back into the population. The first strategy is undertaken in the design phase of the enhancement fishery. Its practical application in this simulation study involved protecting around 5% of the existing abalone habitat to create a minimum viable population of breeding adults. When scaled up to the entire fishery in Western Australia, such a strategy would achieve protection of between 150 and 200 populations of a similar size. This conservation model of multiple smaller scale areas is an ideal strategy for sedentary populations with localised dispersal [97,98]. Designated populations could be kept as an entirely protected genetic resource, or provide an adequate residual buffer from which future wild stocks for breeding could be selected.

The second genetic management strategy is to implement best-practice “breeding wild” principles [90]. This is the area of policy most likely to benefit from the necessary shift in management effort required for an enhancement fishery. It is practically achievable for the relevant government agency to manage the wild broodstock collection and selection procedure for spawning. Each potential breeding animal can be individually tagged, sampled for genetic identity, and subject to a “once-only” use policy enforced by compliance tracking of the physical evidence (e.g. dead shell with tag number intact) and genetic evidence (e.g. genetic analysis of broodstock and progeny links at 6 months age). In the specific case for Australian abalone, governmental agencies and industry undertake stock surveys across the spatial extent of their fisheries. Collection and tracking of broodstock designated for enhancement programmes could be integrated into existing survey activities. Such policies will increase the ability of commercial hatcheries to implement best practice genetic management in all aspects of their breeding, not just for their enhancement programmes.

Using spatially explicit estimates of biomass and yield allowed for the economic value of enhancement to be explicitly evaluated in relation to natural recruitment, and the net result showed positive benefits in terms of yields, GVP, and profitability. Most important of all is the ability to control recruitment rates, and consequently productivity, of the fishery. While the economic viability will always be subject to the uncertainty inherent in stock-recruitment-environment relationships, predictive models linking known recruitment to future abundance, i.e. replacement rather than recruitment prediction [101,102], are generally more reliable than those which predict future recruitment from existing stocks [13,69]. Theoretical and empirical emphasis on the replacement equation [102], largely ignored in the literature to date [101], will become necessary to assist in developing viable enhancement programs.

The future of enhancement in Australian invertebrate fisheries

Ecologically and economically sustainable seafood production are major objectives in any commercial fishery. An enhancement fishery that promotes sustainability by the judicious addition of new recruits is better managed than one which relies only on uncertain natural recruitment. To arrive at an integrated enhancement fishery however, strategic goals are needed. The first is the development of the science itself, and this is primarily about defining the context. When pared down into its fundamental activity, fisheries enhancement is the manipulation of a species' abundance and its habitat in time and space, and is therefore an ecological discipline. A well utilised university textbook has, as its title, "Ecology: The experimental analysis of distribution and abundance" [103], and this succinctly describes enhancement science. Conceptualising stock enhancement as an ecological practice or as 'ecological enhancement' is valid and could mitigate some of the negative legacy associated with its history by shifting the focus towards a new positive discipline. Natural resource management agencies are far more likely to engage with positive process than negative legacies.

The practical example of an enhancement fishery given in this review has demonstrated the strategic approach. It has expanded the context of enhancement science by turning it towards an ecological viewpoint, for example using ecological and genetic principles to define the conservation measures within the enhancement fishery. It has also recognised the linked social and institutional/ governance issues involved in a common property access resource and presented a model based on existing principles and trade-offs of management measures. It has explicitly recognised the potential threat to wild stock diversity posed by enhancement programme and provided a two-pronged strategy to counteract it. However, it also posits that existing measures of population differentiation are not an absolute index of a species' potential to diversify, but a local expression in an open-ended evolutionary trajectory. The overriding objective must always be to maintain the potential for adaptive evolution, not to arbitrarily constrain genetic diversity to current states observed in extant populations. Local adaptation and hybridisation are synergistic, not antagonistic.

Re-establishing the context of stock enhancement in the field of ecological enhancement is a necessary and positive step forward for it recognises that renewable resources can be enhanced as well as depleted. It has been long established that there is an evolutionary tendency in animal populations to diversity, high-efficiency, and food chains of limited length [104]. Any activity that promotes one or more of these tendencies is desirable. In the case of enhancement, it is the high-efficiency tendency that is initially being promoted, with increased localised diversity sometimes a by-product if relevant structuring processes (shelter, water flow, spatial area) are altered favourably for a specific community over a sufficient time period. An Australian example of habitat modification, resulting in ecological enhancement via an increase in diversity of invertebrate species is found in a study of the Bandy-Creek boat harbour in Esperance [105].

The main question is "does the cost and risk associated with developing an enhancement capacity outweigh the benefits?" The long term answer must be yes, because understanding recruitment will inevitably lead to greater biomass, higher catch rates (greater economic viability), and greater capacity to both plan for, and respond to market demands. However, in the short to medium term, a vicissitude of issues precludes the majority of species from development in this area. These include unfavourable ecological profiles, low commercial harvest value, high costs of developing the knowledge base and technological capacity, and lack of policy framework with proven economic viability to safely and profitably manipulate stock abundance at a large scale. The onus therefore is to carefully select the species and to develop an appropriate long-term strategy to develop the fishery from one of catch only, to catch and enhancement. High-value sedentary invertebrate fisheries have long been recognised as suitable candidates for commercial scale stock enhancement [70,106], and the example given in this study provides a blueprint which could be adapted to other invertebrate species.

In summary, for a commercial wild fishery to evolve into an integrated enhancement fishery, governance mechanisms promoting this evolution must be cognisant of the following tenets:

(a) Sustainability can be arrived at using both natural recruitment and assisted recruitment, and the existing population equilibrium and harvest capacity is likely to change in response to the assisted recruitment.

(b) There is an economic inequality between natural and assisted recruitment, and a shift in value from natural to assisted recruitment is likely to evolve over time as increasing expertise and economic efficiencies combine to drive down the cost of enhancement.

Commercialising enhancement – beyond the Responsible Approach?

As an epigram, the 'Responsible Approach', first presented in 1995 [2], has been successful in condensing the disparate disciplines and complexities of enhancement science into an ethically-based focus and most major reviews within the last two decades have referred to its principles [1,3–8,70,107–109]. In business parlance, it is a successful commercial brand, with one major drawback. So far, no large-scale enhancement fishery has evolved out of its directives. This can be validly ascribed to the short evolutionary time it has been existence and complexity of the issues it has grappled with. Within Australian invertebrate fisheries, the leading candidate for development into a fully integrated enhancement fishery is *Haliotis laevigata*. Whether the eventual commercial practice will still be the "Responsible Approach" is the pertinent question, however the example presented in this review provides a clear road map of how the principles can be upheld in a commercialisation project. More broadly as a discipline however, enhancement science is poised to make further inroads into traditional fisheries ecology because of its evolving context and explicit recognition that enhancement is an intervention into complex and coupled ecological and socio-economic systems [95].

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Extension and Adoption

All three articles will be submitted to a fisheries policy journal (e.g. Marine Policy) for publication. Once the articles are published they will be widely disseminated to Australian fisheries researchers, managers and economists through *inter alia* universities, the FishEcon Network and the Australian Fisheries Management Forum (AFMF). It is envisaged that the articles would be used in fisheries masterclasses, workshops and presented at manager meetings as part of improving education and awareness of how to incorporate bio-economics and enhancement into fisheries management frameworks.

Project materials developed

Three journal articles, which are included in this report